ECONOMIC ANALYSIS OF THE CALIFORNIA TOXICS RULE

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EXECUTIVE SUMMARY

This document presents the U.S. Environmental Protection Agency's (EPA or the Agency) economic analysis of the California Toxics Rule (CTR), a regulatory action that establishes numeric water quality criteria for priority toxic pollutants necessary for the State of California to meet the requirements of the Clean Water Act (CWA).

BACKGROUND

Under the CWA, states have primary authority for establishing designated uses for water bodies and for developing water quality criteria to protect those designated uses. Under Section 303(c)(2)(B) of the CWA, whenever a state adopts new water quality standards, or reviews or revises existing water quality standards, it must adopt numeric water quality criteria for priority toxic pollutants (as defined by Section 307(a) of the CWA and for which the Agency has issued a criteria guidance document per Section 304(a) of the CWA) if the absence of such criteria could reasonably be expected to interfere with a designated use of a water body.

In April 1991, California adopted two statewide water quality control plans -- the Inland Surface Waters Plan (ISWP) and the Enclosed Bays and Estuaries Plan (EBEP) -- establishing water quality criteria for the state, in part, to comply with Section 303(c)(2)(B). In November 1991, EPA disapproved some portions of each plan. In December 1992, EPA promulgated the National Toxics Rule (NTR) (57 FR 60848, December 22, 1992) for several states that had not yet met the requirements of the CWA, including the State of California for those portions of the statewide plans that it had disapproved.

Shortly after the ISWP and EBEP were adopted, several parties filed lawsuits in State Court against the California State Water Resources Control Board (SWRCB) for not complying with state law when the two statewide water quality plans were adopted. In March of 1994, the State Court issued a final decision in a consolidated case requiring the SWRCB to rescind the two plans. The SWRCB took formal action to rescind the plans in September of 1994. Since then, the State of California has been without a complete set of water quality criteria for priority toxic pollutants because only the criteria promulgated by EPA in the 1992 NTR and criteria in existing Regional Basin Plans (issued by Regional Water Quality Control Boards) remain in effect. The CTR establishes the remaining criteria that will satisfy Section 303(c)(2)(B).

In California, the State is the National Pollutant Discharge Elimination System (NPDES) permit issuing authority. There are presently 184 major point sources of which 128 are publicly owned treatment works (POTWs) and 56 are industries that directly discharge to California's inland waters, enclosed bays, and estuaries. These major point sources may be impacted when the State implements water quality standards based on criteria in the final CTR. In addition there are 1,057 minor point source dischargers. These minor dischargers are not expected to incur significant impacts as a result of State implementation of CTR water quality criteria.

PURPOSE OF THE ANALYSIS

Under Executive Order (EO) 12866 (58 FR 51735, October 4, 1993), the Agency must determine whether a regulatory action is "significant" and therefore subject to the requirements of the EO [i.e., drafting an Economic Analysis (EA) and review by the Office of Management and Budget (OMB)]. EO 12866 defines "significant" as those actions likely to lead to a rule having an annual effect on the economy of $100 million or more, or adversely and materially affecting a sector of the economy, productivity, competition, jobs, the environment, public health or safety, or state, local, or tribal governments or communities (also known as "economically significant").

Pursuant to the terms of the order, EPA has determined that this final rule is not "significant." The CTR establishes ambient water quality criteria which, by themselves, have no impact or effect. In addition, the costs and benefits of the CTR could be negligible since implementation of permits under the CTR may not differ significantly from how the state may implement permits under current law.[1] However, EPA also acknowledges that, in the absence of the rule, current permit requirements and current effluent concentrations may continue in the future. In this case there may be a cost to some dischargers for complying with new water quality standards after those standards are translated into permit limits. Therefore, consistent with the intent of EO 12866, EPA developed this EA. EPA intends for the EA to inform the public about how entities might be affected by implementation of CTR-based water quality standards in the NPDES permit program.

The State of California has significant flexibility and discretion as to how it chooses to implement the CTR within the NPDES permit program. EPA's analysis assumes implementation procedures based on a combination of EPA guidance and current permit conditions for the facilities examined. This is appropriate because if the state does not adopt statewide implementation provisions, the CTR-based water quality standards would be implemented using existing state basin plan provisions and EPA regulations and guidance. However, a more precise measure of costs and benefits may not be known until the state adopts its implementation provisions.

This economic analysis develops revised estimates of the potential benefits and costs associated with implementing the CTR (EPA is promulgating the rule as final). EPA revised its analysis in response to comments on the EA that accompanied the proposed rule and to reflect more recent data and information to refine the analysis of benefits and costs. Wherever possible, the costs and benefits are expressed in monetary terms.

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[1] That is, the state could rely on its narrative toxicity standard and implementation of the standard using best professional judgment to set numeric water quality-based effluent limits for toxic pollutants in permits. Federal permit regulations (40 CFR 122.44(d)(1)(vi); 40 CFR 123.25) require that each permit contain effluent limits for toxic pollutants when a pollutant has reasonable potential to cause or contribute to an excursion above a state's narrative standard. The basis for such limits could include EPA's 304(a) criteria guidance or other appropriate scientific information. Therefore, this approach could result in permit limits that are nearly identical to those that would result from implementation of CTR-based numeric standards, which are also based on the latest available scientific information.

BASELINES FOR ESTIMATING BENEFITS AND COSTS

Analysis of the potential benefits and costs associated with implementing the CTR requires that a baseline be established. The baseline describes what would occur in the absence of a regulation and provides an initial starting point for measuring the incremental cost and benefit of regulatory compliance.

EPA established potential compliance costs under two scenarios of pollutant loadings from point source dischargers. For a low cost scenario, EPA established pollutant loadings based on effluent data, specifically, the maximum effluent concentration reported in the most recent three years of monitoring data. However, if this value exceeded an existing permit limit for a given pollutant (i.e., showing the facility to be out of compliance), EPA used the permit limit as the discharge concentration. As a high scenario, EPA established pollutant loadings based on effluent data or the existing permit limit if no data were available. EPA developed these scenarios to reflect the uncertainty associated with establishing existing levels of pollutants in the effluents of point source dischargers given the limitations of current analytical methods. That is, because the CTR criteria are often below current analytical detection levels, it can be difficult to determine whether a facility may face incremental costs under the rule.

Establishing a baseline for estimating potential benefits proved more difficult because EPA does not have information on water quality conditions that would result from implementation of all current regulations and programs designed to control toxic pollutants. Therefore, EPA established a water quality baseline using information on current conditions using California's Water Quality Assessment (WQA) database, a database developed and maintained by the SWRCB. The WQA database contains information on pollutants that adversely affect water quality in water bodies that have been evaluated, the sources of these pollutants, the beneficial uses impaired, and a rating of water quality. The WQA database used for the analysis was updated in 1994 and is described in detail in U.S. EPA (1996). To identify the extent to which California waters are impaired by the toxic pollutants addressed by the CTR, EPA relied on the WQA ratings of good, medium, and poor. EPA defined impaired waters for this analysis as those that are rated medium or poor for one or more toxic pollutants addressed by the CTR [although, as described in U.S. EPA (1996), an exact matching of the WQA database to the pollutants addressed by the CTR was not possible].

EPA also obtained baseline information related to potential benefits from California's Toxic Substances Monitoring Program (TSMP). The TSMP monitors the occurrence of toxic pollutants in California's waters through sampling and analysis of fish tissue and contains freshwater tissue samples collected throughout the state. Fish tissue contaminant levels also were obtained from EPA's 1992 National Study of Chemical Residues in Fish and from a 1994 study by the San Francisco RWQCB, Contaminant Levels in Fish Tissue from San Francisco Bay. These sources also are described in detail in U.S. EPA (1996).

ANALYSIS OF COSTS

The method used to estimate potential compliance costs associated with the final CTR is generally the same as that used to estimate costs of the proposed rule. However, to address comments raised during the public comment period, EPA gathered additional data and information to refine the analysis of potential costs and pollutant loading reductions. A large part of the effort was directed toward obtaining the most recent NPDES permits and effluent monitoring data. Efforts also were directed toward increasing the sample size of minor wastewater treatment plants and minor industrial facilities.

EPA's method involved developing detailed estimates of the potential impact of the CTR on a sample of point source dischargers to California's inland waters and enclosed bays and estuaries and then extrapolating these results to the universe of potentially affected facilities. The impact of the CTR will vary depending upon the procedures that will be used to implement the criteria. These procedures typically specify the methods for assessing the need for water quality based effluent limits (WQBELs) and, if WQBELs are required, the method for deriving WQBELs from applicable water quality criteria. For this analysis, EPA derived WQBELs using implementation procedures based on the methods recommended in the Technical Support Document for Water Quality-based Toxics Control (TSD) (U.S. EPA, 1991).

Where reasonable potential was determined for a pollutant at a facility, EPA calculated a projected CTR-based WQBEL in accordance with the TSD procedures. If the existing NPDES limit was more stringent that the CTR-based limit, then no cost or load reductions were assigned to the facility. However, if the CTR-based limit was more stringent than the existing NPDES permit limit, or, in the absence of a existing limit, if the CTR-based limit was more stringent than the maximum observed effluent concentration, EPA estimated the cost that the facility would likely incur to meet the more stringent limit. To estimate these costs, EPA established a decision framework to ensure consistency in selecting control options.

Changes to the Methodology

In analysis of the final CTR, EPA revised its methodology for calculating a projected effluent quality (PEQ) to address mathematical problems encountered due to limited data sets for some facilities. Based on a review of the available data, EPA determined that using one-half of the method detection level instead of "zero" for non-detects resulted in a more accurate coefficient of variation. Furthermore, if greater than 20 data points were available for a pollutant, the 99th percentile value was calculated from the data set to represent the PEQ.

Changes to the methodology also included revisions to the treatment process optimization costs and waste minimization/pollution prevention cost estimates. Because process modification costs are expressed as a range of values, the costs assigned for a facility were proportional to the flow to be treated and the loading reduction required. For waste minimization/pollution prevention, the cost for POTWs with a design flow greater than five MGD was increased from $400,000 to $2 million in the high scenario. This new cost, replacing the previous average per facility cost of $100,000, is the highest pollution prevention cost estimate derived by EPA in assessing of compliance costs resulting from implementation of the proposed Great Lakes Water Quality Guidance (SAIC, 1993).

EPA also eliminated the use of a cost “trigger” under the high scenario. In analysis of the proposed CTR, EPA used an industrial category threshold of $500/pounds-equivalent (lb-eq) for triggering compliance through regulatory alternatives. (The low scenario used a facility-specific threshold of $200/lb-eq for triggering compliance thru regulatory alternatives.) However, for the final rule, EPA assumed that no regulatory alternatives would be available under the high scenario. In other words, all necessary pollutant reductions were assumed to be achieved through either treatment or a waste control program of some type (e.g., waste minimization, pollution prevention).

Finally, EPA revised its estimate of the number of indirect dischargers that may be affected by more stringent CTR-based WQBELs applied to POTWs. EPA estimated that there are 2,144 SIUs that discharge to POTWs located on California inland surface waters and enclosed bays and estuaries. Previously, EPA assumed that 30% of these SIUs would be impacted under the low scenario and 10% would be impacted under the high scenario. Based on comments received indicating that the number of facilities affected was understated, EPA increased these percentages to 70% for the low scenario and 30% for the high scenario. (The percentage of SIUs impacted under the low scenario is greater than under the high scenario because the low scenario relies more heavily on source controls as a low-cost control option.) However, EPA believes that these revised assumptions are unrealistic and that the original 30% and 10% estimates more closely reflect the likely impact.

Results

EPA estimates that the potential annual cost of implementing the CTR is approximately $33.5 million under the low scenario and $61.0 million under the high scenario. As shown in **Exhibit ES-1**, indirect dischargers bear most of these costs in the low scenario. Under the high scenario, direct dischargers are expected to incur most of the potential costs. However, EPA believes that the high scenario likely overstates potential costs because it reflects the use of conservative (i.e., tending to err on the high side) assumptions. Specifically, under the high scenario, EPA estimates potential costs based solely on the presence of a WQBEL in a permit (as opposed to having monitoring data to show that the pollutant is actually there) and assumes that regulatory alternatives will not be used to mitigate excessive cost impacts.

Under the low scenario, where the baseline represents existing effluent concentrations, the expected reduction in pollutant loadings resulting from implementation of the CTR is approximately 1.1 million toxic lb-eq per year, or 50% of the baseline load of 2.2 million toxic lb-eq per year. Under the high scenario, the expected reduction in pollutant loadings is approximately 2.7 million toxic lb-eq per year, or 15% of the baseline load of 18.5 million toxic lb-eq per year.

Exhibit ES-1. Summary of Potential Annualized Costs (Millions of 1998 First Quarter Dollars)

Discharger Category	Low Scenario	High Scenario
Direct Dischargers	$9.9	$50.9
Indirect Dischargers	$23.6	$10.1
Total	**$33.5**	**$61.0**

ANALYSIS OF BENEFITS

EPA developed both a qualitative assessment of benefits and a quantified and monetized assessment of benefits.

Qualitative Assessment of Benefits

Toxics reductions under the CTR may provide ecologic benefits through increased ecosystem stability, resilience, and overall health. The potential benefits are difficult to quantify because of the complexity, scale, and uncertainties of the interaction of the multitude of ecological systems and toxics to be affected by the final rule. However, because of the extensive variety, proportion, and geographic area of the affected aquatic systems, the diversity and uniqueness of California ecological resources, and the large number of toxics to be regulated under the CTR, these benefits may be substantial, including (U.S. EPA, 1997):

- Reductions in toxics loadings that lead to improved conditions for California fish spawning and/or migration in bays/harbors and estuaries, lakes, rivers, streams, and saline lakes

- Reductions in bioaccumulative chemicals of concern that currently may affect fish and wildlife throughout the state, including selenium, mercury, PCBs, dioxins, and chlorinated pesticides

- Reductions in toxics that improve conditions for the successful recovery of federal and state threatened and endangered species, such as the delta smelt, desert pupfish, California brown pelican, bald eagle, California clapper rail, California tiger salamander, and western snowy plover

- Reductions in toxics that decrease adverse toxics-related impacts on aquatic and terrestrial wildlife in two important areas of California: the San Francisco Bay watershed and the Central Valley (see case studies in U.S. EPA, 1997)

- Reductions in the concentrations of both selenium and pesticides in the waters that feed the Salton Sea that may improve conditions for the restoration and maintenance of currently declining populations of wildlife, including threatened and endangered species such as the California brown pelican, peregrine falcon, bald eagle, Yuma clapper rail, and desert pupfish (see Case Studies in U.S. EPA, 1997)

! Improved water quality and associated improvements in survival, growth, and reproductive capacity of aquatic and aquatic-dependent organisms that will help restore and sustain California's ecological diversity.

Quantified and Monetized Assessment of Benefits

EPA's method for estimating the potential benefits of the CTR closely resembles the method used in evaluation of the proposed CTR (the results have been updated to incorporate the revised estimates of pollutant loading reductions and a slight modification to how the reductions are incorporated). EPA quantified and monetized three categories of potential benefits: (1) human health risk reductions, (2) recreational angling benefits, and (3) passive use values. However, in response to comments, EPA conducted additional searches to identify California-specific literature related to the contribution of point sources to the toxic-related water quality problems in California and the values held by California residents for reducing toxic contamination in the state's waters. EPA incorporated the results of these searches into this analysis. Also, where possible, EPA updated the data underlying the analysis.

Contribution of Point Sources to Total Toxic Loadings

EPA's method for estimating potential benefits generally involved estimating the value of eliminating toxic impairment from California waters and then determining the extent to which the potential loadings reductions associated with the CTR might contribute to that value. For this analysis, EPA assumed that there was a direct linear relationship between the estimated reduction in toxic-weighted pollutant loadings and potential benefits (although this assumption may or may not be correct). EPA also developed assumptions regarding the relative share of total toxic loadings to California waters that are attributable to point sources. These estimates are shown in **Exhibit ES-2** and represent the toxic-weighted average across the pollutants evaluated.

ES-2. Estimated Share of Total Toxic Pollutant Loadings Attributable to Point Sources for California Water Bodies

Water Body	Toxic Pollutant Loadings Attributable to Point Sources (%)
San Francisco Bay	1-10
Other bays and estuaries	42–64[1]
Freshwaters and saline lakes	3

[1] The lower-bound estimate is for nonurban bays and the upper-bound estimate is for urban bays.
Source: Based on EPA analysis of NOAA (1988a); NOAA (1988b); NOAA (1988c); Davis, et al. (1991); California RWQCB (1997); Central Valley RWQCB; and California 1994 WQA database, as originally presented in U.S. EPA (1997).

Human Health Benefits

EPA assessed the human health risks from the consumption of contaminated fish tissue, and the potential reductions in these risks for two populations of anglers: San Francisco Bay anglers and freshwater anglers in California. EPA conducted the assessment for San Francisco Bay anglers as a case study example of the health risks for anglers fishing in enclosed bays and estuaries. However, because only two other health advisories have been issued for enclosed bays and estuaries in California, this case study may represent an upper-bound estimate of baseline health risks associated with enclosed bays and estuaries.

EPA assessed baseline human health risks (cancer and systemic effects) based on reported contaminant levels in fish tissue samples collected from San Francisco Bay and freshwater fisheries throughout California. The approach used follows standard EPA methodology for estimating health risks as described in detail in U.S. EPA (1997). EPA then estimated the potential reduction in baseline risk levels that might result from implementation of the CTR, considering the relative contribution of point sources to the contamination problem.

Exhibit ES-3 presents the potential reductions in cancer risks for recreational anglers. EPA estimated reductions in statistical cancer cases for anglers with average consumption rates. The lower-bound estimate of reductions in statistical cancer cases is less than one because the lower-bound estimates of statewide loadings reductions for all carcinogens that accumulate in fish is small. Using an estimated value of a statistical life of $2.7 million to $9.6 million (American Lung Association, 1995, updated to 1998 first quarter dollars using the Consumer Price Index) and assuming that all cancers are fatal, potential human health benefits of reduced cancer cases in recreational anglers range from $0.10 million to $4.20 million per year.

Exhibit ES-3. Potential Human Health Benefits of Reducing Cancer for Recreational Anglers[1]

Water Body	Annual Reduction in Cancer Cases	Annual Monetized Benefits (millions of 1998 first quarter dollars)[1]
San Francisco Bay	0.04 - 0.05	$0.10 - $0.45
Freshwater Resources	0.44	$1.17 - $4.20

[1] Based on an average consumption rate (21.4 g/day) and a value of a statistical life of $2.7 million to $9.6 million (American Lung Association, 1995, updated to 1998 first quarter dollars using the CPI). Values based on the estimates of reductions in fish tissue concentration contamination. Note that there is currently a debate regarding the accuracy of the CPI.

Systemic (noncancer) risks are assessed by means of a hazard quotient (HQ) for each contaminant. A HQ of one or greater implies that chronic chemical exposures exceed EPA-established thresholds of toxicity, and are indicative of potential for adverse health effects. For PCBs, EPA expects the hazard quotient associated with the average consumption rate to be reduced from 2.26 to a range of 1.51-1.66 for San Francisco Bay anglers and from 1.40 to 1.01 for freshwater anglers. However, for high consumers (90th percentile), the HQ for PCBs is expected to be reduced from 11.31 to a range of 7.54-8.29 for San Francisco Bay anglers and from 7.02 to 5.04 for freshwater anglers.

EPA estimated that the HQ for mercury will be reduced for both the average and 90th percentile consumption rates, however baseline levels exceed 1.0 for high consumers only. For high fish consumers (90th percentile), EPA expects the HQ for mercury to be reduced from 3.77 to a range of 1.01-1.11 for San Francisco Bay anglers and from 3.12 to 0.90 for freshwater anglers.

Recreational Angling Benefits

In addition to health risks, concerns regarding adverse health effects from eating contaminated fish also may reduce the value of the recreational fishery because the ability to consume fish may be an important attribute of the overall fishing experience (Knuth and Connelly, 1992; Vena, 1992; FIMS and FAA, 1993; West et al., 1993). This reduction in value may occur because fewer fishing trips are taken or because the value of a trip is reduced. In addition, reduced toxic contamination may increase stability, resilience, and overall health of numerous ecosystems, which may increase catch rates as well as angling effort in California. Thus, the potential recreational benefits of the CTR may include an increase in the value of fishing experiences and an increase in participation.

Because the analysis of recreational angling value is conducted at the statewide level and does not consider numerous site-specific characteristics that will affect the level of benefits from the rule, the results are only intended to provide a rough approximation of the potential magnitude of recreational benefits. A case study approach would be required to more accurately characterize the anticipated angling benefits at any specific water body in California.

Increased Value of Existing Trips. EPA was unable to identify any studies regarding the value to California anglers of reducing toxic contamination of surface waters to help estimate the value of an improved fishing experience. However, a 1992 study of the Wisconsin Great Lakes open water sport fishing (Lyke, 1993) does reveal the significance of the contamination problem to the anglers. Lyke estimated the value of the Great Lakes trout and salmon fishing to anglers if it were "completely free of contaminants that may threaten human health." Lyke's estimates indicate benefits of 11% to 31% of the value of the fishery.

To transfer the Lyke results, EPA first estimated the number of fishing days in California that occur in toxic-impaired waters, distinguishing between water body type (e.g., freshwater river versus saltwater). Next, EPA multiplied the number of fishing days by an average consumer surplus for the different modes of fishing to obtain a baseline value of the fishery. EPA then multiplied by Lyke's estimate of 11% to 31% to obtain the value of a "contaminant-free" fishery. Finally, EPA multiplied by the midpoint of the low and high estimates of expected reduction in loadings and the low and high assumed contribution of point sources to total loadings to obtain the portion of these benefits that may be potentially attributable to point source controls. This approach results in potential benefits of between $1.53 million and $12.99 million per year.

Value of Increased Participation. In addition to increasing the value of existing angling days, reduced toxic loadings also may increase participation levels. Toxic contamination may discourage recreational fishing participation because of concern that consumption is unsafe. Similarly, knowledge of toxic contamination alone, regardless of consumption concerns, may reduce anglers' participation at a given site. Improving water quality to achieve toxic water quality criteria may restore this lost participation.

However, estimating lost participation is difficult and a limited number of studies have estimated reductions in participation due to water quality degradation. A thorough review of the literature revealed several studies that estimate the percentage of *people* that would take fewer trips, not the percentage decrease in angling days. However, these anglers are not expected to eliminate trip-taking. Therefore, using the various study results, EPA reasonably assumed that there may be a 5% to 10% reduction in trips attributable to poor water quality. Because public knowledge of toxic contamination varies across water bodies, EPA conservatively assumed a 5% increase in angler participation in estimating the benefits from increased angling participation for all waters except San Francisco Bay. Since a fish consumption advisory was issued for the Bay in 1994, EPA assumed a 10% increase in angler participation for the Bay.

EPA estimated the value of increased angling participation by multiplying the number of toxic-affected fishing days by 5% (10% for San Francisco Bay) and then valuing these days using estimated consumer surplus values. To estimate the portion of these benefits attributable to implementation of the CTR, EPA multiplied by the midpoint of expected reduction in toxic-weighted pollutant loadings and the relative contribution of point sources to total loadings. Potential benefits due to increased participation that may be attributable to the CTR range from $0.7 million per year to $2.2 million per year.

Nonconsumptive Wildlife Recreation Values

The *1996 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation* (U.S. Department of the Interior, Fish and Wildlife Service, and U.S. Department of Commerce, Bureau of the Census, 1998) indicates that 5.96 million California residents aged 16 or older participated in wildlife watching in 1996. This participation included 17.9 million trips away from home (at least 1 mile) for the primary purpose of observing, photographing, or feeding wildlife. These estimates do not include secondary wildlife-watching activities, such as observing wildlife while pleasure driving (U.S. Department of the Interior, Fish and Wildlife Service, and U.S. Department of Commerce, Bureau of the Census, 1998). Approximately 5.7 million California residents aged 16 or older also participated in wildlife-related activities around the home, including observing, photographing, or feeding wildlife.

Research has shown that nonconsumptive wildlife recreation (viewing wildlife) is highly valued. For example, Rockel and Kealy (1991) estimate a total annual value nationwide of between $8.7 billion and $165 billion in 1980 dollars (with the range of results indicating a sensitivity of their model to functional form). Cooper and Loomis (1991) estimated the total annual value for bird viewing in California's San Joaquin Valley to be $64.7 million (in 1987 dollars), based on willingness to pay (WTP) estimates for all Californians. Cooper and Loomis found that WTP increased as the number of birds seen increased, with diminishing marginal returns evident in their results (Cooper and Loomis, 1991).

As described in the EA that accompanied the proposed rule, CTR-related improvements in aquatic habitats may lead to healthier and more diverse populations of avian and terrestrial species and may manifest in increased participation and increased user day values for wildlife viewing activities. Without specific information as to the potential magnitude of changes in wildlife populations and thus viewing opportunities that may result from the toxic pollutant loading reductions anticipated under the rule, nonconsumptive wildlife recreation values cannot be estimated. Given the high baseline value, however, these benefits may be appreciable.

Passive Use (Nonuse) Values

Individuals may value reduced toxic concentrations in California aquatic environments apart from any values associated with their direct or indirect use of the resource. These passive use (nonuse) values are difficult to estimate in the absence of carefully designed and executed contingent value surveys. "Benefits transfer" techniques, however, can be used to develop a rough approximation of the potential magnitude of these passive use values.

Fisher and Raucher (1984) conducted an extensive review of the economics literature providing empirical evidence of the use and nonuse values associated with improved water quality and/or fisheries. Their review indicated that nonuse values are estimated to be *at least* half as great as recreational values. The authors concluded that if passive use values (for example, ecologic values) are applicable to a policy action, using a 50% approximation is preferred, with proper caveats, to omitting passive use values from a benefit-cost analysis. EPA believes his research is applicable to the CTR. Therefore, EPA estimated passive use values for the CTR as one-half of recreational fishing benefits thus estimating the amount of passive use value that recreational angling households are willing to pay, above their recreational use values, to preserve or enhance water quality. This suggests that the potential magnitude of passive use values associated with implementation of the CTR for households with recreational anglers may range from $1.1 million per year to $7.6 million per year.

EPA also estimated passive use values for nonangling households assuming that passive use values are 30% to 90% of the passive use values held by angling households. Assuming that there are 9.7 million nonangling households in California, EPA anticipates passive use benefits for nonangling households range from $2.3 million to $47.3 million per year.

Summary of Monetized Benefits

A summary of the estimated monetized benefits from implementation of the CTR is provided in **Exhibit ES-4**. Human health benefits are estimated for San Francisco Bay and statewide freshwater resources; all other benefits are estimated statewide.

Exhibit ES-4. Summary of Annual Benefits from Implementation of the CTR (Millions of 1998 First Quarter Dollars)

Benefit Category	Annual Value
Human Health (cancer risk)	
San Francisco Bay	$0.1 - $0.4
Other saltwater resources	+
Freshwater resources	$1.2 - $4.2
Recreational Angling	
Increased value of existing trips	$1.5 - $13.0
Increased participation	$0.7 - $2.2
Wildlife Viewing	+
Passive Use	
Households with recreational anglers	$1.1 - $7.6
Other households	$2.3 - $47.3
Omitted Benefits[1]	+
Total	$6.9 - $74.7

[1] Benefits not monetized include noncancer human health effects, water-related recreation apart from fishing, and consumptive and nonconsumptive land-based recreation.
+: Positive benefits expected but not monetized.

The benefits estimates are subject to a number of omissions, biases, and uncertainties. It is difficult to assess the overall impact of these factors on the estimates because the degree to which they might cause the estimates to be underestimated or overestimated cannot be predicted with accuracy. Among the key factors, however, the omission of potential benefit categories may have the most significant impact and would contribute to an underestimate of benefits. In particular, the inability to quantify and monetize the extent to which the CTR may enhance water-related recreation apart from fishing or consumptive and nonconsumptive land-based recreation, such as picnicking and hunting, may cause an underestimate of benefits. Although the scope of the benefits analysis has not allowed a quantitative assessment of these values at either baseline or post-CTR conditions, these benefits may be appreciable.

COMPARISON OF BENEFITS AND COSTS

A direct comparison of the monetized annual (steady-state) benefits of the CTR and annualized costs shows benefits and costs to be generally commensurate given the uncertainty in the analysis and that several categories of benefits are unmonetized. As shown in **Exhibit ES-5**, the estimate of monetized benefits ranges from $6.9 million per year to $74.7 million per year. Annualized costs are $33.5 million under the low scenario and $61.0 million under the high scenario.

Exhibit ES-5. Comparison of Annual Potential Benefits and Costs of Implementing the CTR (Millions of 1998 First Quarter Dollars)

Comparison Method	Monetized Benefits	Annualized Costs	
		Low Scenario	High Scenario
Direct Annual Comparison[1]	$6.9 - $74.7	$33.5	$61.0

[1] These monetized costs and benefits are not directly comparable since several categories of benefits have not been monetized.

Because the benefits and costs associated with implementation of the CTR may be characterized by an initial outlay of capital costs and a gradual phase-in of benefits, **Exhibit ES-6** presents a present value of benefits and costs over 30 years. This method applies a present value social accounting in which the streams of future benefits and costs are discounted to their present values to reflect society's rate of time preference. EPA considered two different phase-in scenarios to account for the potential delay in realizing benefits since many of the pollutants addressed by the CTR are persistent in the environment. To the extent that benefits of reducing toxic pollutants under the CTR are realized sooner, these scenarios may result in an underestimate of the present value of benefits. EPA assumed that there is a 7% opportunity cost of capital and that capital is replaced every 10 years. Since the life of capital typically exceeds 10 years, this assumption may result in an overestimate of costs. EPA calculated the present value of the streams of benefits and costs using discount rates of 3% and 7%.

As shown in Exhibit ES-6, discounted costs fall within the range of discounted benefits under the low scenario, but discounted costs exceed discounted benefits in three of the four cases shown for the high scenario. However, the assumption that capital is replaced every 10 years likely overstates costs. At the same time, benefits may be understated because some categories are not monetized and full benefits may be realized sooner than 10 or 20 years. Thus, EPA expects that the present value of benefits and costs is more commensurate than shown.

Exhibit ES-6. Comparison of Discounted Benefits and Costs of Implementing the CTR (Millions of 1998 First Quarter Dollars)[1]

Schedule of Benefits	Benefits[2]	Costs[3]	
		Low	High
3% Discount Rate			
10-Year Phase-In of Benefits	$108-$1166	$617	$1033
20-Year Phase-In of Benefits	$82-$883	$617	$1033
7% Discount Rate			
10-Year Phase-In of Benefits	$63-$683	$421	$767
20-Year Phase-In of Benefits	$45-$480	$421	$767

[1] Present values over 30 years.
[2] Benefits are phased in proportionately over 10 and 20 years, and have their full value in the remaining years. Benefits are not directly comparable to costs since several categories of benefits have not been monetized.
[3] Reflects capital costs plus a 7% cost of capital in years 1, 11, and 21, operating and maintenance costs in years 2 through 30.

CONCLUSIONS

Comparison of annual values of benefits and costs resulting from implementation of the CTR shows estimated costs falling within the range of monetized benefits. Comparison of 30-year present values of benefits and costs also shows costs under the low scenario to fall within the range of monetized benefits although costs under the high scenario generally fall just outside this range. However, EPA believes that benefits may actually be higher than shown because some categories of potential benefits have not been quantified or monetized. EPA was not able to quantify or monetize potential improvements in water-related recreation apart from fishing, such as boating, swimming, picnicking, and related in-stream and stream-side recreational activities. EPA was also unable to quantify or monetize potential improvements in wildlife viewing. Research indicates that wildlife viewing is a highly valued activity and that California residents value reductions in toxic pollutants that may affect wildlife resources. Thus, these omissions may result in an underestimate of benefits. In addition, using a capital life of 10 years likely overestimates potential compliance costs.

1.0 INTRODUCTION

This document presents the U.S. Environmental Protection Agency's (EPA or the Agency) economic analysis of the California Toxics Rule (CTR), a regulatory action that establishes numeric water quality criteria for priority toxic pollutants necessary for the State of California to meet the requirements of the Clean Water Act (CWA).

1.1 BACKGROUND

Under the CWA (33 U.S.C. 1251 et seq.), states have primary authority for establishing designated uses for water bodies and for developing water quality criteria to protect those designated uses. Under Section 303(c)(2)(B) of the CWA, whenever a state adopts new water quality standards, or reviews or revises existing water quality standards, it must adopt numeric water quality criteria for priority toxic pollutants (as defined by Section 307(a) of the CWA and for which the Agency has issued a criteria guidance document per Section 304(a) of the CWA) if the absence of such criteria could reasonably be expected to interfere with a designated use of a water body.

In April 1991, California adopted two statewide water quality control plans, the Inland Surface Waters Plan (ISWP) and the Enclosed Bays and Estuaries Plan (EBEP) establishing water quality criteria for the state, in part, to comply with Section 303(c)(2)(B). In November 1991, EPA disapproved some portions of each plan. In December 1992, EPA promulgated the National Toxics Rule (NTR) (57 FR 60848, December 22, 1992) for several states that had not yet met the requirements of the CWA, including the State of California for those portions of the statewide plans that it had disapproved.

Shortly after the ISWP and EBEP were adopted, several parties filed lawsuits in State Court against the California State Water Resources Control Board (SWRCB) for not complying with state law when the two statewide water quality plans were adopted. In March of 1994, the State Court issued a final decision in a consolidated case requiring the SWRCB to rescind the two plans. The SWRCB took formal action to rescind the plans in September of 1994. Since then, the State of California has been without a complete set of water quality criteria for priority toxic pollutants: only the criteria promulgated by EPA in the 1992 NTR and criteria in existing Regional Basin Plans (issued by Regional Water Quality Control Boards) remain in effect. The CTR establishes the remaining criteria that will satisfy Section 303(c)(2)(B).

1.2 PURPOSE OF THE ANALYSIS

Under Executive Order (EO) 12866 (58 FR 51735, October 4, 1993), the Agency must determine whether a regulatory action is "significant" and therefore subject to the requirements of the EO [i.e., drafting an Economic Analysis (EA) and review by the Office of Management and Budget (OMB)]. EO 12866 defines "significant" as those actions likely to lead to a rule having an annual effect on the economy of $100 million or more, or adversely and materially affecting a sector of the economy, productivity, competition, jobs, the environment, public health or safety, or state, local, or tribal governments or communities (also known as “economically significant”).

Pursuant to the terms of the order, EPA has determined that the CTR is not a “significant” rule. The CTR establishes ambient water quality criteria which, by themselves, have no impact or effect. In addition, the costs and benefits of the CTR could be negligible since implementation of permits under the CTR may not differ significantly from how the state may implement permits under current law.[1] However, EPA also acknowledges that, in the absence of the rule, current permits that do not contain numeric effluent limitations for toxic pollutants and current effluent concentrations may continue in the future. In this case there may be a cost to some dischargers for complying with new water quality based effluent limitations after those standards are translated into permit limits. Therefore, consistent with the intent of EO 12866, EPA developed this EA. EPA intends for the EA to inform the public about how entities might be affected by implementation of CTR-based water quality standards in the National Pollutant Discharge Elimination System (NPDES) permit program.

The State of California has significant flexibility and discretion as to how it chooses to implement the CTR within the NPDES permit program. EPA’s analysis assumes implementation procedures based on a combination of EPA guidance and current permit conditions for the facilities examined. This is appropriate because if the state does not adopt statewide implementation provisions, the CTR-based water quality standards would be implemented using existing state basin plan provisions, and EPA regulations and guidance. However, a more precise measure of costs and benefits may not be known until the state adopts its implementation provisions.

This analysis develops revised estimates of the potential benefits and costs associated with implementing the CTR. EPA revised its analysis in response to comments on the EA that accompanied the proposed rule and to reflect more recent data and information collected and used to refine the analysis of benefits and costs. These data and analyses are described in detail in the chapters that follow.

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[1] That is, currently, even without this rule, the state could rely on its narrative toxicity standard and implementation of the standard using best professional judgment to set numeric water quality-based effluent limits for toxic pollutants in permits. Federal permit regulations (40 CFR 122.44(d)(1)(vi); 40 CFR 123.25) require that each permit contain effluent limits for toxic pollutants when a pollutant has reasonable potential to cause or contribute to an excursion above a state’s narrative standard. The basis for such limits could include EPA’s 304(a) criteria guidance or other appropriate scientific information. Therefore, this approach could result in permit limits that are nearly identical to those that would result from implementation of CTR-based numeric standards, which are also based on the latest available scientific information.

1.3 STRUCTURE OF THE REPORT

This document identifies the need for the regulation, assesses potential costs and benefits, and analyzes alternative regulatory options. Wherever possible, the costs and benefits are expressed in monetary terms. The report is organized as follows:

- **Chapter 2, Need for the Regulation**, discusses the statutory requirement for the rule and the nature of the environmental problems caused by the presence of toxic pollutants in California waters that are regulated by the rule.

- **Chapter 3, Baseline for Estimating Benefits and Costs**, describes the baseline for analysis of the potential incremental benefits and costs of the rule.

- **Chapter 4, Analysis of Costs and Cost-Effectiveness**, describes the methodology for estimating potential costs and the results of the cost analysis.

- **Chapter 5, The Benefits Associated with the CTR: Methods and Concepts**, provides a discussion of concepts applicable to the analysis of benefits.

- **Chapter 6, Qualitative Assessment of Potential Ecological Benefits**, describes the types of ecological benefits anticipated to result from state implementation of the CTR.

- **Chapter 7, Benefits Methodology Issues: Contribution of Point Sources to Toxics-Related Water Quality Problems**, describes the method used to develop estimates of the potential contribution of point sources to toxic-related water quality problems.

- **Chapter 8, Quantified and Monetized Benefits Estimates**, presents the analysis of quantified and monetized benefits resulting from state implementation of the CTR.

- **Chapter 9, Comparison of Potential Benefits to Costs**, compares the potential benefits and costs estimated in the previous chapters.

- **Chapter 10** provides references.

- **Appendix A, Alternatives Analysis,** provides estimates of the potential cost and cost-effectiveness of two alternative regulatory options considered but not selected for the CTR: a less stringent human health risk level and applying toxic metals criteria in total recoverable form.

- **Appendix B, Compliance Cost Decision Matrix,** describes the assumptions EPA used to develop a forecast of how facilities may comply with limits established under the CTR.

! **Appendix C, Detailed Cost Estimates,** provides additional details on the estimated costs.

2.0 NEED FOR THE REGULATION

This chapter discusses the statutory authority for the CTR and the environmental factors that indicate a need for water quality criteria for toxic pollutants for California inland surface waters and enclosed bays and estuaries.

2.1 STATUTORY REQUIREMENT

Section 303(c)(2)(B) of the CWA requires states to adopt numeric water quality criteria for priority toxic pollutants for which EPA has issued Section 304(a) criteria guidance and whose presence could reasonably be expected to interfere with designated uses. Priority toxic pollutants are identified in Section 307(a) of the CWA. The CTR establishes numeric water quality criteria for priority toxic pollutants for the State of California that fulfill the requirements of Section 303(c)(2)(B).

EPA is promulgating this rule to fill a gap in California water quality standards. This gap resulted when the California Superior Court for the County of Sacramento ordered the SWRCB to rescind two statewide water quality control plans for inland surface waters and enclosed bays and estuaries. These plans contained water quality criteria for priority toxic pollutants for which EPA had issued CWA Section 304(a) criteria guidance. Thus, the State of California is currently without water quality criteria for many priority toxic pollutants as required by the CWA, necessitating this action by EPA.

When these federal criteria take effect, they will be the legally applicable water quality standards in the State of California for inland surface waters and enclosed bays and estuaries. The State will use these standards for all purposes and programs under the CWA. These criteria do not change or supersede any criteria previously promulgated for the State of California in the NTR, as amended (57 FR 60848, December 22, 1992, as amended by 60 FR 22228, May 4, 1995).

2.2 AN OVERVIEW OF ENVIRONMENTAL CONCERNS

The CTR addresses important environmental problems in California water bodies. Control of toxic pollutants in surface waters is necessary to achieve the CWA's goals and objectives. Many of California's monitored river miles, lake acres, and estuarine waters have elevated levels of toxic pollutants. Recent studies of California water bodies indicate that elevated levels of toxic pollutants exist in fish tissue; these discoveries have resulted in fishing advisories or bans.

Toxic pollutants covered by the CTR impair many of California's surface water resources. For this assessment, EPA has defined "impaired" waters as those that have been assessed and rated by the State of California as having medium or poor water quality for at least one toxic water quality pollutant or groups of pollutants. EPA has further defined "impaired" as meaning at least one designated use shows some degree of impairment. Information provided in this assessment, together with other data sources, indicates that toxic pollutants or groups of pollutants adversely affect large areas of surface water in California and their associated beneficial uses. According to U.S. EPA (1997), major impacts include the following:

- Available data suggest that over 800,000 acres of assessed bays, estuaries, lakes, and wetlands may be impaired by one or more toxic pollutants, as are over 3,700 miles of rivers. Most notably, more than two-thirds of the assessed area of both bays and saline lakes may be adversely affected by toxic pollutants.

- Inorganic pollutants such as metals and trace elements (particularly selenium) are the most significant categories of toxic pollutants affecting the water quality in assessed waters statewide. Pesticides are also associated with large areas of water quality impairment.

- On the basis of the areal extent of contamination and the uses of affected water bodies, San Francisco Bay and the Central Valley appear to be the areas most influenced by toxic contamination. In addition, toxic pollutants are responsible for impaired water quality in a high percentage of river and saline lake areas in the Colorado River Basin. These areas constitute those most extensively affected by toxic pollutants, but waters in all regions of California show some degree of impairment by toxics.

- Both point and nonpoint sources play a role in contributing to toxic pollution. Agriculture, primarily agricultural drainage, is the most frequently cited source of pollutants that impair rivers and is also frequently cited as a contributor to impaired lakes and reservoirs. Urban runoff and other nonpoint sources (e.g., deposition, spills) are most frequently cited as contributing factors to water quality problems in toxics-impaired bays. Mining also is a frequently cited source (mining operations may or may not be a point source), particularly for lakes and reservoirs, and toxic pollutants discharged by municipal wastewater treatment plants contribute to the impairment of a variety of water body types, particularly estuaries and wetlands.

- Currently, there are 12 fish consumption health advisories in waters covered by the CTR (9 inland water bodies and 3 enclosed bays and estuaries) because of high levels of contamination in fish tissue by mercury, PCBs, chlordane, dioxin, DDT, pesticides, and selenium. The advisories range from avoiding consumption of all species to limiting consumption of a few species to several meals per month. In addition, the state has four waterfowl health warnings for consuming waterfowl taken from the Grasslands area, Suisun Bay, San Pablo Bay, and San Francisco Bay based on elevated selenium levels in duck, greater and lesser scaup, and scoters.

3.0 BASELINE FOR ESTIMATING BENEFITS AND COSTS

An analysis of the potential benefits and costs associated with implementing the CTR requires that a baseline be established. The baseline describes what would occur in the absence of a regulation and provides an initial starting point for measuring the incremental cost and benefit of regulatory compliance. This chapter describes the baseline EPA established for analyzing the potential costs and benefits anticipated under the CTR. It also discusses other sources of toxic pollutants, and thus potential benefits and costs relevant to the CTR, that EPA did not address in this analysis.

3.1 POINT SOURCE DISCHARGES

The CTR establishes criteria for all designated priority toxic pollutants, except those addressed in the NTR, for California inland surface waters and enclosed bays and estuaries. EPA analyzed the potential benefits and costs for point source discharges to comply with these criteria. To do this, EPA used a sample of facilities to represent the universe of all California point source dischargers. To establish baseline permit limits, effluent concentrations, and controls, EPA obtained the most recent NPDES permit and monitoring data available for point source dischargers. Sources of monitoring data included EPA's Permit Compliance System (PCS), hard-copy Discharge Monitoring Reports (DMRs), the nine Regional Water Quality Control Boards (RWQCBs), and the sample facilities. EPA used the most recent 3-year period of data available for each facility to evaluate the potential impact of the CTR.

3.1.1 Baseline Effluent Concentration

EPA established baseline effluent concentrations for pollutants determined to have a reasonable potential to exceed the projected CTR-based limit under two scenarios. For a low scenario, EPA considered only those pollutants that had been detected in the effluent during the past 3 years. EPA then used the maximum effluent concentration reported in the most recent 3 years of monitoring data as the baseline pollutant discharge concentration. However, if this value exceeded an existing permit limit for a given pollutant, EPA used the permit limit as the baseline discharge concentration.

As a high scenario, EPA established baseline effluent concentrations as being equal to existing permit limits, whether or not the pollutant had been detected in the effluent. This typically provides an upper bound on discharge concentrations because most facilities discharge at a level below their current limit. However, if there was no permit limit for a priority pollutant, the maximum effluent concentration was used as the baseline for those pollutants found to have reasonable potential to exceed the projected CTR-based limit based on effluent data.

Thus, the low scenario provides a more likely scenario of the pollutants for which reasonable potential would be determined because it is based on actual data indicating that the pollutants are present in the effluent. In comparison, the high scenario provides an upper bound on the pollutants for which reasonable potential could exist. That is, pollutants are included in the high scenario analysis if the facility currently has an effluent limit for the pollutants even if there are no data to indicate that they are present in the effluent. There are many more pollutants considered in the high scenario than in the low scenario. One reason for this is that California's inland surface water plan (now rescinded) required permit limits to be established for all pollutants in the plan regardless of whether or not there was effluent data indicating their presence in the effluent. As a result, many facilities, including minor facilities, have limits for pollutants not detected in their effluent.

3.1.2 Baseline Pollutant Controls

EPA established the baseline for analysis of the pollutant controls necessary to meet projected CTR-based limits using the treatment systems in place at the sample facilities as described in the facility's NPDES permit. This baseline was used even in cases in which the existing maximum effluent concentration exceeded the existing permit limit at a facility. In theory, this assumption could result in overstating the pollutant controls necessary to meet the CTR-based limits if additional treatment were required to come into compliance with existing limits. However, in practice, only small differences were observed between current limits and maximum effluent concentrations.

3.2 WATER QUALITY

One of the most challenging analytic problems faced in estimating potential benefits attributable to implementation of the CTR is the need to account for the appropriate water quality baseline. A benefits analysis, for the most part, is able to measure improvements only from current or observable conditions. However, the appropriate baseline should account for water quality as it occurs assuming all current programs and legal requirements under the CWA, and other statutes or initiatives, are met. Therefore, there is an important distinction between current conditions and the conditions that reflect full implementation of existing programs.

An empirical approach to estimating the benefits relevant strictly to the CTR would be to estimate the reductions in toxic pollutant loadings from current conditions to the CTR-relevant baseline and then from this baseline to conditions following CTR implementation. EPA estimated pollutant loadings and reductions from the CTR-relevant baseline to conditions following implementation. These estimates indicate that the CTR may have a significant impact relative to loadings at its baseline. However, there is no empirical information with which to discern how this reduction compares to the difference between current conditions and conditions that reflect full implementation of existing programs.

Because EPA does not have information on water quality conditions under the CTR-relevant baseline, EPA established a water quality baseline using information on current conditions. EPA used several sources of information in this analysis, including California's Water Quality Assessment (WQA) database, a database developed and maintained by the SWRCB. The WQA database contains information on pollutants that adversely affect water quality in water bodies that have been evaluated, the sources of these pollutants, the beneficial uses impaired, and a rating of water quality. The WQA used for this analysis was updated in 1994. This analysis is described in detail in U.S. EPA (1997); the results are summarized briefly in Chapter 2.

To identify the extent to which California waters are impaired by the toxic pollutants addressed by the CTR, EPA relied on the WQA ratings of good, medium, and poor. Good quality waters are defined in the WQA as waters that support and enhance designated beneficial uses. Medium quality waters are those that support designated beneficial uses with occasional degradation and include waters suspected to be poor where available data are inadequate to allow a definitive conclusion. Poor waters are those that cannot reasonably be expected to support designated beneficial uses. EPA defined impaired waters for this analysis as those that are rated medium or poor for one or more toxic pollutants addressed by the CTR [although, as described in U.S. EPA (1996), an exact matching of the WQA database to the pollutants addressed by the CTR was not possible].

Another source of baseline information is California's Toxic Substances Monitoring Program (TSMP). The TSMP monitors the occurrence of toxic pollutants in California's waters through sampling and analysis of fish tissue and contains freshwater tissue samples collected throughout the state. Fish tissue contaminant levels also were obtained from EPA's 1992 National Study of Chemical Residues in Fish and from a 1994 study by the San Francisco RWQCB, Contaminant Levels in Fish Tissue from San Francisco Bay. These sources also are described in detail in U.S. EPA (1996).

3.3 COSTS AND BENEFITS NOT ANALYZED

Although in this analysis EPA focused on estimating benefits and costs from controlling point source discharges from NPDES-permitted facilities, EPA believes that a comprehensive watershed approach that addresses all significant sources of problem pollutants may present a more cost-effective compliance approach and may be necessary to achieve water quality standards. However, the total costs of actions necessary to implement a watershed approach in a given area can be adequately estimated only after an in-depth site-specific study of the water body. Therefore, the total costs estimated in this analysis may not result in full attainment of water quality standards in all California water bodies. Accordingly, the benefits estimated here include only those that may occur as a result of loadings reductions from point sources typically subject to numeric water quality-based effluent limits (WQBELs).

The state may ask or require other sources to implement best management practices (BMPs) or to participate in a comprehensive watershed management planning approach. Control strategies for wet weather discharges and nonpoint sources are an important piece of EPA and California's current overall strategy to improve water quality. Many of the programs developed to control wet-weather discharges and nonpoint sources are already in place. Costs due to these programs already have been incurred or will soon be incurred owing to existing federal, state, and local environmental programs. The categories of nonpoint sources and wet weather discharges that are likely to contribute to toxic impairment of water bodies, yet are not always subject to numeric effluent limits, are described below. Programs to control these types of pollution that need to continue if all California waters are to ultimately meet water quality standards also are described.

3.3.1 Agricultural Runoff

Agriculture is one of the largest sources of pollutants in California. Toxic water quality problems result from the application of fertilizers and pesticides and from the discharge of used irrigation water. Pesticides and fertilizers are carried into water bodies via rain and soil erosion. Irrigation water must be drained from fields resulting in the discharge of pesticides, selenium, metals, and other trace contaminants. This irrigation drainage must be transported to holding ponds, evaporation ponds, local water bodies, or reintroduced to the local irrigation system.

As a result of existing federal and state laws, much research and time has been spent attempting to alleviate the difficult problems caused by agricultural runoff. Unlike most point source discharges, polluted runoff from agricultural lands cannot be effectively diminished by treatment systems. Instead, controls focus on reduction in the use of pesticides and changes in the use of water and land. Improvements in irrigation techniques and the reuse of drainage water on salt-tolerant plants can reduce the amount of polluted drainage. Retirement of agricultural lands that have high levels of salts is another alternative to reducing polluted drainage.

3.3.2 Inactive and Abandoned Mines

California has more than 15,000 inactive, abandoned mines. Only six major NPDES permits have been issued for mine discharges. Although 27 additional mines have been issued minor NPDES permits, the vast number of inactive and abandoned mines is not currently permitted. Acid mine drainage results in the discharge of metals such as cadmium, copper, lead, mercury, and zinc.

Technologies used to control mine discharge include prevention and treatment. Prevention may include diversion of local streams away from reactive material, covering reactive mine waste, mixing reactive waste with limestone to buffer acid, disposing of reactive mine waste underwater to eliminate reaction with air, impounding mine drainage to keep it from entering surface waters, and sealing the mine portal to flood the mine, which suppresses the formation of acid mine drainage. Treatment methods vary depending on the site and extent of pollution and involve control of the mine drainage before it enters surface waters. Treatment techniques include chemical precipitation, ion exchange, construction of wetlands, and evaporation of mine discharge in surface impoundments (California State Water Resources Control Board, 1995).

Efforts are already under way to clean up some mine sites under existing state and CWA requirements (storm water regulations) and Superfund [the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA)]. However, in order to reach full compliance with water quality standards, additional assessment and treatment may be necessary for some California water bodies.

3.3.3 Urban Runoff

Urban runoff in California has been shown to be a significant contributor to water quality problems. Urban runoff is currently regulated as an NPDES point source for large towns and as a nonpoint source for medium and small towns. Most cities in California have separate systems for wastewater, which handle normal used water flows, and storm drainage, which divert storm water to prevent flooding. When rainfall picks up pollutants, such as toxic metals and pesticides that accumulate on the ground, storm water drains can carry harmful amounts of these pollutants into rivers, lakes, and bays.

Programs designed to control storm water pollution stress BMPs and also often emphasize pollution prevention (e.g., street cleaning or the reduction in use of pesticides and fertilizers) and public education. Public outreach is designed to address proper use, storage, and disposal of household chemicals, pesticides, oil, and other wastes. Efforts to control urban runoff through BMPs are also under way through both NPDES storm water permits and through nonpoint source planning. For example, under its existing NPDES storm water permit, the cities and counties of the Los Angeles area plan to spend $15 million annually on public education and a program to curb illegal dumping (California State Water Resources Control Board, 1996).

Since some of the sources discussed above are exempt from federal permitting requirements, the State of California must develop alternative strategies and controls to protect or restore water quality that is affected by these sources. The State of California established three general management approaches to address nonpoint source problems in its 1988 Nonpoint Source Management Plan, including voluntary implementation of BMPs, regulatory-based encouragement of BMPs, and waste discharge limitations. In most cases, the RWQCBs decide the mix of options that will be used to address any given nonpoint control problem.

4.0 ANALYSIS OF COSTS AND COST-EFFECTIVENESS

This chapter presents the analysis of the potential costs resulting from implementing the CTR. The method used to calculate costs is generally the same as that presented in SAIC (1997). However, EPA made some changes to the sample and the methodology in response to comments and to improve the analysis. The methodology is described in Section 4.1. Changes to the analysis since proposal of the draft CTR are explained in Section 4.2. The results of the analysis, including costs, pollutant loading reductions, and cost-effectiveness by discharger category, are presented in Section 4.3. Section 4.4 summarizes the sources of uncertainty in the cost analysis.

4.1 METHODOLOGY

As described in SAIC (1997), to estimate potential costs and pollutant loading reductions attributable to implementation of the proposed CTR, and for the final CTR, EPA developed detailed estimates for a sample of point source dischargers to California's inland waters and enclosed bays and estuaries, and then extrapolated these results to the universe of potentially affected facilities. The population of NPDES-permitted facilities that discharge into inland surface waters and enclosed bays and estuaries in California includes 184 major dischargers and 1,057 minor dischargers. EPA selected 16 major dischargers and 11 minor dischargers to represent the various discharger categories and geographic distribution of the universe of facilities.

EPA then estimated the potential impact of the CTR on these sample facilities, as described below. The impact of the CTR will vary depending upon the procedures that will be used to implement the criteria. These procedures typically specify the methods for assessing the need for WQBELs and, if WQBELs are required, the method for deriving WQBELs from applicable water quality criteria. For this analysis, EPA derived WQBELs using implementation procedures based on the methods recommended in the Technical Support Document for Water Quality-based Toxics Control (TSD) (U.S. EPA, 1991).

4.1.1 Method for Determining Reasonable Potential to Exceed CTR Water Quality Criteria

The NPDES permit regulations in 40 CFR 122.44(d) and 123.25 require that WQBELs be derived for toxic pollutants that are discharged at a level that has a reasonable potential to cause or contribute to an exceedance of water quality standards. To determine whether there is reasonable potential with respect to projected CTR-based limits, EPA followed several conventions. First, for those toxic pollutants for which limits are already established in a facility's current permit, EPA assumed that a reasonable potential existed. Second, for those pollutants with no limit in the existing permit but that were detected in the effluent (as reported in the permit application or as a result of monitoring conditions contained in the NPDES permit), EPA determined reasonable potential using the method recommended in EPA's TSD, described below. Finally, if all monitoring data for a facility were reported as below analytical detection levels, even if the reported detection limit was above EPA-approved analytical method detection levels, EPA assumed that no reasonable potential existed to exceed CTR-based WQBELs.

Where data indicated that a pollutant was present, but no current permit limit had been developed, EPA calculated the projected effluent quality (PEQ) and compared it to projected CTR-based WQBELs for all pollutants of concern. A PEQ is an estimate of the maximum effluent pollutant concentration that is derived from actual measured values taking into account statistical uncertainty. EPA calculated a projected CTR-based WQBEL using all applicable CTR criteria (based on protection of aquatic life and human health) and compared the PEQ to the most stringent of the calculated CTR-based WQBELs. If the PEQ exceeded the projected CTR-based WQBELs, EPA concluded that there is reasonable potential to exceed a CTR-based WQBEL. Pollutants for which EPA determined reasonable potential existed were then analyzed to determine potential controls necessary to achieve the CTR-based WQBEL.

4.1.2 Method for Estimating Potential Costs

Where reasonable potential was determined for a pollutant at a facility, EPA calculated a projected CTR-based WQBEL in accordance with the TSD procedures. If the existing NPDES limit was more stringent than the CTR-based limit, then no cost or load reductions were assigned to the facility. However, if the CTR-based limit was more stringent than the existing NPDES permit limit, or, in the absence of an existing limit, if the CTR-based limit was more stringent than the maximum observed effluent concentration, EPA estimated a cost that the facility would likely incur to meet the more stringent limit.

To estimate costs, EPA performed an engineering analysis of how each sample facility could comply with the projected CTR-based effluent limits. The costs to meet the projected CTR-based limit were performed under two different costing scenarios using a “compliance cost-decision matrix,” developed to predict how a facility would likely achieve the requisite pollutant reduction. EPA developed this matrix to ensure consistency in estimating the general types of controls that would be necessary to comply with the CTR’s more stringent requirements, as well as to integrate other alternatives into the cost analysis. The matrix establishes specific rules to provide reviewing engineers with guidance in consistently selecting options. This matrix is presented in Appendix B.

Under the decision matrix, EPA first considered costs for relatively minor treatment plant operation and facility changes. EPA considered minor, low-cost modifications or adjustments of existing treatment feasible if the literature indicated that the existing treatment process could achieve the revised WQBEL and if the additional pollutant reduction was relatively small (e.g., 10% to 25% of current discharge levels).

When it was not technically feasible to simply adjust existing operations, EPA considered a control strategy of waste minimization/pollution prevention. However, EPA estimated costs for these controls only when they were considered feasible based on the reviewing engineer’s understanding of the treatment processes at a facility. The decision matrix established several rules of thumb for this determination. These rules considered the level of pollutant reduction achievable through waste minimization/pollution prevention techniques, the appropriateness of waste minimization/pollution prevention for the specific pollutant, and knowledge of the manufacturing processes generating the pollutant of concern. In general, detailed treatment and manufacturing process information was not available in NPDES permit files; therefore, the assessment of feasibility was based primarily upon the reviewing engineer's best professional judgment using general knowledge of industrial and municipal operations.

If EPA determined that waste minimization/pollution prevention alone was not feasible to reduce pollutant levels to those needed to comply with the projected CTR-based WQBELs, EPA estimated costs for a combination of waste minimization/pollution prevention, simple treatment, and/or process optimization. If these relatively low-cost controls could not achieve the CTR-based WQBELs, EPA estimated costs for more expensive controls (e.g., end-of-pipe treatment).

Development of end-of-pipe treatment cost estimates began with a review of the existing treatment systems at each facility. EPA considered its Office of Research and Development, Risk Reduction Engineering Laboratory's "RREL Treatability Database" (Version 4.0) in determining the need for additional or supplemental treatment. The pollutant removal capabilities of the existing treatment systems and/or any proposed additional or supplemental systems were evaluated based on the following criteria: (1) the effluent levels that were being achieved currently at the facility; and (2) the levels that are documented in the EPA "RREL Treatability Database." If this analysis showed that additional treatment was needed, EPA estimated capital and operating and maintenance (O&M) costs for unit processes that would achieve compliance with the projected CTR-based effluent limits using the same documentation.

Finally, for a low cost scenario (described below), EPA considered the relationship between the cost of adding the treatment and other types of remedies or controls following the calculation of end-of-pipe treatment costs. Specifically, if the estimated annualized cost for removing a pollutant exceeded a prescribed value [expressed as dollars per pound equivalent ($/lb-eq)] for a facility, EPA assumed that a discharger would use alternative regulatory approaches to comply with CTR-based effluent limits as long as the cost of these options was less than the anticipated pollution control costs. EPA referred to the prescribed value as the "cost trigger." In these situations (this was the case for two pollutants at one sample facility), EPA estimated the cost for special studies or monitoring that may be required to pursue the regulatory alternative instead of treatment costs and did not anticipate pollutant loading reductions at the facility.

The types of alternative regulatory approaches assumed available for dischargers in California include phased total maximum daily loads (TMDLs), water quality standard variances, site-specific criteria, change in designated use, translators for metals, and alternative mixing zones. Whether these options are available to dischargers or not depends on how the State chooses to implement the CTR. EPA accounted for the potential use of such alternatives by employing the cost trigger as part of a low cost scenario. However, to ensure that costs are not underestimated if these alternatives are not available, EPA developed a high cost scenario that does not employ the cost trigger. If the State does not make alternative regulatory approaches available to dischargers, EPA believes that the potential cost impact will lie between the low and high scenario estimates.

Cost Scenarios

Since states are not required to use the methods recommended in the TSD, implementation procedures can vary, and may result in more or less stringent WQBELs. Because of the uncertainty regarding the State of California approach to implementation at this time, EPA developed a range of costs to represent the potential range of impact of the CTR based on certain implementation assumptions. The upper and lower bounds of these cost assumptions are referred to as the "high" and "low" cost scenarios, respectively. The principal differences between these scenarios are described below.

EPA established the baseline for analysis of the pollutant controls necessary to meet projected CTR-based limits using the treatment systems in place at the sample facilities as described in the facility's NPDES permit. This baseline was used even in cases in which the existing maximum effluent concentration exceeded the existing permit limit at a facility. In theory, this assumption could result in overstating the pollutant controls necessary to meet the CTR-based limits if additional treatment were required to come into compliance with existing limits. However, in practice, only small differences were observed between current limits and maximum effluent concentrations in these cases.

Low Scenario

For a low scenario, EPA calculated baseline loadings and CTR-based WQBELs and loadings only for those pollutants that had been detected in the effluent at sample facilities for the most recent 3 years of data since 1993 and that failed the reasonable potential test.

EPA developed a low cost scenario to reflect a lower baseline loadings estimate and a more flexible State implementation approach than the high scenario. The assumptions used for the low scenario result in an estimate based on a smaller number of affected pollutants and a lower amount of incremental pollutant removals necessary to comply with CTR-based effluent limits (as compared with the high scenario). The assumptions used for the low scenario are:

- In the absence of any monitoring data, EPA assumed that no costs would be incurred, even if a permit limit exists that is less stringent than the CTR-based limit. That is, EPA assumed that if a facility is not monitoring for a pollutant, it is not likely to be present in the effluent.

- As described more fully above, if the estimated annualized cost for removal of a pollutant exceeded a cost trigger of $200 per toxic pound-equivalent, EPA assumed that dischargers would use alternative regulatory approaches (as long as the cost of such options was less than the cost of pollution control). In these situations, EPA estimated the cost for alternative approaches and did not anticipate any pollutant loading reductions.

High Scenario

For a high scenario, EPA established baseline loadings and calculated projected CTR-based WQBELs and loadings for all pollutants for which limits had been established in existing NPDES permits (whether or not data indicated that these pollutants were present in the discharge). EPA also established baseline loadings and calculated projected CTR-based WQBELs and loadings for all other CTR regulated pollutants that were detected in the effluent and failed the reasonable potential test.

EPA developed the high cost scenario to reflect a higher baseline loadings estimate and a less-flexible state implementation approach than the low scenario. The assumptions used for the high scenario result in an estimate with a greater number of affected pollutants and a greater amount of incremental pollutant removals necessary to comply with CTR-based effluent limits compared to the low scenario. In addition, all necessary pollutant reductions were assumed to be achieved through either treatment or a waste control program (e.g., waste minimization, pollution prevention). That is, EPA did not employ the cost trigger for the high scenario.

Extrapolation of Costs

After estimating potential capital and O&M costs for each facility under the two scenarios, EPA estimated total annual costs by annualizing capital costs 7% over ten years and then adding in O&M costs. Note that this assumed ten year capital life likely overstates costs because capital equipment may last considerably longer than ten years.

EPA then extrapolated the annual costs based on the percent of the universe of regulated facilities represented by each group of sample facilities. EPA extrapolated major POTWs using three flow strata. EPA extrapolated major industrial facilities using industrial classification groupings (e.g., lumber and paper, electric utilities). Finally, EPA extrapolated minor POTWs and minor industrial facilities as separate groupings (not further distinguished by flow or industrial category).

4.1.3 Method for Estimating Pollutant Loading Reductions

EPA calculated pollutant loading reductions for each facility by calculating the difference between the baseline effluent concentration and the projected CTR-based effluent limitation.

For the low scenario the following apply:

- No reduction was assumed if the difference between the baseline value and the CTR limitation was negative.

- If the existing effluent concentration was above the MDL, but the CTR-based limit was below the MDL, the CTR-based limit, or one-half of the MDL (whichever produces a smaller load reduction) was used for the CTR-based effluent limitation.

- If the maximum reported effluent concentration exceeded the existing permit limit, high scenario assumptions were employed.

For the high scenario the following apply:

- If all effluent data for a pollutant were reported below detection levels, the method detection level (MDL) was used as the maximum observed concentration. If the maximum observed concentration was below the CTR-based limitation, no loading reductions were considered.

- If the difference between the baseline value (existing permit limit or effluent concentration) and the CTR limitation was negative, zero reduction was assumed.

- If both the CTR-based WQBEL and the existing permit limit were below the analytical MDL, one-half of the difference between the existing permit limit and the CTR-based limit was used to estimate the pollutant load reduction.

- If the existing permit limit (or effluent concentration in the absence of a permit limit) was above the MDL, but the CTR limit was below the MDL, the CTR-based limit, or one-half of the MDL (whichever produced a smaller load reduction) was used for the CTR-based limit for calculating pollutant load reductions.

EPA estimated annual baseline pollutant loadings by multiplying the baseline value (expressed in micrograms per liter) by the average daily flow rate (in million gallons per day), or, for publicly owned treatment works (POTWs), by the design flow, a conversion factor (0.00834), and 365 days per year. Then, to determine the reduction in loadings, EPA converted the difference between the most stringent existing permit limitation (or the maximum reported effluent concentration) and the most stringent CTR-based effluent limitation (in concentration units) to pounds per year by multiplying the difference by the facility's average daily flow rate (design flow rate for municipal dischargers), a conversion factor, and 365 days per year. EPA calculated annual pollutant loading reductions for each of the pollutants analyzed at each sample facility for which costs were estimated. The average load reduction then was calculated across sample facilities within each discharge category and extrapolated to the universe of facilities by multiplying the average load reduction by the total number of facilities in the category (EPA extrapolated facility specific costs similarly).

EPA converted baseline pollutant loadings and loading reductions from actual pounds (lb) to toxic-weighted pounds-equivalent (lb-eq) using EPA chronic freshwater aquatic life criteria and toxicity values, standardized to the former chronic aquatic life criterion for copper (copper formerly had a water quality criterion value of 5.6 ug/L). EPA human health criteria also were used in cases in which a human health criterion had been established for the consumption of fish. National water quality criteria have changed over the years, resulting in corresponding changes in toxic weights. Also, because the CTR applies to both freshwater and saltwater, two different sets of toxic weights would be required. To prevent the overstating of pollutant reductions due to the higher toxic weighting factors that would have been calculated using CTR criteria, EPA used previously calculated toxic weights shown in **Exhibit 4-1**. This approach allows for the direct comparison of the loadings reductions predicted from implementation of the CTR criteria and those loading reductions predicted in previous EPA rulemakings.

4.1.4 Method for Estimating Costs to Indirect Dischargers

Because of the uncertainty of the exact controls that POTWs would use as a result of more stringent CTR-based WQBELs, EPA assumed that many POTWs will select the option of controlling discharges to their collection system as a cost-effective means to comply with CTR-based permit limits. If POTWs were to select this method of control, the dischargers to the POTWs would be affected. Therefore, EPA estimated the potential costs to dischargers to POTWs (i.e., indirect dischargers).

EPA's estimate was based in part on information from the San Jose-Santa Clara and Sunnyvale POTWs, which discharge to South San Francisco Bay, and which already have conducted substantial work with indirect dischargers to meet current permit limits. Specifically, these POTWs were required to perform mass audit studies for copper and nickel. These studies estimated the total costs of implementing various combinations of copper and nickel reduction projects (see City of San Jose, 1994; EOA, 1994). Based on these studies, EPA estimated an average cost per significant industrial users (SIU) of $64,395, or $15,705 per year annualized at 7% over a period of 5 years. EPA then multiplied this cost by the percentage of SIUs assumed to be affected under the low and high scenarios (see Section 4.2).

Exhibit 4-1. Toxic Weights of Pollutants Analyzed

Pollutant	Toxic Weight
Antimony *	1
Arsenic	4
Cadmium	5.2
Chromium VI	35.5
Copper	0.47
Lead	1.8
Mercury	500
Nickel	0.036
Selenium	1.1
Silver	47
Thallium*	1
Zinc	0.051
1,2-Dichlorobenzene	0.011
1,2 Dichloroethane*	1
1,2 Dichloropropane*	1
1,2-Trans-Dichloroethylene*	1
1,3-Dichlorobenzene*	1
1,3-Dichloropropylene*	1
1,4-Dichlorobenzene*	1
2,4-Dinitrophenol*	1
2,4,6-Trichlorophenol	0.35
4,4'-DDD	760
4,4'-DDT	6,500
Aldrin	50
alpha-BHC	100
alpha-Endosulfan	100
Benzene	0.018
Benzo (a) Anthracene*	1
Benzo (a) Fluoranthene*	1
Benzo (k) Fluoranthene*	1
beta-BHC	100
beta-Endosulfan*	1
Bromoform*	1
Butylbenzyl-phthalate*	1
Carbon Tetrachloride*	1
Chlordane	2,300
Chlorobenzene*	1
Chlorodibromomethane*	1
Chloroform	0.0021
Dichlorobromomethane*	1
Dieldrin	57,000

Source: EPA/OST 1988 Cost-Effectiveness Criteria and Weights.
*Value was not provided in source document. A toxic weight of 1 was assumed.

4.2 SUMMARY OF CHANGES TO DRAFT ANALYSIS

To address comments raised on the draft economic analysis (SAIC, 1997), EPA gathered additional data and information to refine the analysis of potential costs and pollutant loading reductions attributable to the CTR. A large part of the effort was directed toward obtaining the most recent NPDES permits and effluent monitoring data for the sample facilities. Efforts also were directed toward increasing the sample size of minor POTWs and minor industrial facilities. EPA randomly selected four new minor POTWs and five new minor industrial facilities to add to the sample. The number of sample facilities selected in each RWQCB was roughly proportional to the universe of facilities in the region. The new sample facilities are listed in **Exhibit 4-2.**

Exhibit 4-2. New Minor Sample Facilities

POTWs	Industrial Facilities
Forestville County Sanitation District	Airline Signal Aerospace, Torrance Facility
City of Calistoga	Los Angeles County Department of Parks and Recreation, Lennox County Park
City of Biggs	Sierra Pacific Feather River Division, Mill Creek
Donner Summit Public Utility District Wastewater Treatment Plant	Great Lakes Chemical Corporation
	California Department of Fish and Game, Iron Gate Salmon Hatchery

An original minor industrial facility (Arco Station 434) was deleted from the sample, however, because it ceased operation and no effluent monitoring data were available for the analysis. As a result of the addition of the facilities listed in Exhibit 4-2, the final sample size of minor POTWs was five, and the final sample size of minor industrial facilities was six.

For the major facility sample, EPA randomly selected the San Diego Gas and Electric Facility at South Bay to replace the San Diego Gas and Electric Silvergate Facility. The Silvergate facility is no longer in operation and no effluent monitoring data were available for the analysis. Additionally, EPA removed the Mining Remedial Recovery Company (known as Alta Gold in the original sample) from the sample. The facility is a closed mine with acid mine drainage coming out of many adits disseminated around the property.

EPA also revised its methodology for calculating a PEQ to address mathematical problems encountered due to limited data sets for some facilities. In the original cost estimate, all non-detect values were assumed to be “zero” for the purposes of calculating the coefficient of variation (CV). This approach resulted in unrealistically high CV values in some instances. A high CV value results in the selection of a high PEQ multiplier and, consequently, a high PEQ value. Based on a review of the available data, EPA determined that using one-half of the method detection level, instead of “zero” for non-detects, resulted in a more accurate CV. EPA, therefore, revised the PEQ methodology to use the following:

- For calculation of the CV, half the detection level of the sample was used for effluent data reported as below the detection level.

- If greater than 20 data points were available for a pollutant, the 99th percentile value was calculated from the data set to represent the PEQ.

Changes to the cost estimation methodology included revisions to the treatment process optimization costs and waste minimization/pollution prevention cost estimates. EPA developed process optimization cost estimates to replace the previous average per facility cost of $100,000. Process modification costs are expressed as a range of values and are based on the flow and type of treatment system [e.g., $2,000 to $60,000 for optimization of biological treatment at a less than one million gallons per day (MGD) facility].

For waste minimization/pollution prevention, EPA increased the cost used in the high scenario for POTWs with a design flow of greater than 5 MGD (from $400,000 to $2,000,000). The revised cost is the highest pollution prevention cost estimate derived by EPA in assessing of compliance costs resulting from implementation of the proposed Great Lakes Water Quality Guidance (SAIC, 1993). The revised cost estimates are shown in **Exhibit 4-3**.

Exhibit 4-3. Revised Waste Minimization/Pollution Prevention Cost Estimates

Category	Low Scenario	High Scenario
POTWs with flow greater than 5 MGD	$400,000	$2,000,000
POTWs with flow less than 5 MGD	$400,000	$400,000
Minor POTWs	$50,000	$50,000
Minor Industrial Dischargers	$50,000	$50,000

Previously, when costs exceeded $500 per pounds-equivalent (lb-eq) removed for an industrial category in the high scenario, EPA assumed that regulatory alternatives to treatment would be used. For this revised analysis, the high scenario reflects the assumption that no regulatory alternatives will be available to dischargers. EPA still assumes that regulatory alternatives will be pursued under the low scenario when the cost for an individual facility exceeds $200 per lb-eq removed.

EPA also revised its assumptions regarding the number of indirect dischargers that may be affected as a result of more stringent CTR-based WQBELs at POTWs. EPA estimated that there are 2,144 SIUs that discharge to POTWs located on California inland surface waters and enclosed bays and estuaries. Previously, EPA assumed that 10% to 30% of SIUs would be impacted. Based on comments received indicating that the number of facilities affected was understated, EPA increased this range to 30% to 70%. However, EPA believes that this revised assumption is unrealistic and that the original 10% to 30% more closely reflects the likely impact. Nonetheless, EPA increased the range of affected facilities to ensure that the cost estimates for the final rule account are conservative (i.e., err on the side of higher costs).

EPA applied the 70% estimate under the low cost scenario which reflects less use of end-of-pipe treatment and more use of source controls. EPA applied the 30% estimate under the high scenario which reflects more frequent use of end-of-pipe treatment. EPA estimated costs to indirect dischargers based on the average costs from the mass audit studies conducted by San Jose and Sunnyvale, described above (see City of San Jose, 1994; EOA, 1994). Based on these studies, EPA estimated an average cost per indirect discharger of $64,395, or $15,705 per year annualized at 7% over a period of 5 years.

4.3 RESULTS

EPA estimated the potential annual cost of implementing the CTR ranges from approximately $33.5 million to $61.0 million. As shown in **Exhibit 4-4**, indirect dischargers bear most of these costs in the low scenario. Under the high scenario, direct dischargers were expected to incur most of the potential costs. However, high costs are unlikely because EPA used conservative (i.e., tending to err on the high side) assumptions in calculating CTR-based permit limits and in establishing baseline loadings. For example, the baseline loadings for the high scenario were generally based on current effluent limits rather than actual pollutant discharge data. Most facilities discharge pollutants in concentrations below current effluent limits.

Exhibit 4-4. Summary of Potential Annualized Costs
(Millions of 1998 First Quarter Dollars)

Discharger Category	Low Scenario	High Scenario
Direct Dischargers	$9.9	$50.9
Indirect Dischargers	$23.6	$10.1
Total	**$33.5**	**$61.0**

4.3.1 Low Scenario

Under the low scenario, major permitted dischargers account for the largest share of the costs (91%) compared to 9% for minor dischargers. Of the major dischargers, POTWs are expected to incur the largest share (87%) of the projected costs (see **Exhibit 4-5**). However, distributed among 128 major POTWs in the state, the average cost per plant is approximately $61,000 per year. Chemical and petroleum industries incur the highest cost of the industrial categories (5.6% of the total annual costs, with an annual average of $25,200 per plant). For minor dischargers, only POTWs are expected to incur costs (9%). The average cost per plant for minor POTWs is approximately $5,000, an amount lower than any major facility category.

Nearly 38% of the total compliance costs are for pursuing alternative regulatory approaches. EPA assumed that alternative regulatory approaches would be pursued if the total cost of treatment exceeded a trigger of $200 per pound of pollutant reduced. Annualized costs for developing and implementing waste minimization plans accounted for 57% of the remaining costs. Five pollutants (copper, mercury, tetrachloroethylene, carbon tetrachloride, and methylene chloride) account for 53% of low scenario annual costs. Fifty-five percent of annual costs were for the control of toxic organics; costs to control metals and mercury accounted for 45% of all annual costs.

Exhibit 4-5. Summary of Annual Costs by Discharger Category: Low Scenario (1998 First Quarter Dollars)

Discharger Category	Number of Plants	Total Costs	Category Cost as a Percent of Total Cost	Average Cost per Plant
Major Dischargers				
POTWs	128	$7,841,583	87.0%	$61,262
Chemicals/Petroleum Products	20	$504,016	5.6%	$25,201
Electric Utilities	13	$370,182	4.1%	$28,476
Metals and Transportation Equipment	7	$52,822	0.6%	$7,546
Miscellaneous	12	$240,171	2.7%	$20,014
Lumber and Paper	4	$0	0.0%	$0
Subtotal	**184**	**$9,008,774**	**100%**	**$48,961**
Minor Dischargers				
POTWs	185	$921,894	100%	$4,983
Industrials	872	$0	0.0%	$0
Subtotal	**1057**	**$921,894**	**100%**	**$872**
All Dischargers				
Total	**1241**	**$9,930,668**	**NA**	**$8,002**

4.3.2 High Scenario

Under the high scenario, major permitted dischargers account for 94% of the annual costs compared to 6% for minor sample facilities. For major dischargers, POTWs were expected to incur approximately 87% of the total projected annualized cost (see **Exhibit 4-6**). However, distributed among the 128 major POTWs in the state, the average cost per plant is approximately $325,000 per year. Chemical and petroleum industries incur the highest cost among the industrial categories (9% of the total estimated annual cost, and averaging just over $221,000 per plant annually). For minor facilities, the average cost per plant for POTWs is $7,800, compared to $1,600 per plant for industrial facilities.

Over 91% of the annual costs are for waste minimization and treatment process optimization costs. Waste minimization represents nearly 84% of the total annual costs. EPA assumed that waste minimization and process optimization would be pursued because the increment of pollutant removal is relatively small (i.e., less than 25% of current effluent levels) and many of the sample facilities already possess treatment processes that could be enhanced to achieve CTR-based effluent limits. Capital and operating and maintenance (O&M) costs make up less than 9% of total annual costs. Four pollutants (tetrachloroethylene, silver, copper, and mercury) resulted in 41% of the estimated high scenario annual costs. Costs to control metals and mercury accounted for more than 50% of annual costs; costs to control toxic organic pollutants accounted for slightly less than 50%.

Exhibit 4-6. Summary of Annual Costs by Discharger Category: High Scenario (1998 First Quarter Dollars)

Discharger Category	Number of Plants	Total Costs	Category Cost as a Percent of Total Cost	Average Cost per Plant
Major Dischargers				
POTWs	128	$41,599,147	86.5%	$324,993
Chemicals/Petroleum Products	20	$4,425,287	9.2%	$221,264
Electric Utilities	13	$370,182	0.8%	$28,476
Metals and Transportation Equipment	7	$351,815	0.7%	$50,259
Miscellaneous	12	$1,321,720	2.7%	$110,143
Lumber and Paper	4	$0	0.0%	$0
Subtotal	**184**	**$48,068,151**	**100%**	**$261,240**
Minor Dischargers				
POTWs	185	$1,448,691	51.1%	$7,831
Industrials	872	$1,386,377	48.9%	$1,590
Subtotal	**1057**	**$2,835,068**	**100%**	**$2,682**
All Dischargers				
Total	**1241**	**$50,903,219**	**NA**	**$41,018**

Note: Totals may not add up due to rounding.

4.4 POLLUTANT LOADING REDUCTIONS AND COST-EFFECTIVENESS

Exhibits 4-7 and **4-8** present the annual unweighted and toxic-weighted baseline pollutant loadings and loadings reductions, respectively. As shown in Exhibit 4-8, under the low scenario, where the baseline represents existing effluent concentrations, the expected reduction in pollutant loadings resulting from implementation of the criteria contained in the CTR is approximately 1.1 million toxic lb-eq per year, or 50% of the baseline load of 2.2 million toxic lb-eq per year. Under the high scenario, the expected reduction in pollutant loadings resulting from the implementation of the CTR-based WQBELs is approximately 2.7 million toxic lb-eq per year, or 15% of the baseline load of 18.5 million toxic lb-eq per year.

Exhibit 4-7. Baseline Pollutant Loadings and Reductions (Not Toxicity-Weighted, Lbs/yr)

Pollutant	Low			High		
	Existing	Reduction	% Reduction	Existing	Reduction	% Reduction
Antimony (Sb)	0	0		0	0	
Arsenic (As)	148	0	0.0%	319,804	0	0.0%
Cadmium (Cd)	82	0	0.0%	45,764	225	0.5%
Chromium VI (Cr-VI)	1,759	1,327	75.4%	102,848	14,167	13.8%
Copper (Cu)	386,839	44,675	11.5%	451,966	82,241	18.2%
Lead (Pb)	311,631	23,268	7.5%	454,881	36,765	8.1%
Mercury (Hg)	2,116	1,583	74.8%	2,477	1,627	65.7%
Nickel (Ni)	294,259	0	0.0%	1,509,107	277,120	18.4%
Selenium (Se)	152	0	0.2%	57,256	54	0.1%
Silver (Ag)	3,679	2,252	61.2%	103,751	16,479	15.9%
Thallium (Tl)	0	0		0	0	
Zinc (Zn)	468,416	0	0.0%	1,468,700	71,476	4.9%
1,2 Dichlorobenzene	0	0		14,319,172	421,101	2.9%
1,2 Dichloroethane	0	0		2	0	0.0%
1,2 Dichloropropane	0	0		2	0	0.0%
1,2-Trans-Dichloroethylene	0	0		0	0	
1,3 Dichlorobenzene	0	0		4,425,512	0	0.0%
1,3-Dichloropropylene	0	0		0	0	
1,4 Dichlorobenzene	0	0		163,541	0	0.0%
2,4-Dinitrophenol	0	0		0	0	
2,4,6 Trichlorophenol	879	756	86.0%	1,876	756	40.3%
4,4'-DDD	0	0		0	0	
4,4'-DDT	1	0	0.0%	1	0	0.0%
Aldrin	0	0	0.0%	0	0	0.0%
alpha-BHC	9	0	0.0%	27	0	0.0%
alpha-Endosulfan	0	0		5	0	0.0%
Benzene	0	0		41,957	0	0.0%
Benzo (a) Anthracene	1	0	0.0%	1	0	0.0%
Benzo (a) Fluoranthene	1	0	0.0%	1	0	0.0%
Benzo (k) Fluoranthene	17	0	0.0%	17	0	0.0%
beta-BHC	34	0	0.0%	96	0	0.0%
beta-Endosulfan	0	0		4	0	0.0%
Bromoform	0	0		924	0	0.0%
Butylbenzyl-phthalate	0	0		0	0	
Carbon Tetrachloride	5,586	3,502	62.7%	5,598	3,508	62.7%
Chlordane	0	0		0	0	0.0%
Chlorobenzene	0	0		0	0	
Chlorodibromomethane	12,532	10,296	82.2%	12,533	10,296	82.2%
Chloroform	250,555	223,085	89.0%	1,336,084	522,958	39.1%
Dichlorobromomethane	57,456	54,669	95.1%	57,786	54,679	94.6%
Dieldrin	0	0	0.0%	0	0	0.0%
Di-n-Butyl Phthalate	0	0		0	0	
Endosulfan Sulfate	0	0		0	0	
Endrin	0	0		6	0	0.0%
Endrin Aldehyde	0	0		0	0	
Fluoranthene	0	0		186,752	0	0.0%
Fluorene	0	0		0	0	
gamma-BHC	130	41	31.2%	213	41	19.1%
Heptachlor	0	0	0.0%	0	0	0.0%
Heptachlor epoxide	0	0	0.0%	0	0	0.0%
Hexachlorobenzene	0	0		291	280	96.1%
Methylene chloride	16,885	5,253	31.1%	1,453,701	5,253	0.4%
PCBs	0	0		0	0	51.9%
Pentachlorophenol	879	773	88.0%	11,583	5,510	47.6%
Phenol	0	0		767,755	0	0.0%
TCDD	0	0		0	0	0.0%
Tetrachloroethylene	6,832	4,487	65.7%	6,930	4,492	64.8%
Toluene	0	0		126,606,651	42,110,050	33.3%
Toxaphene	0	0		0	0	2.0%
Trichloroethylene	0	0		6	0	0.0%
Total Reductions	**1,820,879**	**375,967**	**20.6%**	**153,915,584**	**43,639,078**	**28.4%**

Exhibit 4-8. Toxicity-Weighted Baseline Pollutant Loadings and Reductions (Lbs-eq/yr)

Pollutant	Low			High		
	Existing	Reductions	% Reductions	Existing	Reductions	% Reduction
Antimony (Sb)	0	0		0	0	
Arsenic (As)	592	0	0.0%	1,279,218	0	0.0%
Cadmium (Cd)	424	0	0.0%	237,971	1,170	0.5%
Chromium VI (Cr-VI)	62,454	47,113	75.4%	3,651,115	502,936	13.8%
Copper (Cu)	181,814	20,997	11.5%	212,424	38,653	18.2%
Lead (Pb)	560,936	41,882	7.5%	818,785	66,178	8.1%
Mercury (Hg)	1,057,793	791,601	74.8%	1,238,464	813,467	65.7%
Nickel (Ni)	10,593	0	0.0%	54,328	9,976	18.4%
Selenium (Se)	167	0	0.2%	62,982	59	0.1%
Silver (Ag)	172,906	105,829	61.2%	4,876,297	774,534	15.9%
Thallium (Tl)	0	0		0	0	
Zinc (Zn)	23,889	0	0.0%	74,904	3,645	4.9%
1,2 Dichlorobenzene	0	0		157,511	4,632	2.9%
1,2 Dichloroethane	0	0		2	0	
1,2 Dichloropropane	0	0		2	0	
1,2-Trans-Dichloroethylene	0	0		0	0	
1,3 Dichlorobenzene	0	0		4,425,512	0	0.0%
1,3-Dichloropropylene	0	0		0	0	
1,4 Dichlorobenzene	0	0		163,541	0	0.0%
2,4-Dinitrophenol	0	0		0	0	
2,4,6 Trichlorophenol	307	264	86.0%	656	264	40.3%
4,4'-DDD	0	0		0	0	
4,4'-DDT	4,670	0	0.0%	9,720	0	0.0%
Aldrin	8	0	0.0%	17	0	0.0%
alpha-BHC	950	0	0.0%	2,686	0	0.0%
alpha-Endosulfan	0	0		500	0	0.0%
Benzene	0	0		755	0	0.0%
Benzo (a) Anthracene	1	0	0.0%	1	0	0.0%
Benzo (a) Fluoranthene	1	0	0.0%	1	0	0.0%
Benzo (k) Fluoranthene	17	0	0.0%	17	0	0.0%
beta-BHC	3,409	0	0.0%	9,553	0	0.0%
beta-Endosulfan	0	0		4	0	0.0%
Bromoform	0	0		924	0	0.0%
Butylbenzyl-phthalate	0	0		0	0	
Carbon Tetrachloride	5,586	3,502	62.7%	5,598	3,508	62.7%
Chlordane	0	0		465	0	0.0%
Chlorobenzene	0	0		0	0	
Chlorodibromomethane	12,532	10,296	82.2%	12,533	10,296	82.2%
Chloroform	526	468	89.0%	2,806	1,098	39.1%
Dichlorobromomethane	57,456	54,669	95.1%	57,786	54,679	94.6%
Dieldrin	9,717	0	0.0%	20,028	0	0.0%
Di-n-Butyl Phthalate	0	0		0	0	
Endosulfan Sulfate	0	0		0	0	
Endrin	0	0		566	0	0.0%
Endrin Aldehyde	0	0		0	0	
Fluoranthene	0	0		171,812	0	0.0%
Fluorene	0	0		0	0	
gamma-BHC	9,128	2,849	31.2%	14,925	2,849	19.1%
Heptachlor	799	0	0.0%	1,701	0	0.0%
Heptachlor epoxide	0	0	0.0%	0	0	0.0%
Hexachlorobenzene	0	0		209,741	201,623	96.1%
Methylene chloride	7	2	31.1%	610	2	0.4%
PCBs	0	0		3,001	1,557	51.9%
Pentachlorophenol	439	387	88.0%	5,792	2,755	47.6%
Phenol	0	0		21,497	0	0.0%
TCDD	0	0		16,065	0	0.0%
Tetrachloroethylene	506	332	65.7%	513	332	64.8%
Toluene	0	0		708,997	235,816	33.3%
Toxaphene	0	0		8,118	163	2.0%
Trichloroethylene	0	0		6	0	0.0%
Total Reductions	**2,177,628**	**1,080,192**	**49.6%**	**18,540,452**	**2,730,194**	**14.7%**

Exhibit 4-9 shows the 10 largest percentage toxic-weighted reductions by pollutant anticipated under the low and high scenarios. Under the low scenario, mercury is anticipated to be reduced by more than 73%; silver accounts for nearly another 10% reduction. Overall, organic removals account for just under 7% of the total reductions. The top two organics, dichlorobromomethane and chlorodibromomethane, are reduced by 5.1% and 1.0%, respectively. The small number of pollutants for which pollutant reductions were observed was partially because, under the low scenario, EPA assumed that alternative regulatory approaches would be sought for a number of pollutants and did not take credit for potential pollutant loading reductions.

Under the high scenario, just over 80% of the total projected toxic-weighted annual reductions will come from reducing metals, including mercury, while nearly 19% of expected reductions are for organic pollutants. Of the metals that will be reduced, mercury accounts for just under 30% of the total annual reductions and silver accounts for another 28%. Of the organics, toluene and hexachlorobenzene account for 8.6 and 7.4%, respectively, of the total annual reductions, while two other organic pollutants are reduced at relatively small percentages.

Exhibit 4-9. Ranking of Ten Highest Toxic-Weighted Pollutant Reductions

Low Scenario		High Scenario	
Pollutant	**Reduction as a Percent of Total**	**Pollutant**	**Reduction as a Percent of Total**
Mercury	73.3%	Mercury	29.8%
Silver	9.8%	Silver	28.4%
Dichlorobromomethane	5.1%	Chromium VI	18.4%
Chromium VI	4.4%	Toluene	8.6%
Lead	3.9%	Hexachlorobenzene	7.4%
Copper	1.9%	Lead	2.4%
Chlorodibromomethane	1.0%	Dichlorobromomethane	2.0%
Carbon tetrachloride	0.3%	Copper	1.4%
gamma-BHC	0.3%	Chlorodibromomethane	0.4%
Chloroform	<0.1%	Nickel	0.4%
Total	**100%**	**Total**	**100%**

Note: Totals are rounded.

The estimated cost-effectiveness of the rule is shown in **Exhibit 4-10** and ranges from $22 per toxic lb-eq to $31 per toxic lb-eq. In the low scenario, the highest cost-effectiveness value was observed for the electric utilities category ($43 per toxic lb-eq), while the lowest was for the chemicals and petroleum products category at $6 per toxic lb-eq. In the high scenario, the highest cost-effectiveness value was also for the metals/transportation equipment category at $223 per toxic lb-eq, while the lowest was for POTWs at $21 per toxic lb-eq. For comparison, **Exhibit 4-11** presents cost-effectiveness estimates from previous EPA rulemakings.

Exhibit 4-10. Annual Baseline Loads, Load Reductions, and Cost-Effectiveness

Category	Low Scenario			High Scenario		
	Annual Costs[1]	Loading Reductions[2]	Cost-Effective-ness[3]	Annual Costs[1]	Loading Reductions[2]	Cost-Effective-ness[3]
POTWs - Indirect Dischargers	$7.8 $23.6	0.91	$35	$41.6 $10.1	2.47	$21
Chemicals/Petroleum Products	$0.5	0.08	$6	$4.4	0.12	$36
Electric Utilities	$0.4	0.01	$43	$0.4	0.01	$43
Metals/Transport Equipment	$0.05	0.001	$38	$0.4	0.002	$223
Miscellaneous	$0.2	0.02	$10	$1.3	0.03	$38
Lumber/Paper	$0	0	NC	$0	0	NC
All Dischargers[4]	**$33.5**	**1.08**	**$31**	**$61.0**	**2.73**	**$22**

Note: Detail may not add to total due to rounding.
[1] Millions of 1998 first-quarter dollars.
[2] Millions of toxic-weighted pounds (lb-eq)
[3] $/lb-eq
[4] Including major and minor dischargers.
NC: Not calculated.

4.5 SOURCES OF UNCERTAINTY IN THE ANALYSIS

The estimates of potential compliance costs are based on assumptions to facilitate analysis and to overcome data limitations, where necessary. EPA generally designed these assumptions to be "conservative," that is, to err on the side of estimating more stringent and costly controls than would actually be required. Some of these assumptions also may tend to overstate pollutant loading reductions. **Exhibit 4-12** provides a summary of EPA's assumptions and the potential impact on the analysis of costs and benefits.

Exhibit 4-11. Estimated Incremental Cost-Effectiveness for Direct Dischargers by Industry[1] (1998 First Quarter Dollars)

Industry	Incremental Cost-Effectiveness for Selected Technology Options ($/lb-eq removed)[2]
Aluminum Forming	174.71
Battery Manufacturing	2.89
Coil Coating – Can making	14.44
Coal Mining	None
Coil Coating	70.75
Copper Forming	38.98
Electronics I	583.32
Electronics II	Not Available
Foundries	121.28
Inorganic Chemicals I	<1.45
Inorganic Chemicals II	8.67
Iron and Steel	2.89
Leather Tanning	None
Metal Finishing	17.32
Nonferrous Metals Forming	99.62
Nonferrous Metals Manufacturing I	5.77
Nonferrous Metals Manufacturing II	8.67
OCPSF	7.22[3]
Pharmaceuticals	1.45
Plastics Molding and Forming	None
Porcelain Enameling	8.67
Petroleum Refining	None
Pulp and Paper (PCB control for De-ink subcategory only)	25.99
Textile Mills	None

[1] Toxic and nonconventional pollutants only.
[2] Updated from 1981 dollars. Reflects incremental cost-effectiveness to proceed from current levels to levels represented by best available technology economically achievable.
[3] Reflects costs and removals of both air and water pollutants.
Source: EPA, 1992.

Exhibit 4-12. Biases and Uncertainties in the Analysis

Assumption	Potential Impact on Costs	Potential Impact on Benefits	Comments
Methods used to determine reasonable potential and calculate CTR-based WQBELs based on the EPA *Technical Support Document for Water Quality-based Toxics Control* (or TSD)	?	?	The TSD provides methods that account for sample size and effluent variability. If state implementation procedures are not comparable, TSD methods may over- or understate costs.
Use of 1:1 translator to convert dissolved-form criteria to total recoverable criteria for purposes of determining reasonable potential	+	+	Tends to result in more stringent effluent limits. Tends to overestimate the reasonable potential to exceed CTR-based limits.
Plant design flow used in calculating CTR-based effluent limits	+	+	Tends to overestimate the costs and pollutant loading reduction required to achieve CTR-based limits.
Zero dilution assumed in the absence of data or information related to critical low flow for the receiving water	+	+	Tends to make WQBELs more stringent. Tends to overestimate the costs and pollutant loading reductions required to achieve CTR-based limits.
Highest reported ambient receiving water concentration used to represent the background concentration when calculating CTR-based WQBELs	+	+	Using the highest reported value potentially denies the discharger use of a portion of the assimilative capacity of the receiving water. Tends to result in a greater need for treatment, and thus, potentially higher costs and pollutant loading reductions required to achieve CTR-based limits.
In the absence of ambient receiving water data, zero used as the background concentration	-	-	Assuming zero in the absence of background data potentially allows the discharger a larger portion of the assimilative capacity of the receiving water. Tends to underestimate costs and pollutant loading reductions required to achieve CTR-based limits.
Maximum pollutant effluent concentrations observed during the monitoring period used for estimating costs if CTR-based WQBELs were exceeded (low-end scenario)	+	+	Overstates the need for pollutant reductions to meet CTR-based WQBELs; tends to overestimate costs and pollutant loading reductions required to achieve CTR-based limits

Exhibit 4-12. Biases and Uncertainties in the Analysis (Continued)

Assumption	Potential Impact on Costs	Potential Impact on Benefits	Comments
Existing permit limit, or maximum pollutant effluent concentration in the absence of a permit limit, used for estimating costs if CTR-based WQBELs were exceeded (high scenario)	+	+	If facility is in compliance with effluent limits (i.e., discharging at levels below the permit limit), overstates the need for pollutant reductions to meet CTR-based WQBELs. Tends to overestimate costs and pollutant loading reductions required to achieve CTR-based limits.
Capital costs amortized over 10 years	+	0	The useful life of most equipment currently is more than 10 years. Tends to overestimate the annual costs to a facility.

\+ potentially upward bias.
\- potentially downward bias.
0 neutral bias.
? direction of bias unknown.

5.0 THE BENEFITS ASSOCIATED WITH THE CTR: METHODS AND CONCEPTS

The benefits analysis presented in this document provides insight into both the types and the magnitude of the benefits expected to arise as a result of implementing the CTR.[1] This chapter presents economics concepts and analytical issues associated with defining benefit categories and developing quantified and monetized benefits estimates. Section 5.1 describes the economic concepts used in the benefits analysis. Section 5.2 discusses the limitations of the analysis.

5.1 ECONOMIC CONCEPTS APPLICABLE TO THE BENEFITS ANALYSIS

This EA uses a conceptual foundation of "economic benefits" and assigns appropriate benefit categories to define and measure those benefits attributable to implementing the CTR. The sections below define terms used in that conceptual foundation and describe the concepts.

5.1.1 Economic Benefits

The term "economic benefits" refers to the dollar value associated with all the expected positive impacts of the CTR, that is, all CTR-related outcomes that lead to higher social welfare. The monetary value of benefits is the sum of the predicted changes in "consumer (and producer) surplus." These "surplus" measures are standard and widely accepted terms of applied welfare economics, and reflect the degree of well-being enjoyed by people given different levels of goods and prices (including those associated with environmental quality).

This conceptual foundation raises several relevant issues and potential limitations for the benefits analysis. First, the standard economic approach to estimating environmental benefits is anthropocentric—all benefit values arise from how environmental changes are perceived and valued by humans. This leads to the issue of how to define and measure "ecologic benefits" that may arise above and beyond the values humans place on environmental quality improvements (e.g., the protection and enhancement of habitat and living species). A related second point is that the benefits of all future outcomes are valued in present-day values. All future physical outcomes, near-term as well as long-term, associated with reduced pollutant loadings need to be predicted and then translated into the framework of present-day human activities and concerns.

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[1] Hereafter, references to the benefits resulting from the CTR, refer to the benefits that occur after implementation of the NPDES permits program to meet water quality standards established with CTR criteria. For this analysis, compliance with the CTR is expected to occur immediately. In reality, compliance, and thus costs and benefits, will occur as permits come up for review and are changed in accordance with revised water quality standards.

5.1.2 Benefit Categories Applicable to the CTR

To develop a benefits analysis, first the types or categories of benefits that apply must be defined. In this analysis, EPA relied on a set of benefits categories that applies to changes in the water resource environment. As reflected in **Exhibit 5-1**, benefits are categorized according to direct use of, or contact with, the resource.

Exhibit 5-1. Potential Benefits of Water Quality Improvements

Use Benefits	
In-Stream	Commercial fisheries, shell fisheries, and aquaculture; navigation Recreation (fishing, boating, swimming, etc.) Subsistence fishing Human health risk reductions
Near-Stream	Water-enhanced non-contact recreation (picnicking, photography, jogging, camping, etc.) Non consumptive use (e.g., wildlife observation)
Option Value	Premium for uncertain future demand Premium for uncertain future supply
Diversionary	Industry/commercial (process and cooling waters) Agriculture/irrigation Municipal drinking water (treatment cost savings and/or human health risk reductions)
Aesthetic	Residing, working, traveling, and/or owning property near water, etc.
Passive Use Benefits	
Bequest	Intergenerational equity
Existence	Stewardship/preservation Vicarious consumption
Ecologic	Reduced mortality/morbidity for aquatic and terrestrial wildlife Improved reproductive success for aquatic and terrestrial wildlife Increased diversity of aquatic and terrestrial wildlife Improved conditions for successful recovery of threatened and endangered species Improved integrity of aquatic and aquatic-dependent ecosystems

Use Benefits

Use benefit categories include in-stream, near-stream, and diversionary uses of the impacted waters and encompass both consumptive (fishing) and nonconsumptive activities (e.g., wildlife observation). In most applications to pollutant reduction scenarios, the most prominent use benefit categories are those related to recreational fishing, boating, and swimming.

Whether recreational use benefits reflect society's prime motivation for environmental protection measures is unclear. Many benefits analyses, however, focus on recreational values because they are well understood, there is a large body of empirical research to draw upon, and the associated benefits tend to be quite large. Recreational activities have received considerable empirical attention from economic researchers over the past two decades. The research relating to recreational fishing and similar activities generally indicates that water-based recreation is a highly valued activity in today's society.

Another use benefit category of potential significance for water quality regulations is human health risk reductions. Health risk reductions can be realized through actions that reduce human exposures to risk-posing contaminants, such as exposure through the consumption of fish or drinking water containing elevated levels of pollutants. Cost savings associated with the removal of contaminants from public drinking water supply systems is another form of a potential use benefit.

Passive Use (Nonuse) Benefits

Improved environmental quality can be valued by individuals apart from any past, present, or anticipated future use of the resource in question. Such passive or nonuse values have been categorized in several ways in the economics literature, typically embracing the concepts of existence, bequest, and stewardship. These nonuse values are associated with the purely public good aspects of environmental improvement in that the utility derived by an individual is entirely non-rival (an increase in utility derived by one individual does not reduce the welfare enjoyed by any other individual) and nonexcludable (there is no feasible way to exclude any individual from deriving utility from a nonuse aspect of an environmental improvement).[2]

Passive use values may be significant, but are difficult to quantify. Whereas human uses of a resource can be observed directly and valued with a range of technical economic techniques, passive use values can be ascertained only from asking survey respondents to reveal their values. The uncertainty in ascertaining passive use values has led to considerable debate as to whether they exist for applicable changes in environmental quality and, if so, whether they are of an appreciable magnitude relative to use values.[3] For the CTR, it is believed that passive use benefits are relevant and may be appreciable.

5.1.3 The Concept and Applicability of Ecologic Benefits

Among the relevant passive use values associated with the CTR are ecologic benefits associated with decreasing the level of toxic compounds found in California waters, sediment, and associated biota. Such ecologic benefits are likely to embody reduced risks of direct mortality, and increased reproductive success, in a range of important fish and wildlife species, as well as improved ecosystem health. The species include, but are not limited to, bald eagles, other

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[2] Many direct use benefits also arise from the public good context except, for example, to the extent that recreational benefits associated with improved water quality may be impeded by lack of access (private property holdings along the shoreline) or congestion. Nonuse benefits, on the other hand, are strictly of the nature of pure public goods, as neither access nor crowding are applicable to nonuse.

[3] For example, see Chapter 7 of the Regulatory Impact Analysis of the Proposed Great Lakes Water Quality Guidance, developed for U.S. EPA, April 15, 1993.

piscivorous avian species, mammalian species that feed on fish and crustaceans, and a wide range of aquatic species such as trout and other salmonids.

Some ecologic benefits clearly will have positive impacts that will manifest as use values (e.g., recreational angling, birdwatching). But of greater relevance is the applicability of ecologic benefits under the traditional passive use categories of existence and bequest values. One way to distinguish this, suggested by some analysts, is that passive use values remain anthropocentric, whereas ecologic benefits are held completely distinct from human valuation—making them additive to nonuse values. The question then becomes one of how to assign values to ecologic benefits for the purpose of setting priorities in policymaking.

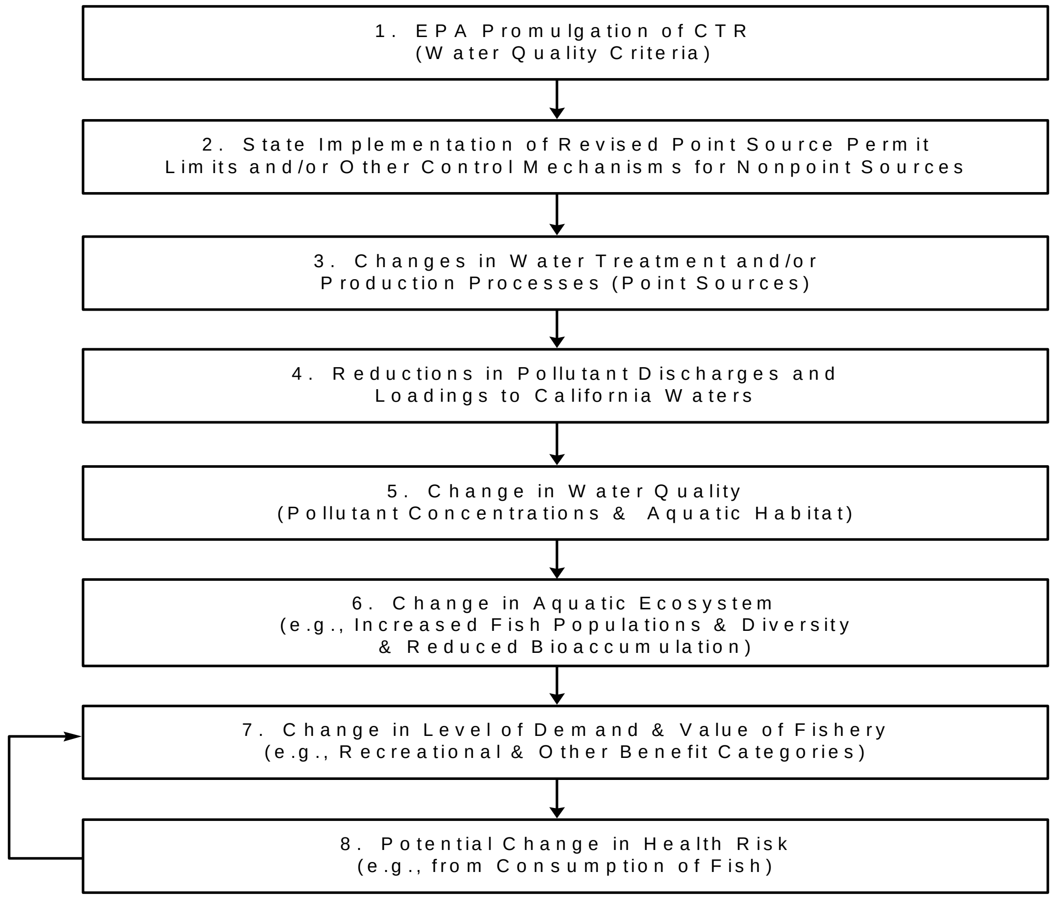
For the purposes of this EA, EPA addressed ecologic benefits in two manners. First, Chapter 6 provides a qualitative (and semi-quantitative) discussion of the physical relationships, mechanisms, and beneficial ecologic outcomes that may result from implementation of the proposed CTR. Second, for the purpose of the empirical efforts to monetize benefits, the CTR's ecologic benefits are considered to be included within passive use values and potential recreation benefits in which improved ecosystem health might be manifested.

5.2 LIMITATIONS INHERENT IN THE BENEFITS ANALYSIS

5.2.1 Causality: Linking the CTR to Beneficial Outcomes

In conducting a benefits analysis for anticipated CTR-related changes in pollutant loadings to California's waters, a chain of events must be specified and understood. As shown in **Exhibit 5-2**, this chain spans the spectrum of institutional relationships and policymaking; the technical feasibility of pollution abatement and facility-level decision-making regarding process and technology choices; the physical-chemical properties of receiving streams and their consequent linkages to biologic/ecologic responses in the aquatic environment; and human responses and values associated with these changes.

Exhibit 5-2. Chain of Events in CTR Benefits Analysis



The first two steps of Exhibit 5-2 reflect the institutional aspects of implementing the CTR, through which publication of the rule's water quality criteria is ultimately linked to state efforts to control pollutant loadings. In waters not meeting the water quality criteria, state regulators must assess how to allocate the necessary pollutant loadings reductions among various point and nonpoint sources. To the extent that these loadings reductions are assigned to point source dischargers, the state's actions will be manifested in revised point source discharge permits. The costing analysis for the CTR presumes that all loadings reductions will be generated through point source controls; however, it is possible that state regulators will implement the rule such that nonpoint source control efforts may be used in addition to some portion of the point source controls assumed here.

In steps 3 and 4, the revised state permit limits ultimately result in a change in pollutant loadings for targeted contaminants (as well as those removed incidental to the improved wastewater treatment or process changes), from an appropriately defined set of baseline loadings. The actual manner in which the loadings reductions are achieved will depend on treatment technology and process changes selected by individual facilities. These technology choices will determine the compliance costs and loadings reductions.

Next, as shown in steps 5 and 6 of Exhibit 5-2, pollutant loading reductions (from step 4) need to be converted into changes in environmental conditions such as physical/chemical parameters (in-stream pollutant concentrations) and the consequent improvement in biota (e.g., increased diversity and size of fishery populations). In lieu of detailed water quality and ecologic (e.g., fisheries) modeling, which was infeasible within the time frame and budget limits of this analysis, this benefits analysis relies on a more ad hoc characterization of the specific pollutants addressed and their links to restricted beneficial uses of the resource. These are described, in part, in Chapter 6.

Finally, in steps 7 and 8, the analysis reaches the stage at which anthropocentric benefit concepts begin to apply, such as illustrated by the link between improved fisheries and the enhanced enjoyment realized by recreational anglers. These final steps reflect the focal point of the quantitative benefits analysis presented in Chapter 8, and are defined by the benefits categories described above. But as noted below, there are several issues that inhibit the ability to accurately forecast the extent to which the CTR may generate such benefits.

5.2.2 Temporal and Spatial Issues

As noted above, it is important to recognize the analytic challenges and resulting limitations associated with estimating the benefits of reducing discharges of toxic pollutants to all California waters. An empirical benefits assessment is a difficult and uncertain undertaking under the best of circumstances. In the case of the CTR, the challenges and limitations are magnified by several important considerations, including, but not limited to the following:

- *The time path to ecosystem recovery from near-term reductions in toxic loadings.* Many of the toxic compounds relevant to the CTR are persistent in the environment; therefore, even the total elimination of additional loadings of these compounds may not immediately alter water column or fish-tissue concentrations. A significant portion of the benefits may be realized only in the relatively distant future.

- *The geographic scope of contamination and of benefit-generating activities throughout the varied watershed ecosystems of California.* Typically, the areal extent of toxic contamination is very widespread, even if it originates from a well-defined source at a specific location. Contamination becomes even further dispersed through uptake in the food chain. Thus, the benefits of reducing toxic discharges within the state's watersheds are likely to extend beyond the boundaries of the state's "impaired" waters.

The time-path issue can be addressed, in part, through the use of alternative discounting regimes in the benefits analysis. The geographic scope issue is more difficult to address empirically, other than to recognize the high probability that beneficial results of the CTR will be realized beyond the boundaries of impaired state waters.

5.2.3 Attribution of Benefits to the CTR

For this analysis, EPA had data and information to estimate large-scale changes in water quality beyond present day conditions and then attributed the CTR for its contribution to these changes. First, the current total pollutant loadings from all sources that are contributing to the toxics-related water quality problems observed in the state are assessed. This defines the overall magnitude of the loadings "problem." Second, the share of the total loadings problem that is attributable to sources that are likely to be controlled via the CTR are estimated. Since this analysis was designed to focus only on those controls imposed on point sources, this stage of the process entailed examining the portion of total loadings originating from point sources (see Chapter 7). Third, the percent reduction in point source loadings expected due to implementation of the CTR is estimated, then applied to the share of point source loadings.

For example, if the total benefits of moving from baseline water quality to having all of California's waters completely unimpaired were estimated to be $500 million per year, and point sources contributed 40% of the toxic-weighted pollutant loadings that contributed to baseline impairments, then one would estimate (absent more refined data) that perhaps $200 million of the potential water quality benefits would be attributable to the potential elimination (100% reduction) of all point source discharges. If the CTR was expected to achieve a 50% reduction in the offending point source discharges, one would then develop an estimate of $100 million as a rough approximation of CTR-related benefits.[4] Thus, total baseline pollutant loads, and anticipated loadings reductions, are used as a means to approximate roughly the share of total potential water quality benefits that may be attributed to the rule. In the example above, the CTR would be viewed as addressing 20% of the total loadings problem (reducing by 50% the 40% of total loadings due to point sources).

Yet one of the difficulties in applying the loadings-based attribution approach is obtaining and interpreting data on baseline loadings. The problem entails two significant challenges:

! *Developing reliable estimates of both ongoing point source loadings and current nonpoint source loadings.* This is difficult because nonpoint loadings come from a wide variety of sources that are difficult to measure, including atmospheric deposition and agricultural and urban runoff. Thus, nonpoint source loading estimates are probably highly imprecise and very incomplete because they likely omit sources underestimating load estimates. Even point source estimates of

[4] Forty percent of $500 million equals $200 million; 50% of this $200 million equals $100 million.

loadings are imprecise because discharged concentrations may be below detection limits (i.e., “hidden loads” may exist in discharge data).

- *Accounting for the share of the current loadings versus those attributable to historical discharges from point and nonpoint sources.* Many of the pollutants addressed by the CTR are persistent (e.g., metals) and bioaccumulative (e.g., dioxins, PCBs, and selected agricultural chemicals). Their presence in the water column, sediment, and biota of California waters may be largely due to historical discharges rather than current loadings. The degree to which historical loads contribute to present-day concentrations will vary according to many complex contaminant- and site-specific factors. However, historical loads may, in some instances, be the predominant source of toxics-related water quality problems. In such instances, efforts to control current discharges may be of relatively limited effectiveness and value.

These complicating factors are difficult to account for in the attribution analysis. Nonetheless, they need to be kept in mind when interpreting the loadings data that are available for an apportionment analysis. These issues are described in greater detail in Chapter 7.

6.0 QUALITATIVE ASSESSMENT OF POTENTIAL ECOLOGICAL BENEFITS

This chapter describes the types of ecological benefits anticipated to result from implementation of the CTR. Improvements in ambient water quality, anticipated under the rule, are expected to result in substantial ecologic benefits through improvements in ecosystem health. This chapter provides an overview of the adverse effects of toxics on California's diverse ecological systems, shows how improved ambient water quality can translate into improved ecosystem health, and qualitatively assesses the ecologic benefits anticipated under the proposed rule.

Section 6.1 gives an overview of the diversity of ecological systems in California. Section 6.2 summarizes the occurrence and ecological effects of toxics in California aquatic systems. Section 6.3 describes how CTR-related toxics reductions may result in improved ecosystem health through ecological and toxicological interactions. Section 6.4 provides a qualitative discussion of potential ecologic benefits of the proposed rule.

6.1 ECOLOGICAL DIVERSITY OF AQUATIC ENVIRONMENTS IN CALIFORNIA

California is one of the most biologically diverse areas in the world (U.S. EPA, 1997). Within its 160,000 square miles of land, and hundreds of thousands of acres and miles of estuaries, wetlands, rivers, streams, and lakes, California harbors more unique plants and animals than any other state in the nation. The diverse climates, landscapes, habitats, and migration barriers such as mountains and deserts, have led to the evolution of a large number of isolated species and varieties of animals, many of which are found only in California (Steinhert, 1994, as cited in U.S. EPA, 1997). For example, there are 46 species of amphibians, 96 species of reptiles, 563 species of birds, 190 species of mammals, 8,000 species of plants, and 30,000 species of insects recorded in the state. In addition, 63 types of freshwater fish are found only in California (Moyle, 1994, as cited in U.S. EPA, 1997). Additionally, California's aquatic systems provide important habitat for migratory species such as waterfowl.

Unfortunately, California's ecological diversity is threatened (U.S. EPA, 1997). On average, more than 20 percent of the naturally occurring species of amphibians, reptiles, birds, and mammals are classified as endangered, threatened, or of "special concern" by state and federal agencies. California has more threatened and endangered species than any other state in the United States. Many of these species exist in or are dependent on aquatic resources during all or part of their lives, and consequently may be adversely affected by toxic discharges to surface waters (U.S. EPA, 1997).

6.2 OCCURRENCE AND ECOLOGICAL EFFECTS OF TOXICS IN CALIFORNIA AQUATIC SYSTEMS

Current concentrations of toxics in California's aquatic systems may pose substantial risk to resident and migratory biota through direct and indirect pathways of exposure in the surface waters, diets, or sediments. It appears that a variety of toxics are widely distributed throughout California, which increases the likelihood that many of the resources are exposed to concentrations potentially causing adverse effects on ecological resources (U.S. EPA, 1997). Toxicity may occur with either acute (short-term) or chronic (long-term, sublethal) exposure to contaminants. Exposure to chronic, low levels of toxics found in California's aquatic environments can adversely affect the resources by causing physiological and behavioral impairments in organisms, contamination or reduction of food-web resources, and alteration of habitats. Improving ambient water quality would put the ecological and biological resources at less risk of exposure. Improved water quality through toxics reductions would also reduce the risk of disturbances to the ecological integrity and important habitats of the biological resources of California.

A key to understanding the potential benefits of the proposed rule on the ecological resources of California is a knowledge of the occurrence, exposure pathways, and effects of toxics occurring in California's aquatic systems. These factors are discussed below.

6.2.1 Occurrence of Toxics-Related Impairments

As shown in **Exhibit 6-1**, California's aquatic ecosystems in all areas of the state exhibit impaired water quality from toxics such as metals, selenium, pesticides, and priority organics such as PCBs (U.S. EPA, 1997).[1] The Analysis of the Potential Benefits Related to Implementation of the California Toxics Rule (U.S. EPA, 1997) summarizes ambient water quality impairment in California and notes the following:

- Available data suggest that over 800,000 acres of assessed bays, estuaries, lakes, and wetlands may be impaired by one or more toxic pollutants, as are over 3,700 miles of rivers. Most notably, over two-thirds of the assessed area of both bays and saline lakes may be adversely affected by toxics.

- Inorganic pollutants such as metals and trace elements (particularly selenium) are the most significant categories of toxic pollutants affecting the water quality in assessed waters statewide. Pesticides are also associated with large areas of water quality impairment.

[1] Impaired waters are defined as those that have been rated by the State of California as medium or poor for at least one toxic pollutant or group of pollutants. California's medium and poor waters correspond to U.S. EPA's categories of not fully or partially supporting designated uses. The medium and less severely impaired waters were grouped together into the partially supporting category. The remaining waters classified as poor were placed in the not fully supporting category.

! Trace elements (especially selenium) may be responsible for water quality impairment in 52% of all bays, 55% of rivers and streams assessed, and 16% of all lakes and reservoirs. In addition, trace elements may impair water quality in all saline lakes in the state.

! Based on the areal extent of contamination and the uses of affected water bodies, San Francisco Bay and the Central Valley appear to be the areas most influenced by toxic contamination. In addition, toxics are responsible for impaired water quality in a high percentage of river and saline lake areas in the Colorado River Basin. These areas constitute those most extensively affected by toxics, but waters in all regions of California show some degree of impairment by toxics.

! Both point and nonpoint sources play a role in contributing to toxic pollution. Agriculture, primarily agricultural drainage, is the most frequently cited source of pollutants that impair rivers and is also frequently cited as a contributor to the impairment of lakes and reservoirs. Urban runoff and "other" nonpoint sources (e.g., deposition and spills) are most frequently cited as contributing factors to water quality problems in toxics-impaired bays. Mining is the most frequently cited source (mining operations may or may not be a point source), particularly for lakes and reservoirs, and toxics discharged by municipal wastewater treatment plants contribute to the impairment of a variety of water body types, particularly estuaries and wetlands.

! Toxic pollutants are of concern in a large number of waters designated for the support of terrestrial and aquatic wildlife. In addition, water quality in 175,000 acres of bays/harbors, 52,000 acres of estuaries, 102,000 acres of lakes, 1,000 miles of rivers and streams, and 11,000 acres of saline lakes that support fish spawning and/or migration may be impaired by toxics.

! Toxics may contribute to impaired water quality in approximately 176,000 acres of bays or harbors, 1,856 river miles, 230,000 acres of saline lakes, and 5,000 acres of estuaries designated for the support of rare, threatened, or endangered species.

! Currently, there are 12 fish consumption health advisories in waters covered by the CTR (9 inland water bodies and 3 enclosed bays and estuaries) because of high levels of contamination in fish tissue from mercury, PCBs, chlordane, dioxin, DDT, pesticides, and selenium. Some of these tissue contaminants are also hazardous to fish and piscivorous (fish-eating) species as well.

! Currently, there are four waterfowl health warnings for consuming waterfowl taken from the Grasslands area, Suisun Bay, San Pablo Bay, and San Francisco Bay because of elevated selenium levels in waterfowl such as duck, greater and lesser scaup, and scoters. Selenium contaminant levels are also a concern for waterfowl health.

Exhibit 6-1. Summary of Baseline California Regional Water Quality Assessments[1]

Region	Areal Extent of Toxics Impairment	Pollutants of Concern	Primary Pollutant Sources	Key Water Bodies Impaired	Ecological Resources Potentially Affected
Region 1: North Coast Region	55% of bays (16,500 acres); minor impairment of other water bodies	Metals, pesticides	Mix of point sources (municipal and industrial effluent) and nonpoint sources (agriculture and urban runoff)	Arcata Bay, Humboldt Bay	Wildlife habitat; fish spawning and migration; rare and endangered species
Region 2: San Francisco Bay	Large areas impaired by toxics, including 70% of bays (200,000 acres); 60% of wetlands (57,000 acres); 39% of rivers (244 miles); 172,000 acres impaired supporting fish spawning/migration and rare and endangered species	Metals, trace elements, priority organics	Urban runoff and other nonpoint sources affect largest areas; some impairment from municipal and industrial point sources	San Francisco Bay (Lower, Central, South) Suisun Marsh	Wildlife habitat; fish spawning and migration; rare and endangered species; waterfowl; piscivorous wildlife in San Francisco Bay, Lake Herman, Guadalupe Reservoir; and other species
Region 3: Central Coast Region	47% of lakes (11,700 acres); 36% of estuaries (1,700 acres); minor impairment of rivers and bays	Metals, pesticides	Agriculture, mining, unspecified nonpoint sources	Morro Bay, Carpinteria Marsh, Elkhorn Slough	Wildlife habitat; fish migration and spawning; rare and endangered species; piscivorous wildlife in Nacimiento River
Region 4: Los Angeles Basin	Over 90% of bays and estuaries impaired (16,000 acres); minor impairment of rivers and lakes	Pesticides, priority organics, trace elements, metals	Mix of point sources (municipal treatment, “other” point sources) and nonpoint sources (agriculture, hydrological modification, and urban runoff)	Mugu Lagoon, San Gabriel River (lower), Los Angeles River (upper)	Wildlife habitat; fish migration and spawning; rare and endangered species; piscivorous wildlife in Lake Nacimiento and Los Angeles Harbor

Exhibit 6-1. Summary of Baseline California Regional Water Quality Assessments[1] (Continued)

Region	Areal Extent of Toxics Impairment	Pollutants of Concern	Primary Pollutant Sources	Key Water Bodies Impaired	Ecological Resources Potentially Affected
Region 5: Central Valley Region	Large areas impaired by toxics, including 100% of estuaries (48,000 acres); 23% of lakes (120,000 acres); 21% of rivers (1,200 miles); 48,000 acres of Delta waterways impaired for fish spawning/migration and rare and endangered species	Metals, trace elements	Agriculture, mining; smaller areas affected by municipal treatment, urban runoff, storm sewers, and other nonpoint sources	Delta Waterways, Clear Lake, American River, Feather River, Sacramento River, Grasslands, Marshes, Shasta Lake	Wildlife habitat; fish spawning and migration; rare and endangered species, piscivorous wildlife in Clear Lake, Lake Berryessa, and Grasslands area; waterfowl in Grasslands area
Region 6: Lahontan Region	34% of saline lakes (66,000 acres); 19% of lakes (36,000 acres); 13% of rivers (372 miles)	Metals, trace elements, priority organics	Naturally occurring levels of metals and trace elements; lesser areas affected by agriculture, land development, and mining	Eagle Lake, Owens River, Truckee River, Honey Lake	Wildlife habitat; fish spawning and migration; rare and endangered species
Region 7: Colorado River Basin	60% of rivers (1,400 miles) impaired; 220,000 acres of saline lake (Salton Sea) supporting rare and endangered species and wildlife	Pesticides, trace elements	Agriculture	Salton Sea	Wildlife habitat; rare and endangered species; piscivorous wildlife and waterfowl in Salton Sea
Region 8: Santa Ana River Basin	Over 90% of bays and estuaries impaired (4,000 acres); 27% of lakes (4,000 acres)	Metals, pesticides	Primarily nonpoint sources including agriculture, urban runoff, and land development	Upper Newport Bay	Wildlife habitat; fish spawning and migration; rare and endangered species
Region 9: San Diego Basin	14% of estuaries; minor impairment of other water bodies; 239 acres San Diego Bay impaired supporting fish spawning/ migration and rare and endangered species	Metals, pesticides, priority organics, trace elements	Estuaries affected by land disposal; other water bodies affected by diverse mix of point and nonpoint sources.	San Diego Bay, Tijuana River Estuary	Wildlife habitat; fish spawning and migration; rare and endangered species

[1] Based on 1994 assessment of water quality. Some key water bodies impaired by toxics have changed since that time; however, more recent data were not used in the preparation of this report due to time constraints.
Source: U.S. EPA (1997).

This summary of water quality impairment indeed reveals that a variety of aquatic and terrestrial biota are exposed to the toxics regulated by the CTR.

6.2.2 Exposure Pathways

Toxics present in California's aquatic systems can affect ecological resources through direct or indirect pathways of exposure. Direct pathways of exposure occur when natural resources come in direct contact, either singularly or in combination, with toxics in the water column, sediments, or diet. Indirect pathways of exposure occur when habitat resources (e.g., spawning beds, prey sources) have been reduced or otherwise altered by toxics. Toxics also may be bioaccumulated in aquatic organisms, making them available to terrestrial predators dependent on the aquatic food web of the contaminated system. The extent to which the organisms are adversely affected largely depends on the pathway and duration of exposure as well as the concentration and type of toxics present in the pathway.

6.2.3 Potential Effects of Toxics on Ecological Resources

Ecological resources potentially affected under state implementation include biota and ecosystem function and integrity.

Effects on Biota

Biological organisms are effective receptors for toxics in aquatic systems through the uptake, accumulation, and eventual biological disposition of contaminants. Uptake of toxics results from the following various exposure pathways, singularly or in combination: diet, water, and sediment. Accumulated toxics associated with ambient waters may concentrate in various tissues and organs of biota. The specific tissues/organs affected depend on the exposure pathways, the exposure concentrations, and the ability to metabolize or excrete the accumulated contaminants. An organism's ability to metabolize contaminants largely depends on the presence/absence and relative abundance of various enzymes necessary to transform different components into excretable compounds.

The effects of toxics on aquatic resources must be evaluated because even low contaminant concentrations in water, sediment, or diet may impair fitness, produce adverse physiological effects that lead to death, or lower long-term survivability in the wild. There is extensive documentation of the long-term, injurious effects of inorganic (e.g., heavy metals) and organic (e.g., polychlorinated biphenyls, pesticides, aromatic hydrocarbons) contaminants at relatively low concentrations to aquatic biota (Rand and Petrocelli, 1985; Hoffman et al., 1995).

Exposure to contaminants found in California's aquatic systems can affect various biological levels of organization, resulting in four identified biotic responses: lethal toxicity, sublethal toxicity, bioaccumulation, and habitat alteration. These biotic responses provide broad categorization for a multitude of specific biotic responses **(see Exhibit 6-2)**.

Exhibit 6-2. Biological Organization Levels Associated with Responses to Toxics in Water

Biotic Response	Subcellular	Cellular	Organism	Population	Community	Ecosystem
Lethal Toxicity	✓	✓	✓	✓	✓	✓
Sublethal Toxicity	✓	✓	✓	✓	✓	✓
Bioaccumulation	✓	✓	✓	✓	✓	✓
Habitat Alteration			✓	✓	✓	✓

Lethal toxicity refers to the direct disruption of subcellular or cellular physiological activities that result in death of the organism. The death of individuals from populations can influence the future reproductive viability of populations, and in turn may influence organisms at higher trophic levels. Sublethal toxicity also involves interference of subcellular and cellular processes, but does not result in immediate death; death may occur because of impaired behavior, or impaired physiological or biochemical processes.

Bioaccumulation of contaminants found in California aquatic systems is important because the health of organisms may be affected (e.g., reducing growth or reproduction; increasing susceptibility to disease). Bioaccumulation also results in additional pathways for contaminant transfer throughout the food chain. Impaired physiology or contaminant transfer through food chains owing to bioaccumulation can have dramatic impacts on all levels of biological organization. For instance, accumulated contaminants (or metabolites of these contaminants) transferred through food webs may concentrate in food sources of piscivorous fish, which can adversely affect important recreational or commercial fisheries.

Habitat alteration includes effects on the physical and chemical environment that can result in unsuitable habitat for both resident and migratory biota at the level of the organism and the population. For example, biodegradation of organic contaminants by sediment microbes results in anoxic conditions unsuitable for benthos. The physical and chemical alteration of particular habitats can shift species composition, abundance, and diversity. Any change in species composition directly reflects altered community structure, and can alter ecosystem functions.

Toxics of particular concern in California are listed in **Exhibit 6-3**, along with their potential adverse effects on biota. In addition to the potential adverse effects of toxics discussed above, exposure to certain toxics present in California's aquatic systems, including aromatic hydrocarbons and heavy metals, can increase the rate of genetic mutations. Increased rates of genetic mutations can reduce the fitness of individuals and populations, especially in contaminated areas providing breeding or spawning habitat because there would be greater risk to embryonic life stages undergoing rapid development.

Exhibit 6-3. Overview of Adverse Effects of Toxics

Toxic of Concern	Potential Affected Ecological Resource in California	Potential Adverse Effects on Biota[1]	Reference
Arsenic	Aquatic biota Birds Mammals Water Sediment	Reduced growth and survival Impaired reproduction Impaired physiology Decreased resistance to infection Mutagenic Teratogenic Carcinogenic	Eisler (1988a)
Cadmium	Aquatic biota Birds Mammals Water Sediment	Reduced growth and survival Impaired reproduction Possible mutagen Teratogenic Carcinogenic	Eisler (1985a)
Chromium (III and VI)	Aquatic biota Birds Mammals Water Sediment	Reduced growth and survival Impaired reproduction Mutagenic Teratogenic Carcinogenic	Eisler (1986a)
Copper	Aquatic biota Birds Mammals Water Sediment	Reduced growth and survival Impaired reproduction Impaired metabolism	U.S. EPA (1985); Goyer (1991)
Dioxin	Aquatic biota Birds Mammals Water Sediment	Reduced growth and survival Impaired reproduction Compromised immunity Mutagenic Teratogenic Carcinogenic	Eisler (1986b)
Endosulfan	Aquatic biota Birds Mammals Water Sediment	Reduced growth and survival Impaired reproduction Impaired behavior Suspected mutagen	Verschueren (1983); Smith (1991)
Lead	Aquatic biota Birds Mammals Water Sediment	Reduced growth and survival Impaired reproduction Impaired development Impaired metabolism	Eisler (1988b)

Exhibit 6-3. Overview of Adverse Effects of Toxics (Continued)

Toxic of Concern	Potential Affected Ecological Resource in California	Potential Adverse Effects on Biota[1]	Reference
Mercury	Aquatic biota Birds Mammals Water Sediment	Reduced growth and survival Reduced reproduction Impaired development Impaired behavior Mutagenic Teratogenic Carcinogenic	Eisler (1987a)
Nickel	Aquatic biota Birds Mammals Water Sediment	Reduced growth and survival Reduced reproduction Carcinogenic	U.S. EPA (1980a)
Polycyclic Aromatic Hydrocarbons (PAHs)	Aquatic biota Birds Mammals Water Sediment	Reduced growth and survival Reduced reproduction Compromised immunity Mutagenic (4-7 Ringed PAHs) Teratogenic (4-7 Ringed PAHs) Carcinogenic (4-7 Ringed PAHs)	Eisler (1987b)
Polychlorinated Biphenyls (PCBs)	Aquatic biota Birds Mammals Water Sediment	Reduced growth and survival Reduced reproduction Impaired behavior Compromised immunity Mutagenic Teratogenic Carcinogenic	Eisler (1986c)
Selenium	Aquatic biota Birds Mammals Water Sediment	Reduced growth and survival Reduced reproduction Impaired behavior Impaired physiology	Eisler (1985b)
Silver	Aquatic biota Birds Mammals Water Sediment	Reduced growth and survival Reduced reproduction Impaired physiology	Goyer (1991); U.S. EPA (1980b)
Zinc	Aquatic biota Birds Mammals Water Sediment	Reduced growth and survival Impaired physiology Teratogenic to amphibians Reduced reproduction	Eisler (1993)

[1] The potential for adverse effects to ecological resources are dependent on numerous factors, including the exposure route, the exposure duration, the dose, the sensitivity of the organism, and the bioavailability of the chemical. Information in this table describes the common biological effects associated with each toxic of concern (U.S. EPA, 1997). Effects may not be present for all ecological resources listed. In addition, concentrations of these toxic compounds in California may not be high enough to result in these adverse effects on biota.

Effects on Ecosystems

In addition to adverse effects on biota, toxics also may adversely affect ecosystem function and integrity through direct and indirect effects on biota. The effects of toxics in ecosystems are complex and difficult to estimate because of the diversity of species assemblages and trophic interactions (Barron and Woodburn, 1995). For example, predators may switch to alternate prey (Eaton et al., 1985), or phytoplankton abundance may be maintained by changes in the dominant algae species (Brock et al., 1992). Also, several species may perform similar functions, with sensitive species replaced by more contaminant tolerant species. While these changes may result in overall ecosystem resistance to toxics, there could be major changes in ecosystem structure (Barron and Woodburn, 1995). Contaminant effects on ecosystem structure, however, are likely to be specific to the type and location of the water body and the toxic exposure scenario.

6.3 POTENTIAL ECOLOGIC BENEFITS OF THE RULE

As discussed in Chapter 5, ecosystems and their biological resources provide benefits through enhanced ecological services that often manifest as direct use values, passive use values and, to the extent not reflected elsewhere, ecologic benefits.

This section provides a qualitative description of potential ecologic benefits resulting from improvements in ecosystem health under the proposed rule. A qualitative description of ecologic benefits is provided because of the complexity and diversity of California aquatic systems and the diversity of ecological receptors; the multitude of contaminants and exposure conditions; the complexity of ecosystem structure and function; and uncertainty regarding the extent to which the CTR will result in toxics loading reductions significant enough to generate appreciable changes in ambient concentration and ecosystem health. However, improved water quality may provide potential benefits to the ecological resources ". . . that exist in or are dependent on more than 800,000 acres of assessed bays, estuaries, lakes, and wetlands and more than 3,700 miles of rivers that are now currently impaired by toxic pollutants" (U.S. EPA, 1997). The extent, magnitude, and nature of the ecologic benefits accruing under the CTR will depend on the specific ecosystems and toxics affected, baseline conditions, the degree and type of ambient water quality improvements, and the time horizon for improvement.

Toxics reductions under the CTR may provide ecologic benefits through increased ecosystem stability, resilience, and overall health (U.S. EPA, 1997). Benefits are difficult to quantify because of the complexity, scale, and uncertainties of the interaction of the multitude of ecological systems and toxics to be affected by the proposed rule. However, ecologic benefits from the proposed rule may be substantial because of the extensive variety, proportion, and geographic area of the affected aquatic systems, the diversity and uniqueness of California ecological resources, and the large number of toxics to be regulated under the CTR (U.S. EPA, 1997).

Without conducting a complete analysis as described above, EPA concludes that potential ecologic benefits from implementation of the CTR may include (U.S. EPA, 1997):

- Reductions in toxics loadings that lead to improved conditions for California fish spawning and/or migration in bays/harbors and estuaries, lakes, rivers, streams, and saline lakes

- Reductions in bioaccumulative chemicals of concern that currently may affect fish and wildlife throughout the state, including selenium, mercury, PCBs, dioxins, and chlorinated pesticides

- Reductions in toxics that improve conditions for the successful recovery of federal and state threatened and endangered species, such as the delta smelt, desert pupfish, California brown pelican, bald eagle, California clapper rail, California tiger salamander, and western snowy plover

- Reductions in toxics that decrease adverse toxics-related impacts on aquatic and terrestrial wildlife in two important areas of California: the San Francisco Bay watershed and the Central Valley (see case studies in U.S. EPA, 1997)

- Reductions in the concentrations of both selenium and pesticides in the waters that feed the Salton Sea that may improve conditions for the restoration and maintenance of currently declining populations of wildlife, including threatened and endangered species such as the California brown pelican, peregrine falcon, bald eagle, Yuma clapper rail, and desert pupfish (see Case Studies in U.S. EPA, 1997)

- Improved water quality and associated improvements in survival, growth, and reproductive capacity of aquatic and aquatic-dependent organisms that will help restore and sustain California's ecological diversity.

7.0 BENEFITS METHODOLOGY ISSUES: CONTRIBUTION OF POINT SOURCES TO TOXICS-RELATED WATER QUALITY PROBLEMS

Estimating the benefits of implementation of the CTR is difficult because there is limited information regarding the contribution of point sources to the toxic-related water quality problems in California. This issue of attribution has important implications for the potential benefits of point source controls. Benefits analyses of water quality regulations may be able to utilize existing literature, applied research, and data to estimate society's values for water quality improvements. Often, there are limited data with which the contribution of point source controls to these improvements can be discerned.

To estimate the potential benefits of the CTR, EPA evaluated the limited available data on loadings from various sources to California watersheds. Based on these data, EPA developed ranges of values to reflect the potential contribution of point sources to current toxic-related water quality problems in San Francisco Bay, other bays and estuaries, and freshwater. EPA then used these assumptions to estimate the benefits of the proposed rule (see Chapter 8). This chapter describes the data EPA used to develop the attribution assumptions, and the uncertainties surrounding these estimates, as presented originally in U.S. EPA (1997).

EPA solicited additional data and information on the relative contribution of point sources to toxic pollutant loadings in California waters in the EA that accompanied the proposed rule, however, commenters did not submit any new data. EPA then conducted a new literature search and contacted universities and organizations in search of additional data and studies. In general, the studies found very little detailed information and data. One study of the Santa Monica Bay watershed (California Regional Water Quality Control Board, 1997), however, contains an assessment of the relative loadings from point and nonpoint sources. As described in Section 7.2, EPA updated its analysis to incorporate this information.

7.1 SAN FRANCISCO BAY

EPA used two sources to characterize the relative contributions of point and nonpoint sources of toxic loadings in San Francisco Bay: Davis et al. (1991) and National Oceanic and Atmospheric Administration (NOAA)(1988a). Davis et al. estimated that 5,000 to 40,000 metric tons of at least 65 different pollutants are released annually into the San Francisco estuary from both point and nonpoint sources. They estimated point source loadings based on municipal (POTW) and industrial NPDES effluent monitoring data from 1984 to 1987. They estimated nonpoint source loadings using estimates of urban and nonurban runoff, riverine inputs, atmospheric deposition, oil spills, and contributions from dredging activities. Estimates of urban runoff and dredging came from Gunther et al. (1987)[1]. Estimates of nonurban runoff were based on a NOAA model that factors in sediment loss from nonurban lands and average trace metal concentrations in soil.

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[1] Other studies suggest that Gunther et al. (1987) may have underestimated the contribution from dredging activities.

Riverine inputs were based on pollutants from the Sacramento and San Joaquin rivers. All pollutants transported past the cities of Sacramento and Vernalis were considered riverine input from the Sacramento and San Joaquin Rivers, respectively. The loading from the Sacramento River was based on a 1987–88 study of selenium cycling conducted by the California Department of Water Resource. The loading from the San Joaquin River was based on 1985–87 water quality data collected by the U.S. Geological Survey (USGS). Atmospheric deposition loadings were based on measurements in other parts of the United States, as reported in Gunther et al. (1987).

NOAA (1988a) estimated that approximately 22,000 metric tons of toxic substances are released annually into the San Francisco estuary. They estimated point source loadings based on municipal (POTW) and industrial effluent monitoring data. Estimates of nonpoint source loadings include urban and nonurban runoff, and riverine inputs. NOAA (1988a) estimated urban runoff by combining runoff coefficients and pollutant concentrations with:

- Estimates of total county and city urban land area and population (U.S. Bureau of Census, 1980)
- POTW wastewater and storm water conveyance and treatment data (U.S. EPA, 1982)
- Weather station and precipitation data (provided by NOAA's National Climatic Data Center)
- Information on urban land use activities (obtained from USGS's Land Use Data Analysis System).

To estimate the contribution of nonurban runoff to loadings, NOAA (1988a) examined areas where:

- Farming, silviculture, or other activities have exposed soil to wind, rain, and runoff
- Soil is most erodible
- Large amounts of chemical fertilizers and pesticides have been applied
- Sufficient runoff exists to transport pollutants.

NOAA (1988a) obtained the majority of the data for its analysis from the USGS's Land Use Data Analysis System, the U.S. Department of Agriculture's 1982 National Resource Inventory, and a study by Shacklette and Boerngen (1984). NOAA (1988a) also estimated riverine inputs from the Sacramento and San Joaquin rivers using raw USGS data in a simulation model.

Exhibit 7-1 presents the estimated contribution of point sources to toxic loadings in the San Francisco Bay based on the Davis et al. (1991) and NOAA (1988a) studies. Using these studies, EPA developed toxicity-weighted averages across the pollutants evaluated to reflect the contribution of point sources to San Francisco Bay. This resulted in an estimate of 3.4% for the NOAA data based on the median weight for the class of chlorinated hydrocarbon pesticides,[2] and a range of 1.5% to 7.1% for the Davis et al. data. In light of the uncertainties in the estimates, for the purposes of estimating the potential benefits of the point source controls of the CTR, EPA assumed that point sources contribute between 1% and 10% of total toxic loadings to San Francisco Bay.

Exhibit 7.1 Estimated Contribution of Point Sources to Toxic Pollutant Loadings in San Francisco Bay (Toxic-Weighted)

Pollutant	NOAA[1]	Davis et al.[2]
Zinc	5.2%	4.0–18.9%
Copper	5.5%	4.0–11.9%
Nickel	Not Estimated	26.4–27.6%
Lead	7.4%	2.3–8.5%
Chromium	2.6%	0.8–5.7%
Arsenic	11.4%	3.8–6.1%
Cadmium	15.6%	29.5–64.8%
Selenium	Not Estimated	28.4–28.4%
Mercury	6.9%	26.4–51.8%
Chlorinated Hydrocarbon Pesticides	51.5%	Not Estimated

[1] Toxic-weighting based on loadings from the NOAA (1988a)
[2] Source: U.S. EPA (1997); range based on data from Davis et al. (1991).

EPA's analysis is subject to a number of uncertainties and limitations. First, Davis et al. and NOAA relied on assumptions about concentrations below the detection limit to estimate pollutant loadings. However, the concentration of pollutants below the deduction limit is unknown. Second, studies do not include estimates for some point and nonpoint sources of pollutants such as "historic" loadings from contaminated sediment or point source mine drainage. Third, Davis et al. and NOAA classify riverine inputs as nonpoint sources. It is possible that a portion of these riverine inputs is attributable to point sources; however, this could not be estimated based on available data. Fourth, the data from both studies are based on discharges from the early and mid-1980s and, therefore, may not be representative of current conditions in San Francisco Bay. Finally, Davis et al. did not estimate the contribution from pollutants, other than selenium, in the Sacramento River, and did not have local data for the estimates of urban runoff and atmospheric deposition. The use of data on atmospheric deposition from other parts of the United States would tend to overestimate nonpoint source loadings (and thus underestimate point source loadings) because there are relatively fewer air sources of toxics that

[2] The weights for hydrocarbon pesticides range from 0.35 for trichlorophenol to 57,000 for dieldrin, with a median for the class of 100.

might reach the bay given the prevailing westerly winds off the ocean.

7.2 OTHER BAYS AND ESTUARIES

EPA used NOAA's National Coastal Pollutant Discharge Inventory (1988b and 1988c) to estimate the relative contribution of point sources to toxic loading in five California bays: San Diego, Humboldt, Monterey, Santa Monica, and San Pedro (**see Exhibit 7-2**)[3].

Exhibit 7-2. Estimated Contribution of Point Sources to Toxic Pollutant Loadings in Other California Bays (Toxic-Weighted)[1]

Pollutant	Nonurban Bays		Urban Bays		
	Monterey Bay	**Humboldt Bay**	**San Diego Bay**	**Santa Monica Bay**	**San Pedro Bay**
Arsenic	57.1%	32.7%	87.7%	89.9%	87.4%
Cadmium	83.8%	40.2%	100.0%	100.0%	100.0%
Chromium	15.2%	34.8%	95.2%	88.4%	87.8%
Copper	16.1%	17.0%	86.8%	89.4%	78.7%
Lead	29.9%	19.4%	41.0%	66.7%	26.9%
Mercury	75.6%	8.7%	90.1%	87.9%	81.3%
Zinc	23.7%	27.0%	80.9%	78.2%	70.6%
Chlorinated hydrocarbon pesticides	N/A	N/A	93.0%	99.1%	94.0%
Toxic-Weighted Average	**22.2%**	**33.1%**	**91.1-92.0%***	**87.9-93.3%***	**82.6-88.5%***

[1] Toxic-weighting based on loadings from NOAA, 1981-1984 (nonurban bays) and NOAA, 1988b and 1988c (urban bays). NOAA assessed the following point sources: POTWs, industrial effluents, and power plant effluent. NOAA assessed the following nonpoint sources: urban runoff, cropland runoff, forestland runoff, rangeland runoff, irrigation return flows, and upstream sources.
* Lower bound of range based on median toxic weight for pesticides (100); upper bound of range based on mean toxic weight for pesticides (5,300).
Source: U.S. EPA (1997).

EPA developed toxicity-weighted averages across the pollutants evaluated to reflect the contribution of point sources to each bay. The data showed point sources account for 23.2% and 33.1% of loadings in the nonurban bays (Monterey and Humboldt Bays, respectively), and 91.1%, 87.9%, and 82.6% in the urban bays (San Diego, Santa Monica, and San Pedro Bays, respectively).

In addition, for this revised analysis, EPA combined additional data for the Santa Monica Bay watershed (California Regional Water Quality Control Board, 1997) for seven pollutants with the NOAA data. These new data suggest that point source loadings of copper, lead, and zinc decreased between 1986 and 1992 and that, currently, nonpoint sources are the predominant

[3] Only two of these bays (San Diego and Humboldt) are enclosed bays covered by the rule. EPA assumed that the data for the non-enclosed bays generally will be applicable to enclosed bays.

contributors of lead and zinc (California Regional Water Quality Control Board, 1997)[4] . However, this additional data for copper, lead, and zinc does not change the toxic-weighted average for Santa Monica Bay. This is because mercury and chromium VI have such high toxic weights (500 and 35.5, respectively) compared to the toxic weights for arsenic, cadmium, copper, lead, and zinc (4, 5.2, 0.47, 1.8, and 0.051, respectively). As a result, the overall toxic-weighted average is more closely linked to the point source contribution of mercury and chromium-VI and is not influenced by the new data for Santa Monica Bay.

In general, the available data indicated that urban bays tend to have a greater portion of toxic loadings originating from point sources than do nonurban bays. The data also reveal that the contribution of point sources is much higher in the San Diego, Santa Monica, and San Pedro urban bays than EPA estimated for the urban San Francisco Bay. The reason for this discrepancy is not readily apparent. For urban bays, EPA averaged the mean toxic-weighted point source contributions for the three urban bays as well as the midpoint of the range of point source contribution for San Francisco Bay [(91.1 + 87.9 + 82.6 + 5.0)/4 = 66.7] to estimate that point sources account for 67% of toxic-weighted loadings to urban bays. EPA estimated that point sources account for 28% of toxic-weighted loadings to nonurban bays by averaging the mean toxic-weighted point source contributions for the two nonurban bays [(23.2 + 33.1)/2 = 28.2]. However, these percentages cannot be directly used to attribute benefits to the CTR because EPA was not able to estimate the proportion of benefits that occur in urban bays versus nonurban bays. Therefore, EPA developed a weighted average estimate of the point source contribution of toxic pollutant loadings to California bays and estuaries based on the population and land area around urban and nonurban bays.

Scaling by population implicitly assumes that benefits are proportional to the population living in different areas (e.g., that more fishing occurs in urban bays than nonurban bays) (U.S. EPA, 1997). EPA identified the relevant enclosed bays covered by the rule,[5] and obtained total population living within 10 miles of each bay.[6] The method yields an estimate of approximately 3.1 million people living near urban bays, and 275,000 people living near nonurban bays (U.S. EPA, 1997), and results in a population-weighted attribution estimate of 64%. To scale by land area surrounding the bays, EPA compiled data on total acreage of each of the urban and nonurban bays from California's WQA database (State Water Resources Control Board, 1994) (U.S. EPA, 1997). This approach yielded a land area-weighted average estimate of 42%. EPA used this

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[4] The point source contributions for copper, lead, and zinc were 89.9%, 52.6%, and 75.1% respectively in 1986. In 1992, these contributions were 59.1%, 6.7%, and 40.4% (California Regional Water Quality Control Board, 1987).

[5] Urban bays include San Diego Bay, Mission Bay, Upper and Lower Newport Bay, and Los Angeles-Long Beach Harbor. Nonurban bays include Humboldt Bay, Bodega Harbor, Morro Bay, Drakes's Estero, Tomales Bay, and Carmel Bay (State Water Resources Control Board, 1991).

[6] Census tract-level population data were taken from the 1990 census and aggregated using geographic information system software.

42% to 64% range to attribute potential benefits of the implementation of the CTR in bays and estuaries. The limitations and uncertainties noted in Section 7.1 also apply to this estimated range.

7.3 FRESHWATER RESOURCES

Because of data and resource limitations, EPA could not assess the relative source contribution to specific freshwater resources in California. EPA used data for the Sacramento and San Joaquin rivers, and information on the influence of permitted mines on freshwaters, to develop a statewide estimate of the relative contribution of point sources to toxic pollutant loadings in freshwater. This estimate is based on data from the Central Valley RWQCB. The data include loadings from urban runoff, agricultural drainage, mining drainage, and industrial and municipal point sources. **Exhibit 7-3** shows the percentage of loadings attributable to point sources on each river.

Exhibit 7-3. Estimated Contribution of Point Sources to Toxic Pollutant Loadings in California Rivers (Toxic-Weighted)[1]

Pollutant	Sacramento River	San Joaquin River
Arsenic	22.3%	3.1%
Cadmium	81.6%	5.8%
Copper	72.4%	2.9%
Lead	6.1%	2.8%
Zinc	72.9%	7.2%

[1] Toxic-weighting based on loadings from Central Valley RWQCB, Mass Emission Strategy – Load Estimates.
Source: U.S. EPA (1997).

Because of the influence of permitted mine discharges on the Sacramento River, point source contributions for all pollutants are greater for the Sacramento River than for the San Joaquin River. Using a toxicity-weighted average across all five pollutants, EPA estimated that 46.3% of loadings to the Sacramento River and 3.4% of loadings to the San Joaquin River are associated with point sources.

EPA then used these estimates to develop a weighted-average contribution of point sources to toxic loadings in freshwater by using the estimate for the Sacramento River for river miles under the influence of major permitted mines and the estimate for the San Joaquin River for all other river miles. In California, there are five major mines that have NPDES permits, all of which are located in the Sacramento River watershed.[7] Therefore, EPA estimated that 0.001 percent of all lake acres and 0.05 percent of all river miles are under the influence of the five major NPDES permitted mines (Water Resources Control Board, 1996). Using the estimated point source

[7] A very small percentage of mines in California are permitted because most mines are inactive. EPA estimated river miles under the influence of mining for Lake Shasta (Alta Gold mine and Remedial Recovery), Sacramento River (Iron Mountain mine), South Feather River (Plumas Gold mine), and Pine Creek (U.S. Tungsten Corporation). This analysis does not account for multiple mines under a single permit.

contribution for the Sacramento River and the San Joaquin River, 46% and 30% respectively, EPA then calculated a weighted-average point source contribution of 3% for lakes and 3% for rivers.[8] The 3% for freshwater lakes also is applied to saline lakes.

EPA's analysis for freshwater also is subject to a number of uncertainties and limitations. First, the concentration of pollutants below the detection limit is not known. For this analysis, all samples below the detection limit were assumed to be zero. Second, only a subset of cities in the Central Valley region were incorporated in the estimate of urban runoff. Third, the use of effluent concentration data from the Sacramento County POTW may not be representative of effluent from other facilities. Finally, historic loadings in sediments may not be accounted for in the estimates.

7.4 SUMMARY

Exhibit 7-4 summarizes EPA's estimate of the relative contribution of point sources to total loadings of toxic pollutants in California waters. These estimates represent the toxic-weighted average across the pollutants evaluated. **Exhibit 7-5** summarizes the *key* uncertainties and limitations in the estimates. Because the direction and magnitude of biases generally is not known, it is difficult to assess their overall impact on the estimates.

[8] The calculation assumes a 46% point source contribution for mining impaired water bodies (0.001% of lakes and 0.05% of rivers) and a 3% contribution in other water bodies. The calculations are 0.46 × (0.001%) + 0.03 × (1-0.001%) = 3.00%; 0.46 × (0.05%) + 0.03 × (1-0.05%) = 3.02%.

Exhibit 7-4. Estimated Share of Total Toxic Pollutant Loadings Attributable to Point Sources for California Water Bodies

Water Body	Toxic Pollutant Loadings Attributable to Point Sources (%)
San Francisco Bay	1-10
Other bays and estuaries	42–64[1]
Freshwaters and saline lakes	3

[1] The lower-bound estimate is for nonurban bays and the upper-bound estimate is for urban bays.
Source: Based on EPA analysis of NOAA (1988a); NOAA (1988b); NOAA (1988c); Davis, et al. (1991); California RWQCB (1997); Central Valley RWQCB; and California 1994 WQA database, as originally presented in U.S. EPA (1997).

Exhibit 7-5. Key Uncertainties in the Analysis of Relative Point Source Contribution

Uncertainty	Relative Significance	Potential Direction of Bias on Point Source Contribution to Total Loadings		
		Overstate	Understate	Indeterminant
Generalized from limited loadings data for a small set of water bodies to the extensive system of salt and freshwater in California.	High			✔
Analysis based on a limited set of pollutants. Little information on pesticides. No information on PCBs, dioxin, and certain metals (e.g., silver).	High		✔	
"Historic" loadings not fully accounted for.	High	✔		
Studies used classify riverine inputs as nonpoint sources. Some of these loadings may have originated from point sources.	Medium		✔	
Point source contributions for San Francisco Bay are much lower than for the other urban bays.	Medium			✔

Source: Adapted from U.S. EPA (1997).

8.0 QUANTIFIED AND MONETIZED BENEFITS ESTIMATES

EPA quantified and monetized three categories of potential benefits from implementation of the CTR: (1) human health risk reductions, (2) recreational angling benefits, and (3) passive use values. These benefits estimates are presented in Sections 8.1 through 8.3. In addition, Section 8.4 describes potential categories of benefits that are expected to result from the rule but that EPA could not monetize. Section 8.5 provides a summary of the benefits estimates.

The analysis presented below resembles the analysis that accompanied the proposed CTR (the results have been updated to incorporate the revised estimates of pollutant loading reductions and a slight modification to how the reductions are incorporated). However, in response to comments on the proposed CTR and accompanying EA, EPA continued to search the literature for California-specific valuation research that may be relevant to estimating the benefits of the rule. Below, the results of three studies (Carson et al., 1994; Loomis et al., 1991; and Cooper and Loomis, 1991) are incorporated into this revised analysis. In addition, where possible, EPA updated the data underlying the analysis.

8.1 HUMAN HEALTH BENEFITS

EPA assessed the human health risks from the consumption of contaminated fish tissue, and the potential reductions in these risks expected to result from implementation of the CTR, for two populations of anglers: San Francisco Bay anglers and freshwater anglers in California.

San Francisco Bay represents one of the most important noncommercial fisheries among the bays and estuaries covered by the rule. EPA conducted the assessment for San Francisco Bay anglers as a case study example of the health risks for anglers fishing in enclosed bays and estuaries. In addition, the bay has been adversely affected by toxic pollution, as evidenced by a recently issued fish consumption advisory (FCA). This advisory is due to the concentrations of mercury, PCBs, dioxin, and pesticides in fish from the bay. Despite the issuance of the advisory in December 1994, the bay remains a popular area for anglers (U.S. EPA, 1997). However, because only two other health advisories have been issued for enclosed bays and estuaries in California, this case study may represent an upper-bound estimate of baseline health risks associated with enclosed bays and estuaries.

The freshwater resources in the state also have been adversely affected by toxic pollution. Fish consumption advisories have been issued for nine inland water bodies, including numerous reservoirs, rivers, and creeks in Santa Clara County and the Grassland Area of the Kesterson National Wildlife Refuge in Merced County. **Exhibit 8-1** summarizes the FCAs in place for inland waters and enclosed bays and estuaries in California. **Exhibit 8-2** illustrates the location of these FCAs, as well as the location of NPDES-permitted point source discharges and the density of resident fishing license sales by county.

Exhibit 8-1. Fish Consumption Health Advisories in California

Water Body/Location	Advisory for General Population: Avoid Consumption	Advisory for General Population: Limit Consumption[2]	Advisory for Sensitive Populations[1]: Avoid Consumption	Advisory for Sensitive Populations[1]: Limit Consumption[2]	Contaminants of Concern
Inland Surface Waters					
New River	All species		All species		Pesticides Biological contaminants
Clear Lake (Lake County)		1 lb per month ▸ Largemouth bass over 13" ▸ Channel catfish over 24" ▸ Crappie over 12" 2 lbs per month ▸ Largemouth bass under 13" 3 lbs per month ▸ Channel catfish under 24" ▸ Crappie under 12" ▸ White catfish 6 lbs per month ▸ Brown bullhead ▸ Sacramento blackfish 10 lbs per month ▸ Hitch	All species		Mercury
Lake Nacimiento (San Luis Obispo County)		4 meals per month ▸ Largemouth bass	Largemouth bass		Mercury
Lake Herman (Solano County)		1 lb per month ▸ Largemouth bass	Catfish		Mercury
Lake Berryessa (Napa County)		1 lb per month ▸ Largemouth bass over 15" ▸ Smallmouth bass 2 lbs per month ▸ Largemouth bass under 15" ▸ White catfish 3 lbs per month ▸ Channel catfish 10 lbs per month ▸ Rainbow trout	All fish		Mercury
Grassland Area Kesterson National Wildlife Refuge (Merced County)	Catfish	Max. of 4 oz. every 2 weeks ▸ All fish	All fish		Selenium
Salton Sea		Max. of 4 oz. every 2 weeks ▸ Croaker ▸ Sargo ▸ Tilapia ▸ Orangemouth corvina	All fish		Selenium
Bays and Estuaries[3]					
San Francisco Bay	Striped bass> 35"	Maximum of 2 meals per month ▸ All sport fish	Striped bass> 27" Shark> 24"	Maximum of 1 meal per month ▸ All sport fish	Mercury PCBs Dioxins Pesticides
Belmont Pier/Pier J (Los Angeles Harbor)		Maximum of 2 meals per month ▸ Surf perch			DDT, PCBs
Los Angeles/Long Beach Harbors (esp. Cabrillo Pier)	White croaker	Maximum of 2 meals per month ▸ Queenfish ▸ Surf perch ▸ Black croaker			DDT, PCBs

[1] California EPA defines sensitive populations as women who are pregnant, who may become pregnant, who are breast-feeding, and children under 6 years of age.
[2] California EPA defines a meal as 6 to 8 oz. (170 g to 227 g) of fish for a 154 lb (70 kg) individual. Meal size should be adjusted according to body weight (roughly 1 oz. of fish per 20 lbs of body weight).
[3] In addition to these advisories, California EPA has issued consumption warnings for the following ocean sites in Southern California that are not included within the scope of the California Toxics Rule: Newport Pier, Redondo Pier, Malibu Pier, Short Bank, Malibu/Point Dume, Point Vicente, Palos Verdes-Northwest, White's Point, Los Angeles/Long Beach Breakwater (ocean side), and Horseshoe Kelp. Detailed information on these advisories is available in the California Sport Fishing Regulations Handbook.
Source: U.S. EPA (1997).

EPA assessed baseline human health risks (cancer and systemic effects) based on reported contaminant levels in fish tissue samples collected from San Francisco Bay and freshwater fisheries throughout California. EPA then estimated the potential reduction in baseline risk levels that might result from implementation of the CTR. The approach used follows standard EPA methodology for estimating health risks as described in detail in U.S. EPA (1997).

8.1.1 Estimating the Exposed Population

EPA estimated the potentially exposed population for San Francisco Bay and for statewide freshwater resources based on information regarding recreational anglers. Consequently, this analysis does not include health risks to non-angler family members that consume fish obtained from recreational angling,[1] nor does it consider the benefits to individuals that consume commercially caught fish. EPA assumed that consumption of commercially caught fish from areas affected by implementation of the CTR would be small relative to the consumption of commercially caught fish from other locations. If there were consumption of substantial quantities of commercially caught fish from areas affected by the CTR, benefits would be underestimated.

San Francisco Bay

EPA estimated the potentially exposed angler population for the case study based on the eight counties in the immediate San Francisco Bay area. A survey of fishing activity in central and northern California reported that there are approximately 332,000 saltwater anglers in the eight counties adjacent to San Francisco Bay, and the bay is the destination for approximately 50% of the trips taken near the bay (the area north of Stinson Beach to south of Davenport) (National Marine Fisheries Service, 1987). EPA assumed that one-half of the anglers fish exclusively in the bay and one-half fish exclusively at other Pacific Ocean sites. Applying this assumption, EPA estimated that there are 166,000 anglers using San Francisco Bay.

A portion of the estimated 166,000 saltwater anglers that use San Francisco Bay also may participate in freshwater angling. The 1996 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation (U.S. Department of Interior, Fish and Wildlife Service, and U.S. Department of Commerce, Bureau of the Census, 1998) indicates that 46% of the saltwater anglers (adults and children) in California fish exclusively in saltwater and 54% fish in both saltwater and freshwater. EPA assumed that anglers that split their time spend half of their time using each resource. Therefore, of the 166,000 anglers that use San Francisco Bay for saltwater angling, EPA estimated that 76,360 anglers fish exclusively in the Bay, and the other 89,640 split their time equally between the bay and freshwater resources. Thus, EPA estimated that approximately 121,000 full-time equivalent anglers use San Francisco Bay.

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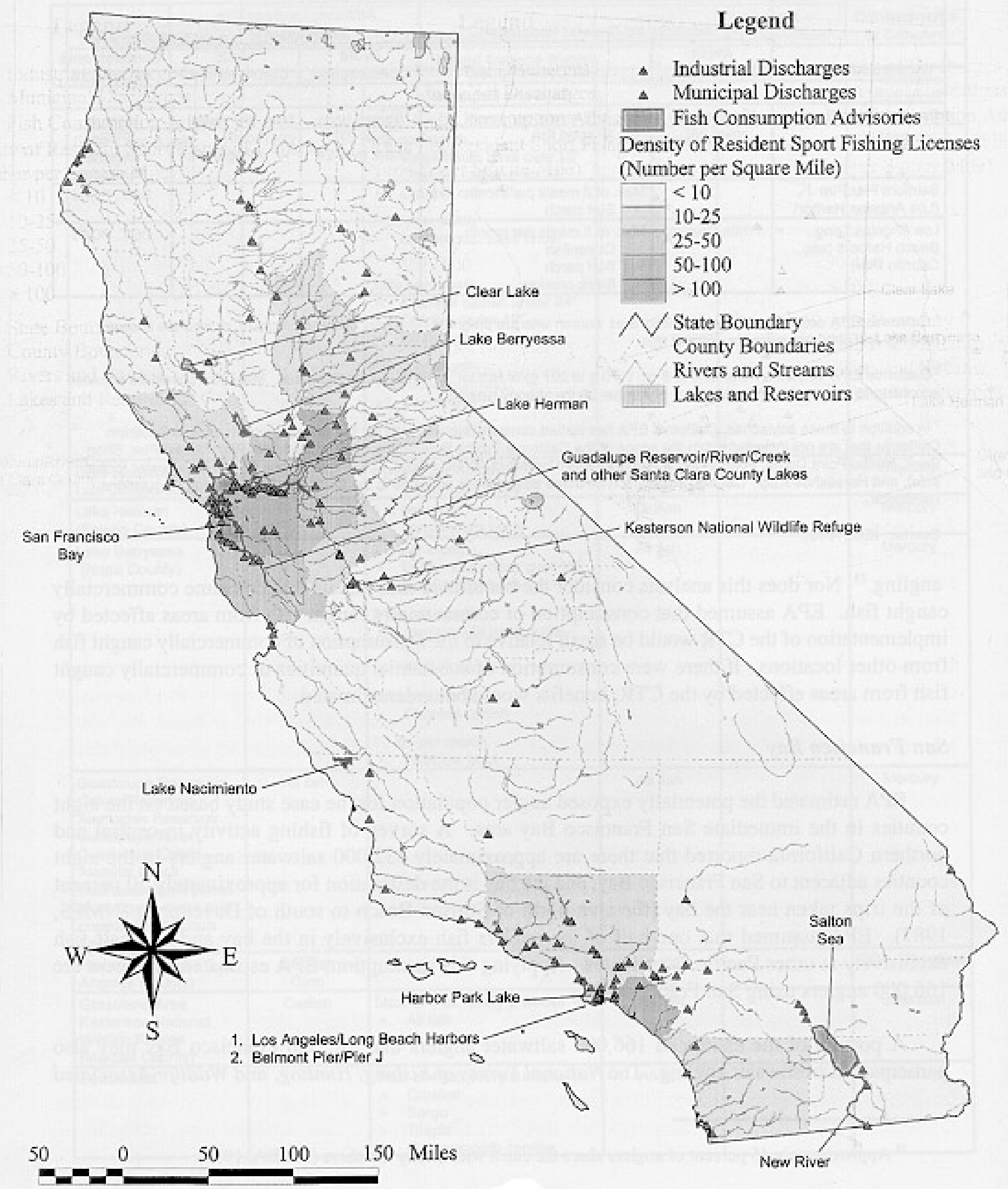
[1] Approximately 45% of anglers share the catch with family members (Save San Francisco Bay Association, 1995).

EXHIBIT 8-2. DENSITY OF FISHING LICENSES IN RELATION TO THE LOCATION OF POINT SOURCE DISCHARGERS IN CALIFORNIA



Because this estimate is based on data collected before the imposition of a FCA for San Francisco Bay, EPA adjusted the population down to account for behavioral responses of anglers to FCAs. Recent literature suggests that between 10% and 37% of anglers take fewer trips in response to FCAs (Fiore et al., 1989; Silverman 1990; Knuth et al., 1993; Knuth and Connelly, 1992; Vena, 1992; West et al., 1993). However, these anglers may not eliminate trip-taking. Therefore, EPA assumed that the FCA resulted in a 10% reduction in anglers using San Francisco Bay (This reduction was not likely to have been offset by population growth since resident fishing license sales in the eight counties adjacent to San Francisco Bay fell 22% between 1987 and 1994). EPA's adjusted estimate of full-time equivalent anglers for San Francisco Bay is 108,900.

Freshwater Resources

EPA used the 1996 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation (U.S. Department of Interior, Fish and Wildlife Service, and U.S. Department of Commerce, Bureau of the Census, 1998) and license sales reported by the California Department of Fish and Game for 1996 to estimate the number of freshwater anglers in California. The survey indicated that there were 2.7-million resident anglers in California, including adults and children. Sixty-three percent of the adult anglers fish exclusively in freshwater, 20% split their angling activity between fresh and saltwater, and the remaining 17% of anglers fish exclusively in saltwater. Assuming that children apportion their angling time in the same way as adults, EPA estimated that 1.7-million anglers fish exclusively in freshwater, and 551,000 split their time between fresh and saltwater resources. EPA assumed that anglers that split their time spend half of their time using each resource. Based on this information, EPA calculated that nearly 2-million full-time equivalent anglers use freshwater resources in California.

EPA reduced this estimate of freshwater anglers by the number of 1-day license sales in 1996 (329,730) to remove infrequent anglers from the estimate of potentially exposed anglers. This calculation uses the proportions for total angling time to apportion the number of 1-day licenses between salt and freshwater resources. Using this approach, EPA estimated a potentially exposed population of 1.7 million full-time equivalent anglers using freshwater resources in California.

8.1.2 Fish Consumption

EPA estimated fish consumption rates for both San Francisco Bay and freshwater anglers using the Santa Monica Bay Seafood Consumption Study (MBC Applied Environmental Services, 1994). For this study, the Santa Monica Bay Restoration project conducted a survey of 554 anglers fishing from beaches, piers, private boats, party boats, and charter boats to determine the level and nature of sport-caught fish consumption. This study reported a median consumption rate of 21.4 g/day and a 90th percentile consumption rate of 107.1 g/day for consuming anglers. Although these estimates were developed by interviewing only consuming anglers, EPA applied them to total anglers because they are supported by fish consumption rates for all anglers

(**Exhibit 8-3**). To the extent that the study does not accurately characterize the fish consumption of anglers using freshwater resources, it will lead to an overestimate or under estimate of risks.

Exhibit 8-3. Consumption Rates for Recreational Anglers

Study	Type of Fishery	Angler Population	Consumption Rate
U.S. EPA, 1989a	Sport-caught fish, nationally	All anglers	20 g/day
Puffer et al., 1981	Sport-caught fish from Los Angeles Bay, CA	Anglers who had creeled fish	37 g/day
MBC Applied Environmental Services, 1994	Sport-caught fish from Santa Monica Bay, CA	Anglers who consume fish	21 g/day

Source: U.S. EPA (1997).

8.1.3 Fish Tissue Contaminant Concentrations

EPA used available data (see U.S. EPA, 1997, Appendix F) to calculate the arithmetic mean of fish tissue contaminant concentrations for San Francisco Bay and for statewide freshwater resources. To determine the concentrations, EPA used one-half the MDL for samples in which contaminants were reported as non-detects (U.S. EPA, 1993).

San Francisco Bay

EPA obtained fish tissue contaminant levels from a 1994 study conducted by the San Francisco Regional Water Quality Control Board (SFRWQCB). The study included fish tissue samples from 16 sampling locations selected to provide a broad geographic coverage of the bay. The sampling survey (SFRWQCB, 1994) included fillets of white croaker, striped bass, perch, and shark. The fish tissue samples were prepared for chemical analysis according to the most common means of consumption (croaker and surf perch fillets with skin, and shark and striped bass without skin).

EPA relied on catch rates reported in the National Marine Fisheries Service Marine Recreational Fishing Statistics Survey of the Pacific Coast (1987, 1988, 1989, and 1993) (**Exhibit 8-4**) to develop species-weighted fish tissue contaminant concentrations for San Francisco Bay. The species consumption weighting factors are used to allocate the amount of fish consumed in proportion to anglers' exposure to individual fish species. EPA assumed that keep rates are comparable across the four species in the analysis. This approach was used for both San Francisco Bay and freshwater, but may not accurately reflect species-weighted fish tissue contaminant concentrations because the approach is based on the number of fish caught rather than the mass of edible fish tissue. In addition, fish tissue contaminant data for jacksmelt, a frequently caught species, was not available. However, the relatively small degree of variation in risks associated with consuming the four species that were included in the analysis suggests that the lack of data on mass consumed is unlikely to significantly overestimate or underestimate bay angler risks (U.S. EPA, 1997).

Exhibit 8-4. Species Weights for San Francisco Bay Fish Consumption

Species	Number of Fish Caught	Consumption Weighting Factors[3]
White croaker	532	43.1%
Surf perch[1]	432	35.0%
Striped bass	171	13.9%
Shark[2]	99	8.0%
Total	1,234	100.0%

[1] Includes shiner, walleye, pile, black, and rubberlip surf perch.
[2] Includes brown, smoothhound, and leopard shark.
[3] Represents the percentage of the total catch for each species. Keep rates are assumed to be comparable for the four species.
Source: National Marine Fisheries Service Marine Recreational Fishing Statistics Survey, Pacific Coast, 1987–1989 and 1993, as cited in U.S. EPA (1997).

Freshwater Resources

EPA obtained fish tissue contaminant concentration data from samples taken between 1988 and 1993 by the California Toxic Substances Monitoring Program. Despite the wide representation of freshwater bodies (224 sampling locations for metals and 170 for organics), this database may not be representative of all freshwater bodies. Sampling under this program has generally been targeted to water bodies with known or suspected water quality impairments. The sampling survey included samples of 32 different freshwater fish species, which EPA combined into five broad groups: trout, bass, catfish, panfish, and other. EPA developed species-weighted fish tissue contaminant concentrations from estimates of fishing activity and keep rates by species (**Exhibit 8-5**). The species consumption weighting factors are used to allocate the amount of fish consumed in proportion to anglers' exposures to individual fish species.

Exhibit 8-5. Species Weights for Freshwater Fish Consumption

Species	Annual Fishing Activity[1] (number of days)	Keep Rate[2]	Keep-Rate Weighted Days[3]	Consumption-Weighting Factors[4]
Trout	16,292	25%	2,660	28.0%
Bass	10,431	25%	1,541	16.2%
Catfish	3,972	80%	3,278	34.6%
Panfish	1,457	90%	1,679	17.7%
Other	5,455	25%	329	3.5%
Total	**28,987[5]**	—	**9,487**	**100.0%**

[1] Source: U.S. Department of Interior, Fish and Wildlife Service, and U.S. Department of Commerce, Bureau of the Census, 1998.
[2] Source: California Department of Fish and Game (1995). Keep rates for bass and trout were 20% to 25%. For purposes of this analysis EPA assumed that the keep rates for bass, trout, and "other" species were 25% (Dennis Lee, California Department of Fish and Game, personal communication, August 1995).
[3] Calculated by multiplying the annual fishing activity days for each species by the keep-rate for that species. The total keep-rate weighted days is a sum of the keep-rate weighted days for all species.
[4] Calculated by dividing the keep-rate weighted days for each species by the total keep-rate weighted days.
[5] Represents total number of fishing days per year. Does not equal the sum of individual species days because more than one species may have been caught during a single fishing day.

8.1.4 Baseline Risk Levels

EPA calculated exposure based on the assumption that each fish contained all contaminants listed at the concentrations shown in U.S. EPA (1997). **Exhibit 8-6** reports the assumed toxicity values for cancer and systemic effects. EPA used standard assumptions regarding length of residence, 70 years, and body weight, 70 kg (U.S. EPA, 1989b).

Exhibit 8-6. Toxicity Values and Contaminants Evaluated in Each Analysis

Contaminant	CSF[1] $(mg/kg\text{-}day)^{-1}$	RfD[1] (mg/kg-day)	San Francisco Bay	Freshwater Resources
Cadmium	NA	1.0×10^{-3}	✔	
Chlordane	1.3	6.0×10^{-5}	✔	✔
Copper	NA	3.7×10^{-2}	✔	✔
4,4-DDT	0.34	5.0×10^{-4}	✔	✔
Dieldrin	16.0	5.0×10^{-5}	✔	✔
Dioxin	1.50×10^{5}	NA	✔	
Endosulfan	NA	6.0×10^{-3}		✔
Endrin	NA	3.0×10^{-4}		✔
Fluoranthene	NA	4.0×10^{-2}	✔	
Fluorene	NA	4.0×10^{-2}	✔	
HCH-alpha	6.3	NA	✔	✔
HCH-beta	1.8	NA	✔	
HCH-gamma	1.3	3.0×10^{-4}	✔	✔
Heptachlor Epoxide	9.1	1.3×10^{-5}	✔	
Heptachlor	4.5	5.0×10^{-4}	✔	
Hexachlorobenzene	1.6	8.0×10^{-4}	✔	✔
Mercury	NA	1.0×10^{-4}	✔	✔
Nickel	NA	2.0×10^{-2}		✔
PCBs[2]	2.0	2.0×10^{-5}	✔	✔
Pyrene	NA	3.0×10^{-2}	✔	
Selenium	NA	5.0×10^{-3}		✔
Silver	NA	5.0×10^{-3}	✔	
Toxaphene	1.1	NA		✔
Zinc	NA	3.0×10^{-1}	✔	✔

[1] CSF = cancer slope factor; RfD = reference dose. Toxicity values obtained from U.S. EPA's Integrated Risk Information System (4th Quarter, 1996), except for the HCH-gamma and dioxin CSFs and the copper RfD, which were obtained from U.S. EPA's Health Effects Assessment Summary Table, 1994.
[2] The CSF is based on EPA's revised October 1,1996, guidance for assessment of carcinogenic human health risks associated with PCB exposure.
NA = Not applicable
Source: U.S. EPA (1997).

San Francisco Bay

Exhibit 8-7 presents estimates of baseline cancer risks for San Francisco Bay anglers. EPA estimated that the individual excess lifetime cancer risk for anglers consuming a mixed species diet at an average consumption rate is 1.8×10^{-4} and statistical excess cancer cases per year at baseline are less than 1. (Potential benefits of the CTR are calculated for the average consumption rate.) However, for anglers consuming at the 90th percentile consumption rate, the individual excess lifetime cancer risk is 9.2×10^{-4}. These risks are dominated by PCBs and dioxin, which contribute 49% and 41%, respectively, to the cancer risk for an average angler.[2]

Exhibit 8-7. Baseline Cancer Risks for Recreational Anglers Consuming San Francisco Bay Fish

Contaminant	**Individual Excess Lifetime Cancer Risk**		**Population Cancer Risk[1] (excess cases per year)**	**Relative Contribution to Risk**
	Average Consumption (21.4 g/day)	**90th Percentile Consumption (107.1 g/day)**		
PCBs[2]	9.0×10^{-5}	4.5×10^{-4}	<1	49.0%
Dioxin	7.6×10^{-5}	3.8×10^{-4}	<1	41.2%
Dieldrin	7.8×10^{-6}	3.9×10^{-5}	0	4.2%
4,4-DDT	4.9×10^{-6}	2.4×10^{-5}	0	2.6%
Chlordane	3.9×10^{-6}	2.0×10^{-5}	0	2.1%
HCH-alpha	4.8×10^{-7}	2.4×10^{-6}	0	0.3%
Heptachlor Epoxide	3.8×10^{-7}	1.9×10^{-6}	0	0.2%
HCH-beta	1.7×10^{-7}	8.5×10^{-7}	0	0.1%
Heptachlor	1.6×10^{-7}	7.7×10^{-7}	0	0.1%
HCH-gamma	9.2×10^{-8}	4.6×10^{-7}	0	<0.1%
Hexachlorobenzene	8.1×10^{-8}	4.1×10^{-7}	0	<0.1%
Total	$\mathbf{1.8 \times 10^{-4}}$	$\mathbf{9.2 \times 10^{-4}}$	**<1**	**100.0%**

[1] Based on average fish consumption (21.4 g/day).
[2] Risk is based on an estimated concentration of PCBs in fish tissue that appears to be calculated by summing Aroclor congeners for 1248, 1254, and 1260. This may result in overstating baseline risks.
Source: U.S. EPA (1997).

Systemic (noncancer) risks are assessed by means of a hazard quotient (HQ) for each contaminant. The HQ is calculated by dividing the expected exposure level (dose) by the oral reference dose (RfD), where the oral RfD indicates the level of chronic exposure below which no adverse health effects are expected. Therefore, a HQ of 1.0 or greater implies that chronic

[2] Risk based on full-time equivalent anglers. Individual baseline risks may be lower by a factor of two for anglers that spend a portion of their time fishing in less-contaminated waters such as anglers that split their fishing activity between saltwater and freshwater, as discussed in Section 8.1.1.

chemical exposures exceed EPA-established "thresholds" of toxicity, and is indicative of potential for adverse health effects. The potential for detrimental health effects increases as the HQ increases above 1.0.

Exhibit 8-8 presents estimated baseline systemic risks for San Francisco Bay anglers. EPA estimated that the HQ for PCBs is 2.3. For anglers with high consumption rates (90th percentile), EPA estimated that the HQs for PCBs, mercury, and dioxin are 11.3, 3.8, and 2.5, respectively.[3]

Exhibit 8-8. Baseline Systemic Risks for Recreational Anglers Consuming San Francisco Bay Fish

Contaminant	Hazard Quotient[1]	
	Average Consumption (21.4 g/day)	90th Percentile Consumption (107.1 g/day)
PCBs	**2.26**	**11.31**
Mercury	0.75	**3.77**
Dioxin	0.51	**2.54**
Chlordane	0.05	0.25
4,4-DDT	0.03	0.14
Dieldrin	0.01	0.05
Zinc	0.01	0.04
Heptachlor Epoxide	<0.01	0.02
Copper	<0.01	0.01
Cadmium	<0.01	<0.01
HCH-gamma	<0.01	<0.01
Silver	<0.01	<0.01
Heptachlor	<0.01	<0.01
Hexachlorobenzene	<0.01	<0.01
Fluoranthene	<0.01	<0.01
Pyrene	<0.01	<0.01
Fluorene	<0.01	<0.01

[1] Hazard quotients above one shown in bold.
Source: U.S. EPA (1997).

Freshwater Resources

Exhibit 8-9 presents estimated baseline cancer risks for California freshwater anglers. EPA estimated that the individual excess lifetime cancer risk at baseline for anglers consuming a mixed species diet at an average consumption rate is 1.5×10^{-4} and there are less than four baseline excess statistical cancer cases per year. For anglers consuming a mixed species fish diet at the 90th percentile consumption rate, EPA estimated that the individual excess lifetime cancer

[3] Risk based on full-time equivalent anglers. The baseline HQ for all contaminants except mercury are estimated to be less than one for anglers that spend a portion of their time fishing in less contaminated waters.

risk is 7.6×10^{-4}. These risks are dominated by PCBs, toxaphene, 4,4-DDT, and dieldrin, which contribute 37%, 21%, 17%, and 16%, respectively, of the cancer risk for an average angler.

Exhibit 8-9. Baseline Cancer Risks for Recreational Anglers Consuming Freshwater Fish in California

Contaminant	Individual Excess Lifetime Cancer Risk		Population Cancer Risk[1] (excess cases per year)	Relative Contribution to Risk
	Average Consumption (21.4 g/day)	90th Percentile Consumption (107.1 g/day)		
PCBs	5.6×10^{-5}	2.8×10^{-4}	2	37.0%
Toxaphene	3.2×10^{-5}	1.6×10^{-4}	1	21.2%
4,4-DDT	2.5×10^{-5}	1.3×10^{-4}	1	16.6%
Dieldrin	2.4×10^{-5}	1.2×10^{-4}	1	16.0%
Chlordane	1.1×10^{-5}	5.3×10^{-5}	<1	7.0%
HCH-alpha	2.0×10^{-6}	1.0×10^{-5}	<1	1.3%
Hexachlorobenzene	1.0×10^{-6}	5.1×10^{-6}	<1	0.7%
HCH-gamma	4.6×10^{-7}	2.3×10^{-6}	<1	0.3%
Total	$\mathbf{1.5 \times 10^{-4}}$	$\mathbf{7.6 \times 10^{-4}}$	**5**	**100.0%**

[1] Based on average fish consumption (21.4 g/day).
Source: U.S. EPA (1997).

Exhibit 8-10 presents the potential baseline systemic risks for California freshwater anglers. EPA estimated that the baseline HQ for PCBs is 1.4. For anglers with high consumption rates (90th percentile), EPA estimated that the baseline HQs for PCBs and mercury are 7.0 and 3.1, respectively.

8.1.5 Potential Risk Reductions Attributable to the Rule

To estimate the potential risk reductions attributable to the CTR, EPA assumed that fish tissue contaminant concentrations would be reduced by the expected reduction in loadings multiplied by the assumed contribution of point sources to total loadings developed in Chapter 7 (**Exhibit 8-11**). As shown in Chapter 4, EPA developed two scenarios of potential baseline pollutant loadings and reductions in loadings attributable to the rule. These scenarios reflect the uncertainty underlying the analysis of potential costs to point source dischargers that results from limited data on the presence of toxic pollutants in the effluents below detectable levels.

Exhibit 8-10. Baseline Systemic Risks for Recreational Anglers Consuming Freshwater Fish in California

Contaminant	Hazard Quotient[1]	
	Average Consumption (21.4 g/day)	90th Percentile Consumption (107.1 g/day)
PCBs	**1.40**	**7.02**
Mercury	0.62	**3.12**
4,4-DDT	0.15	0.74
Chlordane	0.14	0.68
Dieldrin	0.03	0.15
Selenium	0.02	0.12
Endrin	0.01	0.04
Endosulfan	<0.01	0.02
Zinc	<0.01	0.02
Copper	<0.01	0.01
HCH-gamma	<0.01	0.01
Nickel	<0.01	0.01
Hexachlorobenzene	<0.01	<0.01

[1] Hazard quotients above one shown in bold.
Source: U.S. EPA (1997).

In the first scenario (the low cost scenario), EPA estimated baseline pollutant loadings of 1.8 million pounds per year and a reduction of 20.6% of this baseline resulting from the CTR. On a toxicity-weighted basis, this represents baseline loadings of 2.2 million pounds-equivalent per year and a reduction of 49.6%. Under the second scenario (the high cost scenario), EPA estimated baseline pollutant loadings of 153.9 million pounds per year and a reduction of 28.4% from this baseline resulting from the CTR. On a toxicity-weighted basis, this represents baseline loadings of 18.5 million pounds-equivalent per year and a reduction of 14.7%.

The two scenarios reflect use of different assumptions regarding whether pollutants are present in the effluent of point source dischargers with the high cost scenario using permit limits to establish the presence of pollutants (and not actual effluent monitoring data). Thus, the high cost scenario establishes a larger baseline loading of toxic pollutants from point sources compared to the low cost scenario. The high cost scenario also indicates a smaller percentage reduction as a result of the CTR although this increment is larger in absolute terms compared to that resulting under the low cost scenario.

Because of the uncertainty in the analysis of baseline pollutant loadings, EPA used the midpoint between the reductions estimated under the low and high cost scenarios for estimating potential benefits. For human health risk reduction benefits, this is implemented simply as the midpoint between the low and high cost scenario results for each pollutant analyzed (the percentage reductions are the same on an unweighted or toxicity-weighted basis).

Exhibit 8-11. Estimated Reduction in Fish Tissue Contaminant Concentrations Due to Implementation of the CTR

Contaminant	Statewide Reductions in Loadings[1] (%)	Reduction in Fish Tissue Concentration (%)	
		San Francisco Bay[2]	Freshwater Resources[3]
Cadmium	0.2	0	ne
Chlordane	0	0	0
Copper	14.9	0.1 - 1.5	0.4
4,4-DDT	0	0	0
Dieldrin	0	0	0
Dioxin	0	0	ne
Endosulfan	0	ne	0
Endrin	0	ne	0
Fluoranthene	0	0	ne
Fluorene	0	0	ne
HCH-alpha	0	0	0
HCH-beta	0	0	ne
HCH-gamma	25.2	0.3 - 2.5	0.8
Heptachlor Epoxide	0	0	ne
Heptachlor	0	0	ne
Hexachlorobenzene	48.1	0.5 - 4.8	1.4
Mercury	70.3	0.7 - 7.0	2.1
Nickel	9.2	ne	0.3
PCBs	25.9	0.3 - 2.6	0.8
Selenium	0	ne	0
Silver	38.5	0.4 - 3.9	ne
Toxaphene	1.0	ne	0
Zinc	2.4	0 - 0.2	0.1

[1] Represents the midpoint of the low and high cost scenario results.
[2] Calculated by multiplying the percent reduction in point source loading by the estimated point source contribution to total loadings (1%-10%)
[3] Calculated by multiplying the percent reduction in point source loading by the estimated point source contribution to total loadings (3%).
ne= Not evaluated

Exhibits 8-12 and **8-13** present the potential reductions in cancer risks for recreational anglers. EPA estimated reductions in statistical cancer cases for anglers with average consumption rates. Using an estimated value of a statistical life of $2.5 million to $9.0 million (American Lung Association, 1995) updated to first quarter 1998 dollars[4] and assuming all cancers are fatal,

[4] 1995 dollars were updated to first quarter 1998 dollars using the Consumer Price Index (CPI) as reported in the U.S. Department of Labor (U.S. Bureau of Labor Statistics, 1998). Note that there is currently a debate regarding the accuracy of the CPI.

potential human health benefits of reduced cancer cases in recreational anglers range from $0.10 million to $4.20 million per year[5].

Exhibit 8-12. Potential Effect of Implementation of the CTR on Cancer Risks for Recreational Anglers

Contaminant	Baseline Individual Excess Lifetime Cancer Risk		Post-CTR Individual Excess Lifetime Cancer Risk[1]	
	Average Consumption (21.4 g/day)	90th Percentile Consumption (107.1 g/day)	Average Consumption (21.4 g/day)	90th Percentile Consumption (107.1 g/day)
San Francisco Bay				
Total[2]	1.84×10^{-4}	9.20×10^{-4}	1.54×10^{-4} – 1.60×10^{-4}	7.69×10^{-4} – 7.99×10^{-4}
Freshwater Resources				
Total[3]	1.51×10^{-4}	7.60×10^{-4}	1.34×10^{-4}	6.72×10^{-4}

[1] Range based on estimate of reductions in fish tissue concentration contamination (based on projected point source load reductions and the contribution of point sources to total loading).
[2] Total for 11 contaminants listed in Exhibit 8-7.
[3] Total for 8 contaminants listed in Exhibit 8-9.

Exhibit 8-13. Potential Human Health Benefits of Reducing Cancer After Implementation of the CTR to Recreational Anglers[1]

Water Body	Annual Reduction in Cancer Cases	Annual Monetized Benefits (millions of 1998 dollars)[1, 2]
San Francisco Bay	0.04 - 0.05	$0.10 - $0.45
Freshwater Resources	0.44	$1.17 - $4.20

[1] Based on an average consumption rate (21.4 g/day) and a value of a statistical life of $2.5 million to $9.0 million (American Lung Association, 1995). Values based on the estimates of reductions in fish tissue concentration contamination.
[2] Estimates are adjusted from 1995 dollars to first quarter 1998 dollars using the Consumer Price Index (CPI) as reported in the U.S. Department of Labor (U.S. Bureau of Labor Statistics, 1998). Note that there is currently a debate regarding the accuracy of the CPI.

Exhibit 8-14 presents the potential effect of the CTR on systemic risks for recreational anglers, indicating potential reductions in the hazard quotients for PCBs and mercury. For PCBs, EPA expects the hazard quotient associated with the average consumption rate to be reduced from 2.26 to a range of 1.51-1.66 for San Francisco Bay anglers and from 1.40 to 1.01 for freshwater anglers. However, for the high consumption rate (90th percentile), the HQ for PCBs is expected to be reduced from 11.31 to a range of 7.54-8.29 for San Francisco Bay anglers and from 7.02 to 5.04 for freshwater anglers.

EPA estimated that the HQ for mercury will be reduced for both the average and 90th percentile consumption rates, however baseline levels exceed 1.0 for high consumers only. For high fish consumers (90th percentile), EPA expects the HQ for mercury to be reduced from 3.77 to a range of 1.01-1.11 for San Francisco Bay anglers and from 3.12 to 0.90 for freshwater anglers.

[5] Based on the following calculation: (estimated value of a life) x (CPI factor) x (reduction in cancer cases); e.g., $2.5 million x 1.062 x 0.04 = $0.1 million.

Exhibit 8-14. Potential Effect of Implementation of the CTR on Systemic Risks for Recreational Anglers

Contaminant	Baseline Hazard Quotient[1]		Post-CTR Hazard Quotient[1,2]	
	Average Consumption (21.4 g/day)	90th Percentile Consumption (107.1 g/day)	Average Consumption (21.4 g/day)	90th Percentile Consumption (107.1 g/day)
San Francisco Bay				
PCBs	**2.26**	**11.31**	**1.51 - 1.66**	**7.54 - 8.29**
Mercury	0.75	**3.77**	0.20 - 0.22	**1.01 - 1.11**
Dioxin	0.51	**2.54**	0.51 - 0.51	**2.54 - 2.54**
Freshwater Resources				
PCBs	**1.40**	**7.02**	**1.01**	**5.04**
Mercury	0.62	**3.12**	0.18	0.90

[1] Hazard quotients above one shown in bold.
[2] Range based on estimates of reductions in fish tissue concentration contamination.

8.1.6 Uncertainties and Limitations

As described in U.S. EPA (1997), there are numerous uncertainties associated with the assessment of potential human health risks including the following:

> Risks were based on contaminant concentrations found in fish fillets or fish prepared by the most common method for the species (croaker and surf perch fillets with skin, and shark and striped bass without skin). Anglers that consume other body parts or untrimmed fillets (including the skin) face higher risks. The Santa Monica Bay Seafood Consumption Study (MBC Applied Environmental Services, 1994) reported that one-third of all anglers eat fish whole, but gutted, including nearly 50% of Asians and 44% of Hispanics.
>
> Risks were based on tissue contaminant levels measured in raw fish fillets. One study (OEHHA, 1991) found that 4,4-DDT concentrations may decrease by 20% to 80% after cooking (U.S. EPA, 1997).
>
> The assessment does not include potential health risks associated with inorganic arsenic. Arsenic in edible fish tissue is, in almost all cases, present as arsenic-containing organic compounds that are not considered a threat to human health. However, where small amounts of inorganic arsenic are present in edible fish tissue, the analysis will understate potential risks.
>
> Average fish tissue concentrations used in the assessment are calculated using one-half of the MDL for all contaminants reported at below the analytical detection level (but found present in other fish tissue samples taken from the same site).

The risk assessment did not include a separate analysis for low-income anglers. MBC Applied Environmental Services (1994) reported a median fish consumption of 32.1 g/day for anglers with incomes below $5,000, compared to 21.4 g/day for all anglers. Results at the 90th percentile consumption rate included in this analysis covers people consuming higher than average consumption.

Risk reduction based on extrapolation of loadings reduction from sample facilities may overstate or understate actual loadings reduction and actual risk reduction.

The assessment does not account for potential synergistic effects of mixtures of pollutants in fish tissue.

8.2 RECREATIONAL ANGLING BENEFITS

The above section described the potential human health benefits that may result from implementation of the CTR. Concerns regarding adverse health effects from eating contaminated fish also may reduce the value of the recreational fishery because the ability to consume fish may be an important attribute of the overall fishing experience (Knuth and Connelly, 1992; Vena, 1992; FIMS and FAA, 1993; West et al., 1993). This reduction in value may occur because fewer fishing trips are taken or because the value of a trip is reduced. In addition, as described in Chapter 6, reduced toxic contamination may increase stability, resilience, and overall health of numerous ecosystems, which may increase catch rates as well as angling effort in California. Thus, the potential recreational benefits of the CTR may include an increase in the value of fishing experiences and an increase in participation.

This section provides estimates of these two components of recreational angling value. Because the analysis is conducted at the statewide level and does not consider numerous site-specific characteristics that will affect the level of benefits from the rule, the results are only intended to provide a rough approximation of the potential magnitude of recreational benefits. A case study approach would be required to more accurately characterize the anticipated angling benefits at any specific water body in California.

8.2.1 Value of an Improved Fishing Experience

As described previously, toxic contamination is responsible for 12 fish consumption advisories currently in place throughout the state, including advisories for 4,4-DDT, chlordane, dioxin, mercury, PCBs, and selenium (see Exhibit 8-1). These advisories, and knowledge of toxic contamination in other water bodies, may affect anglers' enjoyment of the fishing experience. EPA estimated reductions in mercury and other toxic contaminants in California surface waters as a result of implementation of the CTR. Thus, the rule may reduce concentrations of toxics in fish tissue, increasing value to recreational anglers.

EPA was unable to identify any studies regarding the value to California anglers of reducing toxic contamination of surface waters. However, a 1992 study of the Wisconsin Great Lakes open water sport fishing (Lyke, 1993) does reveal the significance of the contamination problem to the anglers. Lyke estimated the value of the Great Lakes trout and salmon fishing to anglers if it were "completely free of contaminants that may threaten human health." Lyke's estimates indicate benefits of 11% to 31% of the value of the fishery.

Lyke's work estimated the value of reducing toxic contamination in a popular boat fishery that has experienced widespread and highly publicized historical contamination and fish consumption advisories. Thus, the study results may be less applicable for many California anglers because, for example, the fish consumption advisory for San Francisco Bay was issued in 1994 and the fishing experience at many freshwater rivers and streams may differ significantly from Great Lakes trout and salmon angling. However, rather than leave an important category of potential benefits unmonetized, EPA transferred the results from the Lyke study to estimate potential recreational angling benefits of the CTR in California. EPA also considered what the research might indicate about potential benefits for all California waters affected by toxics, not just those waters under fish consumption advisories.[6]

To transfer the Lyke results, EPA first estimated the number of fishing days in California that occur in toxic-impaired waters, distinguishing between water body type (e.g., freshwater river versus saltwater). Next, EPA multiplied the number of fishing days by an average consumer surplus for the different modes of fishing to obtain a baseline value of the fishery. EPA then multiplied by Lyke's estimate of 11% to 31% to obtain the value of a "contaminant-free" fishery. Finally, EPA multiplied by the expected reduction in loadings and the assumed contribution of point sources to total loadings (developed in Chapter 7) to obtain the portion of these benefits that may be potentially attributable to point source controls. These steps are described below.

Estimating Toxic-Impaired Fishing Days

EPA developed estimates of the number of fishing days in freshwater and saltwater sources in California based on information from several sources (National Marine Fisheries Service, 1987–1989 and 1993, Huppert, 1989; U.S. Fish and Wildlife Service, 1993; EPA, 1997). EPA then analyzed the extent of toxic impairment of California waters based primarily on the State of California's WQA database (Water Resources Control Board, 1994) as described in U.S. EPA (1997) and used this information to calculate "toxic-impaired" fishing days. This approach assumes that anglers have not substituted away from contaminated waters.

It also should be noted that EPA defined "impaired" waters as those monitored and rated by the State of California as having medium or poor quality for at least one toxic pollutant or group of

[6] Transferring the Lyke (1993) research to all California waters affected by toxics, but not posing human health risk as indicated by fish consumption advisories, may overstate potential benefits.

toxic pollutants.[7] The State of California has monitored 9% of river and stream miles; 54% of lake and reservoir acreage; and an unknown percentage of bays, estuaries, and saline lakes (U.S. EPA, 1997). Of these monitored waters, the state found that 19% of river and stream miles, 19% of lake and reservoir acreage, 69% of San Francisco Bay, 51% of other California bays, 47% of estuaries, and 69% of saline lakes are "impaired" (U.S. EPA, 1997). EPA assumed for this analysis, maybe conservatively, that California has monitored 50% of bays, estuaries, and saline lakes and then that 50% of unmonitored waters were impaired similarly to monitored waters.[8] To the extent that a substantially greater proportion of waters that have not been monitored are impaired, benefits will be underestimated.

As shown in **Exhibit 8-15**, multiplying the estimated number of fishing days by the percent of monitored waters that are impaired yields estimates of the number of toxic-affected fishing days. EPA estimated a total of 6.4-million fishing days in toxic-impaired waters in California, of which 3.7 million are associated with freshwater fishing and 2.7 million are associated with saltwater fishing.

Exhibit 8-15. Baseline Fishing Days Occurring in Toxic-Impaired Waters[1] in California

	Fishing Days per Year[2]	Percent of Assessed Waters Toxic-Impaired [1,2]	Toxic-Affected Fishing Days[3]
Freshwater Fishing			
Lakes, reservoirs, and ponds	17,826,000	15%	2,673,900
Rivers and streams	10,304,000	10%	1,030,400
Subtotal	28,232,000	—	3,704,300
Saltwater Fishing			
Bays			
San Francisco Bay	750,200[3]	69%	571,638
Other California bays	2,745,800[3]	38%	1,043,404
Estuaries	2,097,600[3]	35%	734,160
Saline lakes	699,200[3]	52%	363,584
Subtotal	6,292,800	—	2,658,786
Total	**34,524,800**	**—**	**6,363,086**

[1] "Impaired" waters are defined as those assessed and rated by the State of California as medium or poor quality for at least one toxic pollutant or group of pollutants. The ratings of these waters corresponds to U.S. EPA's not fully and partially supporting categories.
[2] Based on a total of 6,992,000 total saltwater fishing days. Assumes 50% in bays (e.g., pier fishing), 30% on estuaries, and 10% on saline lakes. Remainder is open sea fishing not addressed by the rule. Estimated fishing days for San Francisco Bay based on estimated number of anglers from health risk analysis (121,000) multiplied by the average days per angler (6.2) from Huppert (1989).
[3] Calculation of toxic-affected fishing days may not be duplicated exactly due to rounding.
Source: Based on U.S. FWS (1993) and U.S. EPA (1997)

Baseline Fishery Value

[7] The California WQA database categories of medium and poor translate to the U.S. EPA categories of not fully supporting and partially supporting. The medium and less severely impaired waters were grouped together into the partially supporting category. The remaining waters classified as poor were placed in the not fully supporting category.

[8] For example, for river and stream miles, the calculation is (19% × 9%) + (19% × 91% × 50%) = 10%.

To estimate the baseline value of the estimated 6.4-million fishing days (in toxic-impaired waters), EPA reviewed the literature for recreational fishing studies that may be appropriate for valuing fishing in California. These studies, listed in **Exhibit 8-16**, suggest consumer surplus associated with freshwater fishing in the range of $25 to $35 per day. This range is consistent with that found by Walsh et al. (1988) in a national review of studies for freshwater fishing. For saltwater fishing, the study results vary more widely, and depend on the mode of fishing (e.g., charter boat, private boat, or shore fishing) and species sought. However, most of the results fall in the range of $50 to $100. This range is also consistent with the average value reported by Walsh et al. for saltwater fishing ($95 per day).

Exhibit 8-16. Studies Revealing Estimates of Consumer Surplus per Fishing Day

Study	Location/Species	Consumer Surplus Estimate ($1996)
Freshwater		
Roach, 1996	American, Feather, Sacramento, and Yuba rivers	$15.24–$36.89; preferred model specification yields $31.17–$36.37 estimate
Hay, 1988	California bass anglers	$31.17
Loomis and Cooper, 1990	Trout in Feather River	$26.69
Walsh, 1988	Average of national studies	$30.85–$40.08
Saltwater		
NOAA, 1986	Marine fishing in Southern California	Charter: $29.74–$66.24 Private: $82.46–$100.02 Shore/pier: $44.23–$84.01
Huppert, 1989	San Francisco Bay, salmon and striped bass	$70.88–$357.36
Walsh, 1988	Average of national studies	$94.89

The ranges of consumer surplus chosen by EPA, $25 to $35 for freshwater and $50 to $100 for saltwater, were adjusted from 1996 to 1998 first quarter dollars using the CPI as reported in the U.S. Department of Labor (U.S. Bureau of Labor Statistics, 1998)[9]. In 1998 dollars, the ranges are $26 to $37 for freshwater and $53 to $105 for saltwater.[10]

Multiplying toxic-impaired fishing days by the relevant range of consumer surplus per day results in estimates of the baseline value of the fishery (See **Exhibit 8-17**). EPA estimated that the baseline value of these waters in California is currently between $237.2 million and $416.2 million per year.

[9] Note that there is currently a debate regarding the accuracy of the CPI.

[10] Based on the following calculations: ($25 × 1.05 = $26); ($35 × 1.05 = $37); ($50 × 1.05 = 53); ($100 × 1.05 = $105).

Exhibit 8-17. Baseline Value of Fishing Days Occurring in Toxic-Impaired Waters in California (1998 First Quarter Dollars)

	Toxic-Affected Fishing Days	Consumer Surplus per Day	Baseline Value ($ millions)
Freshwater Fishing			
Lakes, reservoirs and ponds	2,673,900	$26-$37	$69.5-$98.9
Rivers and streams	1,030,400	$26-$37	$26.8-$38.1
Subtotal	3,704,300		$96.3-$137.1
Saltwater Fishing			
Bays			
San Francisco Bay	517,638	$53-$105	$27.4-$54.4
Other California bays	1,043,404	$53-$105	$55.3-$109.6
Estuaries	734,160	$53-$105	$38.9-$77.1
Saline lakes	363,584	$53-$105	$19.3-$38.2
Subtotal	2,658,786		$140.9-$279.2
Total	**6,363,086**		**$237.2-$416.2**

Potential Benefits Attributable to the CTR

Multiplying the baseline fishery value ($237.2 million to $416.2 million per year) by the increase in value estimated by Lyke (11% to 31%) results in potential benefits of achieving a "toxic-free" fishery of $26.1 million to $129.0 million per year. The next step is to determine the portion of these benefits that might reasonably be attributable to the CTR. EPA believes that the toxicity-weighted results may be the most meaningful for the estimation of benefits although the unweighted results are also important because the pollutants with relatively lower toxic weights can cause problems in a specific waterbody. The toxic-weighted results indicate a 49.6% reduction in pollutant loadings under the low scenario and a 14.7% reduction under the high scenario. As discussed previously, the two scenarios reflect the uncertainties associated with estimation of point source loadings of toxic pollutants. EPA believes that the midpoint between the two scenarios is a reasonable estimate of potential benefits. The midpoint of this range (32.2%) is close to the unweighted reduction in pollutant loadings under the high scenario (28%). Therefore, EPA estimated potential recreational fishing benefits based on a 32.2% reduction in pollutant loadings and assuming that this reduction is indicative of the reduction of impairment from toxics that will be experienced under the CTR.

Thus, EPA multiplied the total potential benefits by 32.2% and then by the percent of total toxic loadings attributable to point sources in California waters, as presented in Exhibit 7-4. As presented in **Exhibit 8-18**, the approach results in potential benefits attributable to the CTR of between $1.53 million per year and $12.99 million per year.

Exhibit 8-18. Potential Recreational Angling Benefits from a "Toxic-Free" Fishery Attributable to Implementation of the CTR (Millions of 1998 First Quater Dollars/Year)

	Baseline Fishery Value	Value of "Toxic-Free" Fishery	Reduction in Toxic-Weighted Loadings Due to the CTR	Assumed Point Source Contribution to Total Loadings	Potential Benefits Attributable to the CTR
Freshwater					
Lakes, reservoirs and ponds	$69.5-$98.9	$7.6-$30.7	32.2%	3%	$0.07 - $0.30
Rivers and streams	$26.8-$38.1	$2.9-$11.8	32.2%	3%	$0.03 - $0.11
Saltwater					
San Francisco Bay	$27.4-$54.4	$3.0-$16.8	32.2%	1%–10%	$0.01 - $0.54
Other bays	$55.3-$109.6	$6.1-$34.0	32.2%	42%–64%	$0.82 - $7.00
Estuaries	$38.9-$77.1	$4.3-$23.9	32.2%	42%–64%	$0.58 - $4.92
Saline lakes	$19.3-$38.2	$2.1-$11.8	32.2%	3%	$0.02 - $0.11
Total	**$237.2-$416.2**	**$26.1-$129.0**	—	—	**$1.53 - $12.99**

8.2.2 Value of Increased Participation

In addition to increasing the value of existing angling days, reduced toxic loadings also may increase participation levels. Toxic contamination may discourage recreational fishing participation because of concern that consumption is unsafe. Similarly, knowledge of toxic contamination alone, regardless of consumption concerns, may reduce anglers' participation at a given site. Improving water quality to achieve toxic water quality criteria may restore this lost participation.

Estimating lost participation, however, is difficult for two reasons. First, little is known about how decreases in participation vary given different levels of contamination. When toxic contamination is not publicized or a fish consumption advisory is not posted, toxic-impaired waters may experience no decrease in fishing since anglers may not change their fishing patterns without knowledge of the contamination. Second, the availability of unaffected substitute sites may simply result in a shift in participation from one site to another. It is difficult, however, to account for substitute sites when estimating benefits for such a large area since the availability of substitute sites may vary greatly depending on geographical location and the economic status of the affected anglers. Participation in unaffected waters may actually decrease if participation shifts to the waters improved by implementation of the CTR. Decreased congestion at unimpaired sites will increase an angler's fishing value, however. EPA was not able to account for the effects of reduced congestion or substitute sites when estimating benefits of increased fishing participation.

Since toxic contamination in California occurs statewide, negative perceptions of California's water quality may also exist statewide. A statewide decrease in the level of toxic contamination on all water bodies may improve perceptions of water quality and thus have a positive impact on participation. In addition, as described in Chapter 6, reduced toxic contamination may increase the stability, resilience, and overall health of numerous ecosystems, resulting in higher catch rates. As a result, even if good substitute sites exist for toxic-affected sites, a minimal increase in participation may result from implementation of the CTR.

A limited number of studies have estimated reductions in participation due to water quality degradation. For example, a survey of New York State anglers (Connelly et al., 1988) found that anglers aware of fish consumption advisories took 17% fewer fishing trips. In a study of lake recreation in Wisconsin, Caulkins et al. (1986) estimated that the number of recreationalists using the site would increase by 12% to 16% as a result of general water quality improvements. Other evidence regarding the behavioral response of anglers to fish consumption advisories suggests that between 10% and 37% of anglers take fewer trips in response to fish consumption advisories (Fiore et al., 1989; Silverman, 1990; Knuth and Connelly, 1992; Knuth et al., 1993; West et al., 1993). All of these studies estimate the percentage of *people* that would take fewer trips, not the percentage decrease in angling days. However, these anglers are not expected to eliminate trip-taking, and, as a result, a 5% to 10% reduction in trips may be reasonably assumed. Because public knowledge of toxic contamination varies across water bodies, EPA conservatively assumed a 5% increase in angler participation in estimating the benefits from increased angling participation for all waters except San Francisco Bay. Since a fish consumption advisory was issued for the Bay in 1994, EPA assumed a 10% increase in angler participation for the Bay.

Potential Benefits Attributable to the CTR

EPA estimated the value of increased angling participation in a similar fashion as it estimated the value of improved fishing experiences. EPA multiplied the number of toxic-affected fishing days by 5% (10% for San Francisco Bay) to estimate the expected increase in participation and valued these days using the estimated consumer surplus values presented in Section 8.2.1. To estimate the portion of these benefits attributable to implementation of the CTR, EPA multiplied by the midpoint of expected reduction in loadings (32.2%) and the attribution assumptions developed in Chapter 7. As shown in **Exhibit 8-19**, potential benefits due to increased participation resulting from the CTR range from \$0.7 million per year to \$2.2 million per year (first quarter 1998 dollars).

Exhibit 8-19. Potential Benefits from Increased Angling Participation (Millions of 1998 First Quarter Dollars/Year)

	Baseline Toxic-Impaired Fishing Days	Additional Fishing Days (5% of Baseline)[1]	Consumer Surplus (per day)	Value of Additional Days	Reduction in Toxic-Weighted Loadings due to the CTR	Assumed Point Source Contribu-tion to Total Loadings	Potential Benefits Attributable to Implementation of the CTR[2]
Freshwater							
Lakes, reservoirs and ponds	2,673,900	133,695	$26–$37	$3.5-$4.9	32.2%	3%	$0.03 - $0.05
Rivers and streams	1,030,400	51,520	$26–$37	$1.3-$1.9	32.2%	3%	$0.01 - $0.02
Subtotal	3,704,300	185,215	$26–$37	$4.8-$6.9	—	—	$0.04 - $0.07
Saltwater							
San Francisco Bay	517,638	51,764	$53–$105	$2.7-$5.4	32.2%	1%–10%	$0.01 - $0.18
Other bays	1,043,404	52,170	$53–$105	$2.8-$5.5	32.2%	42%–64%	$0.37 - $1.13
Estuaries	734,160	36,708	$53–$105	$1.9-$3.9	32.2%	42%–64%	$0.26 - $0.79
Saline lakes	363,584	18,179	$53–$105	$1.0-$1.9	32.2%	3%	$0.01 - $0.02
Subtotal	2,658,786	132,939	$53–$105	$8.4-$16.7	—	—	$0.66 - $2.12
Total	**6,363,086**	**318,154**	—	**$13.2-$23.5**	—	—	**$0.70 - $2.18**

[1] Additional fishing days in San Francisco Bay are estimated to be 10% of the baseline.
[2] Totals may not add up due to rounding.

8.3 NONCONSUMPTIVE WILDLIFE RECREATION VALUES

The 1996 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation (U.S. Department of the Interior, Fish and Wildlife Service, and U.S. Department of Commerce, Bureau of the Census, 1998) indicates that 5.96 million California residents aged 16 or older participated in wildlife watching in 1996. This participation included 17.9 million trips away from home (at least 1 mile) for the primary purpose of observing, photographing, or feeding wildlife. These estimates do not include secondary wildlife-watching activities, such as observing wildlife while pleasure driving (U.S. Department of the Interior, Fish and Wildlife Service, and U.S. Department of Commerce, Bureau of the Census, 1998). Approximately 5.7 million California residents aged 16 or older also participated in wildlife-related activities around the home, including observing, photographing, or feeding wildlife.

Research has shown that nonconsumptive wildlife recreation (viewing wildlife) is highly valued. For example, Rockel and Kealy (1991) estimate a total annual value nationwide of between $8.7 billion and $165 billion in 1980 dollars (with the range of results indicating a sensitivity of their model to functional form). Cooper and Loomis (1991) estimated the total annual value for bird viewing in California's San Joaquin Valley to be $64.7 million (in 1987 dollars), based on willingness to pay (WTP) estimates for all Californians. Cooper and Loomis found that WTP increased as the number of birds seen increased, with diminishing marginal returns evident in their results (Cooper and Loomis, 1991).

As described in the EA that accompanied the proposed CTR, CTR-related improvements in aquatic habitats may lead to healthier and more diverse populations of avian and terrestrial species and may manifest in increased participation and increased user day values for wildlife viewing activities. Without specific information as to the potential magnitude of changes in wildlife populations and thus viewing opportunities that may result from the toxic pollutant loading reductions anticipated under the rule, nonconsumptive wildlife recreation values cannot be estimated. Given the high baseline value, however, these benefits may be appreciable.

8.4 PASSIVE USE (NONUSE) VALUES

As noted in Chapters 5 and 6, individuals may value reduced toxic concentrations in California aquatic environments apart from any values associated with their direct or indirect use of the resource. These passive use (nonuse) values are difficult to estimate in the absence of carefully designed and executed contingent value surveys. "Benefits transfer" techniques, however, can be used to develop a rough approximation of the potential magnitude of these passive use values.

8.4.1 Passive Use Values for Recreational Anglers

Fisher and Raucher (1984) conducted an extensive review of the economics literature providing empirical evidence of the use and nonuse values associated with improved water quality and/or fisheries. Their review indicated that nonuse values are estimated to be *at least* half as great as recreational values. The authors concluded that if passive use values (for example, ecologic values) are applicable to a policy action, using a 50% approximation is preferred, with proper caveats, to omitting passive use values from a benefit-cost analysis.

Several additional research efforts conducted subsequent to the Fisher and Raucher review provide additional support for the observation that omitting passive use values leads, in most cases, to an appreciable underestimate of total benefits. In some instances, such research has been interpreted to suggest that passive use benefits might be as much as (or more than) twice the recreational use values (e.g., Sutherland and Walsh, 1985; Sanders et al., 1990).

To estimate passive use values from estimates of recreational use benefits as described above, it is important to consider the extent to which the primary research efforts have evaluated resources, and changes in resources, that are reasonably comparable to the policy-affected site and the policy-induced environmental impacts. For the CTR, the resources in question are a large share of the water resources throughout California. These waters in general have, at baseline, some degree of toxics-related impairment, and the anticipated change in conditions due to the CTR will reduce the likelihood or severity of impairment in the future.

Generally, it is appropriate to apply the studies reviewed by Fisher and Raucher to the CTR. For example, the Carson and Mitchell study estimates a nationwide value for incremental freshwater quality improvements. Thus, the use of the 0.5 rule of thumb seems appropriate to an application of the CTR.

Studies with ratios of higher passive use to recreational use values may not be as applicable to the CTR. For example, the Sanders et al. results (implying a ratio of approximately 1.8 or 1.9) are based on a study of the value of preserving several free-flowing river segments in Colorado from the development of dams and other major, irreversible hydrological modifications. Given the magnitude and direction of the environmental change evaluated, coupled with the irreversibility of such changes, one would anticipate a relatively higher ratio of existence and bequest values to direct use values than for a rule similar to the CTR.[11]

Based on the available literature and the environmental changes being considered, EPA estimated passive use values for the CTR as one-half of recreational fishing benefits. These estimates are imprecise for several reasons, including the reliance on the benefits transfer technique and the potential that the underlying primary research studies may not themselves be precise or accurate for the environmental applications to which they were directly applied. It also may be the case that this approach underestimates passive use values because the "ecosystem" benefits may not be fully embodied in the contingent valuation studies being applied, or because of potential

[11] The Sanders et al. (1990) study has similar transferability issues. This study shows passive use values that relate to option price (recreational use and option value) with a ratio of 2 or higher, where the scenario is the potential degradation of a relatively pristine resource (Flathead Lake and River) by coal mining. Given the special qualities of the resource being evaluated (high baseline quality, the largest lake in the western United States), and the direction of change being evaluated (potential pollution from coal mining), the passive use values would be expected to be higher relative to use values than would be anticipated in a CTR context (moderate improvements in water quality in a wide variety of already impaired waters).

underestimation of the applicable recreational use values (if recreational benefits are overstated, then the reverse may be true).

In addition, because some primary studies suggest passive use values may exceed one-half of recreational values, and because recreational fishing values alone are used in lieu of total potential recreational values, the use of the 0.5 ratio is conservative. Furthermore, the primary studies reviewed generally are based on separating the respondent's (household's) total willingness to pay into the two components—passive use value and recreational use value. The 0.5 ratio therefore reflects the amount of passive use value that recreational angling households are willing to pay, above their recreational use values, to preserve or enhance water quality. This rule of thumb suggests that the potential magnitude of passive use values associated with implementation of the CTR for users ranges from $1.1 million per year to $7.6 million per year.

Applying the 50% rule of thumb to the CTR is, in essence, providing a rough estimate of passive use values only for those households that have active recreational anglers. Therefore, this estimate likely provides a very conservative lower bound; it implies that only recreational anglers have passive use values. As described below, EPA developed preliminary estimates of passive use values for non-angling households.

8.4.2 Passive Use Values for Non-Angling Households

To account for the passive use values held by non-angling households, which includes other water recreators such as boaters, swimmers, and nonusers, EPA assumed that the number of angling households is equivalent to the number of licensed anglers in the State of California. EPA then subtracted the number of angling households from all households in California to obtain the number of non-angling households. Because it is likely that there is more than one angler in some households, this assumption is conservative in that it will result in a lower estimate of non-angling households and values.

As an upper-bound estimate of passive use values for non-angling households, EPA assumed that these households have a passive use value equal to that of angling households. As a lower-bound estimate, EPA assumed that all non-angling households are nonuser households, and that they hold lower passive use values than angling households. EPA did not find any literature that provides an indication of how much lower these values might be. Some studies, however, provide information on the relationship between total WTP for water quality improvements for users and nonusers.

WTP Values for Users and Nonusers

EPA found several contingent valuation studies that estimated WTP for users and nonusers of water resources (See **Exhibit 8-20**); however, most of these studies have little relevance to the CTR. Brown and Duffield (1995) estimated WTP to protect the instream flow of a single river and a group of five rivers. Olsen et al. (1991) estimated WTP to double the size of salmon and steelhead runs in the Columbia River Basin. Croke et al. (1986–1987) estimated the WTP to improve water impaired by sewer overflows in Chicago to a level acceptable for outings, boating, and fishing. While these studies show how WTP compares for users and nonusers, they do not evaluate water quality controls or improvements similar to those anticipated for the CTR.

Additionally, Bockstael et al. (1989) report WTP to raise Chesapeake Bay water quality so that it is acceptable for swimming. This study evaluated WTP for clean-up efforts devoted to reducing toxic substances, but it also addresses nutrient over-enrichment and the decline of submerged vegetation. Mean WTP to make the bay acceptable for swimming was $38 for nonusers, which is approximately 31% of the value for users ($121). Since WTP for users includes a use value, 31% likely understates the relationship between passive use values between users and nonusers

Exhibit 8-20. Relationship Between Willingness to Pay Values for Users and Nonusers[1]

Study	WTP for Improvement		Ratio of WTP (nonusers to users)
	Users	Nonusers	
Brown and Duffield (1995)			
One river	$10.18	$3.55	35%
Five rivers	$18.02	$2.02	11%
Loomis et al. (1991)[2]			
Salmon improvement	$202	$181	90%
Contamination reduction	$360	$308	86%
Wetland improvement	$286	$251	88%
Olsen et al. (1991)	$6.18	$2.21	36%
Bockstael et al. (1989)	$121	$38	31%
Croke et al. (1986–1987)	$49.63	$45.76	92%

[1] Year of dollars for the WTP values are not reported since only the ratio between nonuser and user values are compared as opposed to the values themselves.
[2] WTP for users reflects survey responses of local households. WTP for nonusers reflects survey responses of the rest of California's households.

Loomis et al. (1991) may be the most applicable study. Loomis et al. (1991) estimated the benefits to California residents near a resource (users) and nonusers of improved fishery, wetland, and waterfowl resources in the San Joaquin Valley. They used a contingent value survey to determine California households' WTP to implement three wildlife programs: wetlands habitat and wildlife maintenance and improvement, wildlife contamination control, and San Joaquin River and salmon improvement. Mean WTP to improve salmon populations was $202 for users and $181 for nonusers. Mean WTP for contamination reduction was $360 for users and $308 for nonusers. Mean WTP for wetland improvement was $286 for users $251 for nonusers.

Thus, Loomis et al. (1991) provides California-specific research suggesting that nonusers' WTP may be 85% to 90% of the WTP for users.

Lower-Bound Estimate

Due to the nature of the impairment addressed by the CTR, it is likely that improvements may be more valued by users than nonusers, who may even be unaware of the contamination. Thus, as a lower-bound estimate, EPA assumed that passive use values for non-angling households may be 30% of those for angling households. This estimate is supported by Bockstael et al. (1989) who evaluated WTP for clean-up efforts devoted to reducing toxic pollutants to improve water quality in the Chesapeake Bay.

To estimate the number of non-angling households, EPA assumed that the number of recreational angling households is equivalent to the number of licensed recreational anglers, or approximately 1.4 million (California Department of Fish and Game, 1996). Subtracting this from the total number of households in California (approximately 11.1 million; U.S. Bureau of the Census, 1995) yields approximately 9.7 million non-angling households. Assuming a passive value that is 30% of the passive value for angling households yields a range of benefits from $2.3 million to $15.8 million per year for all non-angling households.[12]

Upper-Bound Estimate

As an upper bound, EPA assumed that passive use values for non-angling households may be 90% of those for angling households. This estimate is supported by Loomis et al. (1991), who specifically surveyed California residents near a resource (users) and statewide (nonusers). Multiplying the per household annual value for angling households, the number of non-angling households in California (9.7 million), and the assumed 90% passive use value for nonusers, results in a range of benefits from $7.0 million to $47.3 million per year.[13]

The overall passive use benefits for non-angling households range from $2.3 million per year to $47.3 million per year.

[12] EPA calculated a per household value for angling households of $0.80–$5.42 per year by dividing the passive use value estimated in the previous section ($1.1–$7.6 million per year) by the estimated number of angling households (1.4 million). Thirty percent of this value yields $0.24–$1.63 per household for non-angling households.

[13] The per household value for non-angling households is $0.72 - $4.88 in this case.

8.5 TOTAL VALUE OF SIMILAR IMPROVEMENTS

An alternative to estimating the individual categories of benefits resulting from water quality improvements is to estimate a total value for all types of benefits. Carson et al. (1994) and Loomis et al. (1991) provide total values for toxic-related water quality improvements based on contingent valuation research for California households. Carson et al. (1994) estimated the total value of a program to reduce the recovery time for four species in the Southern California Bight (bald eagles, peregrine falcons, white croaker, and kelp bass) that have been adversely affected by 4,4-DDT and PCBs. The authors estimated a household WTP of $55.61 in a one-time payment for the reduced recovery time. Loomis et al. (1991) estimated annual California household WTP of $254 in the form of higher taxes for wetland habitat improvement, $313 for water contamination reduction, and $183 for salmon fishery improvement (the study notes that the benefit of all three aspects would be somewhat less than the sum of the individually estimated benefits). Given the differences between the programs valued in these studies and the CTR, benefits transfer of a per household value for the CTR would be difficult. However, the studies illustrate WTP for the toxic-related water quality improvements anticipated from the CTR.

8.6 SUMMARY OF MONETIZED BENEFITS

A summary of the estimated monetized benefits from implementation of the CTR is provided in **Exhibit 8-21**. Human health benefits are estimated for San Francisco Bay and statewide freshwater resources; all other benefits are estimated statewide.

Exhibit 8-21. Summary of Annual Benefits from Implementation of the CTR (Millions of 1998 First Quarter Dollars)

Benefit Category	Annual Value
Human Health (cancer risk)	
San Francisco Bay	$0.1 - $0.4
Other saltwater resources	+
Freshwater resources	$1.17 - $4.20
Recreational Angling	
Increased value of existing trips	$1.53 - $12.99
Increased participation	$0.70 - $2.18
Wildlife Viewing	+
Passive Use	
Households with recreational anglers	$1.12 - $7.59
Other households	$2.32 - $47.31
Omitted Benefits[1]	+
Total	**$6.94 - $74.71**

[1] Benefits not monetized include noncancer human health effects, water-related recreation apart from fishing, and consumptive and nonconsumptive land-based recreation.
+: Positive benefits expected but not monetized.

The key omissions, biases, and uncertainties associated with the benefits analysis are shown in **Exhibit 8-22**. It was difficult to assess the overall impact of the omissions, biases, and uncertainties on the benefits estimates because the degree to which they might cause the estimates to be underestimated or overestimated cannot be predicted with accuracy. Among the key factors described in Exhibit 8-22, however, the omission of potential benefit categories may have the most significant impact and would contribute to an underestimate of benefits. Several categories of potential or likely benefits were omitted from the quantified and monetized estimates (e.g., see U.S. EPA, 1997). In terms of potential magnitudes of benefits, the following are likely to be the most significant contributors to the underestimation of the monetized values presented in Exhibit 8-21:

> Improvements in water-related (in-stream and near-stream) recreation apart from fishing. The omission of boating, swimming, picnicking, and related in-stream and stream-side recreational activities from the benefits estimates could contribute to an appreciable underestimation of total benefits. Such recreational activities have been shown in empirical research to be highly valued, and even modest changes in participation and or user values could lead to sizable benefits statewide. Some of these activities can be closely associated with water quality attributes, particularly swimming. Other recreational activities may be less directly related to the CTR-induced water quality improvements, but might nonetheless increase due to their association with fishing, swimming, or other activities in which the participants might engage.

> Improvements in consumptive and nonconsumptive land-based recreation, such as hunting and wildlife viewing. CTR-related improvements in aquatic habitats may lead (via food chain and related ecologic benefit mechanisms) to healthier, larger, and more diverse populations of avian and terrestrial species, such as waterfowl, eagles, and otters. Improvements in the populations for these species could manifest as improved hunting and wildlife viewing opportunities, which might in turn increase participation and user day values for such activities. Although the scope of the benefits analysis has not allowed a quantitative assessment of these values at either baseline or post CTR conditions, these benefits may be appreciable.

Exhibit 8-22. Key Omissions, Biases, and Uncertainties in the Benefits Analysis for the CTR

Omissions/Biases/Uncertainties	Direction of Impact on Benefit/Cost Estimates	Comments
The monetized estimate of benefits omits some categories (e.g., noncancer human health effects, water-related recreation apart from fishing, and consumptive and nonconsumptive land-based recreation).	(-) The omission of potential benefit categories will cause benefits to be underestimated.	The potential magnitude of these benefits may be appreciable.
Human health benefits for saltwater anglers were estimated for San Francisco Bay only.	(-) The omission of other saltwaters may cause benefits to be underestimated.	The number of anglers fishing in other bays, estuaries, and saltwater lakes is estimated to be over 0.5 million anglers (U.S. EPA, 1997).
Human health exposure was calculated based on the assumption that each fish contained all contaminants of concern at the concentrations reported in the fish tissue data.	(+) To the extent that not all fish contain all contaminants at the assumed concentrations, benefits may be overestimated.	The uncertainties in estimating fish tissue concentrations are inherent in the approach used to estimate human health benefits.
Human health risks were based on contaminant concentrations in fish fillets or, for some species, fish fillets with skin.	(-) The use of fish fillets will underestimate risks to anglers that consume other body parts or untrimmed fillets.	The Santa Monica Bay Seafood Consumption Study (MBC Applied Environmental Services, 1994) reported that one-third of all anglers eat fish whole (but gutted), including nearly 50% of Asians and 44% of Hispanics.
Human health risks were based on contaminant concentrations in raw fish fillets.	(+) The use of raw fish fillets may overestimate benefits.	OEHHA (1991) noted that DDT concentrations decreased by 20% to 80% after cooking.
Toxic-impaired waters were defined as waters rated as medium or poor quality for at least one toxic pollutant or group of pollutants. The rating of these waters corresponds to U.S. EPA's not fully and partially supporting categories.	(+) The inclusion of medium-rated waters may result in an overestimate of toxic-impaired waters.	Toxic-impaired waters provide the basis for estimating toxic-impaired fishing days and thus recreational angling and passive use benefits.
Estimation of the increased value of current angling and increased participation in recreational angling assumes that anglers have not substituted away from contaminated waters.	(+) The assumption that anglers have not substituted away from contaminated waters is likely to cause benefits to be overestimated.	It is likely that some anglers have substituted away from contaminated waters.
Overall Impact on Benefits Estimates	(?)	The overall impact on benefits is uncertain because the degree to which the omissions, biases, and uncertainties might cause the estimates to be underestimated or overestimated is unknown.

+: Potential overestimate.
−: Potential underestimate.
?: Uncertain impact.

9.0 COMPARISON OF POTENTIAL BENEFITS TO COSTS

This chapter compares the potential benefits and costs attributable to implementation of the CTR. EPA compared these estimates using two approaches: (1) a direct comparison of annualized costs to benefits, and (2) a comparison of discounted benefits and costs.

9.1 COMPARISON OF ANNUALIZED BENEFITS AND COSTS

A direct comparison of the monetized annual (steady-state) benefits of the CTR and annualized costs shows benefits and costs to be generally commensurate given the uncertainty in the analysis and that several categories of benefits are unmonetized. As shown in **Exhibit 9-1**, the estimate of monetized benefits ranges from $6.9 million per year to $74.7 million per year. Annualized costs are $33.5 million under the low scenario and $61.0 million under the high scenario.

Exhibit 9-1. Comparison of Annual Potential Benefits and Costs of Implementing the CTR (Millions of 1998 First Quarter Dollars)

Comparison Method	Monetized Benefits	Annualized Costs	
		Low Scenario	High Scenario
Direct Annual Comparison[1]	$6.9 - $74.7	$33.5	$61.0

[1] These monetized costs and benefits are not directly comparable since several categories of benefits have not been monetized.

9.2 COMPARISON OF DISCOUNTED BENEFITS AND COSTS

Because the benefits and costs associated with implementation of the CTR may be characterized by an initial outlay of capital costs and a gradual phase-in of benefits, **Exhibit 9-2** presents a present value of benefits and costs over 30 years. This method applies a present value social accounting in which the streams of future benefits and costs are discounted to their present values to reflect society's rate of time preference. EPA considered two different phase-in scenarios to account for the potential delay in realizing benefits since many of the pollutants addressed by the CTR are persistent in the environment. To the extent that benefits of reducing toxic pollutants under the CTR are realized sooner, these scenarios may result in an underestimate of the present value of benefits. EPA assumed that there is a 7% opportunity cost of capital and that capital is replaced every 10 years. Since the life of capital typically exceeds 10 years, this assumption may result in an overestimate of costs. EPA calculated the present value of the streams of benefits and costs using discount rates of 3% and 7%.

As shown in Exhibit 9-2, discounted costs fall within the range of discounted benefits under the low scenario, but discounted costs exceed discounted benefits in three of the four cases shown for the high scenario. However, the assumption that capital is replaced every 10 years likely overstates costs. At the same time, benefits may be understated because some categories are not monetized and full benefits may be realized sooner than 10 or 20 years. Thus, EPA expects that the present value of benefits and costs is more commensurate than shown.

Exhibit 9-2. Comparison of Discounted Benefits and Costs of Implementing the CTR (Millions of 1998 First Quarter Dollars)[1]

Schedule of Benefits	Benefits[2]	Costs[3]	
		Low	High
3% Discount Rate			
10-Year Phase-In of Benefits	$108 - $1166	$617	$1033
20-Year Phase-In of Benefits	$82 - $883	$617	$1033
7% Discount Rate			
10-Year Phase-In of Benefits	$63 - $683	$421	$767
20-Year Phase-In of Benefits	$45 - $480	$421	$767

[1] Present values over 30 years.
[2] Benefits are phased in proportionately over 10 and 20 years, and have their full value in the remaining years. Benefits are not directly comparable to costs since several categories of benefits have not been monetized.
[3] Reflects capital costs plus a 7% cost of capital in years 1, 11, and 21, operating and maintenance costs in years 2 through 30.

9.3 CONCLUSIONS

Comparison of annual values of benefits and costs resulting from implementation of the CTR shows estimated costs falling within the range of monetized benefits. Comparison of 30-year present values of benefits and costs also shows costs under the low scenario to fall within the range of monetized benefits although costs under the high scenario generally fall just outside this range. However, EPA believes that benefits may actually be higher than shown because some categories of potential benefits have not been quantified or monetized. EPA was not able to quantify or monetize potential improvements in water-related recreation apart from fishing, such as boating, swimming, picnicking, and related in-stream and stream-side recreational activities. EPA was also unable to quantify or monetize potential improvements in wildlife viewing. Research indicates that wildlife viewing is a highly valued activity and that California residents value reductions in toxic pollutants that may affect wildlife resources. Thus, these omissions may result in an underestimate of benefits. In addition, using a capital life of 10 years likely overestimates potential compliance costs.

10.0 REFERENCES

American Lung Association. 1995. Dollars and Cents: The Economic and Health Benefits of Potential Particulate Matter Reductions in the United States. Prepared by L.G. Chestnut, Hagler Bailly Consulting, Inc. Boulder, CO.

Barron, M.G., and K.B. Woodburn. 1995. Ecotoxicology of Chlorpyrifos. *Rev. Environ. Contamin. Toxicol. Chem* 144:1–93.

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

Brock, T.C.M., M. van den Bogaert, A.R. Bos, S.W.F. van Breukelen, R. Reiche, R.M.M. Roijackers, R.E.M. Suykerbuyk, and Terwoert. 1992. Fate and Effects of the Insecticide Dursban 4E in Indoor *Elodea*-dominated and Macrophyte-free Freshwater Model Ecosystems: II. Secondary Effects on Community Structure. *Arch Environ Contamin Toxicol* 23:391–409.

Brown,T.C., and J.W. Duffield. 1995. Testing Part-Whole Valuation Effects in Contingent Valuation of Instream Flow Protection. *Water Resources Research* 31 (9):2341–51.

CDFG (California Department of Fish and Game, Inland Fisheries Division). 1995. Sacramento River System Sport Fish Catch Inventory, Final Performance Report.

California Regional Water Quality Control Board. 1997. Santa Monica Bay: State of the Watershed.

California State Water Resources Control Board. 1995. California's Rivers and Streams: Working Toward Solutions (for California's Rivers and Streams).

California State Water Resources Control Board. 1996. California 305(b) Report on Water Quality.

Carson, R.T., et al. 1994. Prospective Interim Lost Use Value Due to DDT and PCB Contamination in the Southern California Bight. Report prepared by NRDA, Inc., and Industrial Economics, Inc. to NOAA. La Jolla, CA.

Caulkins, P.P., R. Bishop, and N. Bouwes. 1986. The Travel Cost Model for Lake Recreation: A Comparison of Two Methods for Incorporating Site Quality and Substitution Effects. *American Journal of Agricultural Economics.*

City of San Jose. 1994. Industrial Mass Audit Studies Summary Report.

Connelly, N.A., T. Brown, and B. Knuth. 1988. New York Statewide Angler Survey. Prepared for the New York State Department of Environmental Conservation, Bureau of Fisheries.

Cooper, J and J. Loomis. 1991. Economic Value of Wildlife Resources in the San Joaquin Valley: Hunting and Viewing Values. *Economics and Management of Water and Drainage in Agriculture*.

Croke, K., R. Fabian, and G. Brenniman. 1986–1987. Estimating the Value of Improved Water Quality in and Urban River Systems. *Journal of Environmental Systems* 16(1):13–23.

Davis, J.A., A.J. Gunther, B.J. Richardson, and J.M. O'Connor. 1991. Status and Trends Report on Pollutants in the San Francisco Estuary. Prepared by the San Francisco Bay – Delta Aquatic Habitat Institute for the San Francisco Estuary Project. U.S. EPA, San Francisco, CA.

Eaton, J., J. Arthur, R. Hermanutz, R. Kiefer, L. Mueller, R. Anderson, R. Erickson, B. Nordling, J. Rogers, and H. Pritchard. 1985. Biological effects of continuous and intermittent dosing of outdoor experimental streams with chlorpyrifos. *Aquatic Toxicology and Hazard Assessment: 8th Symposium*. Bahner, R.C., Hansen, D.J., eds. ASTM STP 891. Philadelphia, PA.

Eisenberg, Oliveri & Associates, Inc. (EOA). 1994. City of Sunnyvale Summary of Pollution Prevention Studies. Submitted in fulfillment of CDO 93-086, Provisions IIA and IIB.

Eisler, R. 1985a. Cadmium Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. Prepared by U.S. Fish and Wildlife Service, Patuxent Wildlife Research Center, Laurel, MD. Sponsored by U.S. Department of the Interior. *Biological Report* 85(1.2).

Eisler, R. 1985b. Selenium Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. Contaminant Hazard Review Report #5. USFWS Patuxent Wildlife Research Center, Laurel, MD.

Eisler, R. 1986a. Chromium Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. U.S. DOI. Contaminant Hazard Reviews Report No. 6. *Biological Report* 85(1.6).

Eisler, R. 1986b. Dioxin Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. Prepared by U.S. Fish and Wildlife Service, Patuxent Wildlife Research Center, Laurel, MD. Sponsored by U.S. Department of the Interior. *Biological Report* 85(1.8).

Eisler, R. 1986c. Polychlorinated Biphenyl Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. *U.S. Fish Wildl. Serv. Biol. Rep.* 85(1.7).

Eisler, R. 1987a. Mercury Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. Prepared by U.S. Fish and Wildlife Service, Patuxent Wildlife Research Center, Laurel, MD. Sponsored by U.S. Department of the Interior. *Biological Report* 85(1.10).

Eisler, R. 1987b. Polycyclic Aromatic Hydrocarbon Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. Prepared by U.S. Fish and Wildlife Service, Patuxent Wildlife Research Center, Laurel, MD. Sponsored by U.S. Department of the Interior. *Biological Report* 85(1.11).

Eisler, R. 1988a. Arsenic Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. Prepared by U.S. Fish and Wildlife Service, Patuxent Wildlife Research Center, Laurel, MD. Sponsored by U.S. Department of the Interior. *Biological Report* 85(1.12).

Eisler, R. 1988b. Lead Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. Prepared by U.S. Fish and Wildlife Service, Patuxent Wildlife Research Center, Laurel, MD. Sponsored by U.S. Department of the Interior. *Biological Report* 85(1.14).

Eisler, R. 1993. Zinc Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. Prepared by the U.S. Department of Interior. Contaminant Hazard Reviews Report 26. *Biological Report* 10.

FIMS and FAA (Fishery Information Management Systems, Inc. and Department of Fisheries and Allied Aquaculture). 1993. Estimation of Daily Per Capita Freshwater Fish Consumption of Alabama Anglers. Prepared for Alabama Department of Environmental Management. Montgomery, AL.

Fiore, B.J., H.A. Anderson, L.P. Hanrahan, L.J. Olson, and W.C. Sonzogni. 1989. Sport Fish Consumption and Body Burden Levels of Chlorinated Hydrocarbons: A Study of Wisconsin Anglers. *Archives of Environmental Health* 44(2):82–88.

Fisher, A., and R. Raucher. 1984. Intrinsic Benefits of Improved Water Quality. *Advances in Applied Micro-Economics* 3:37–66.

Goyer, R.A. 1991. Toxic Effects of Metals. In *Casarett and Doull's Toxicology, The Basic Science of Poisons 4th ed*. Ambur, M.O., J.Doull, and C.D. Klaassen, eds. New York: Pergamon Press.

Gunther, A.J., J.A. Davis, and D.J.H. Phillips. 1987. An Assessment of the Loading of Toxic Contaminants to the San Francisco Bay-Delta. Prepared by Aquatic Habitat Institute.

Hay, M.J. 1988. Net Economic Values of Nonconsumptive Wildlife-Related Recreation. Division of Policy and Directives Management, U.S. Fish and Wildlife Service, Washington, D.C.

Hoffman, D.J., B.A. Rathner, G.A. Benton, Jr., and J. Cairns, Jr. 1995. *Handbook of Ecotoxicology*. Boca Raton, FL: Lewis Publishers.

Huppert, D.D. 1989. Measuring the Value of Fish to Anglers: Application to Central California Anadramous Species. *Marine Resource Economics* 6:89–107.

Kersting, K. 1994. Functional Endpoints in Fielding Testing. *Freshwater Field Tests for Hazard Assessment of Chemicals*. I.A. Hill, F. Heimbach, P. Leewangh, and Matthiesen eds. Ann Arbor, MI: Lewis Publishers.

Knuth, B.A., and N.A. Connelly. 1992. Is New York's Health Advisory on Fish Consumption Making a Difference? *Coastlines* 22(4):4–5.

Knuth, B.A., N.A. Connelly, and M.A. Shapiro. 1993. Angler Attitudes and Behavior Associated with Ohio River Health Advisories. Prepared by the Human Dimensions Behavior Research Unit of the Department of Natural Resources of the New York State College of Agriculture and Life Sciences. HDRU Series No. 93-6.

Loomis, J., and J. Cooper. 1990. Economic Benefits of Instream Flow to Fisheries: A Case Study of California's Feather River. *Rivers* ___ 23–30.

Loomis, J, M. Haneman, B. Kanninen, and T. Wegge. 1991. Willingness to Pay to Protect Wetlands and Reduce Wildlife Contamination for Agricultural Drainage. *Economics and Management of Water and Drainage in Agriculture.*

Lyke, A.J. 1993. Discrete Choice Models to Value Changes in Environmental Quality: A Great Lakes Case Study. Dissertation, 376. University of Wisconsin-Madison.

MBC Applied Environmental Services. 1994. Santa Monica Bay Fish Consumption Study – September 1991 to August 1992. Prepared for Santa Monica Bay Restoration Project.

National Marine Fisheries Service. 1987–1989 and 1993. National Marine Recreational Fishery Statistics Survey, Pacific Coast. As cited in Chapter 3 of U.S. EPA (1997).

NOAA (National Oceanic and Atmospheric Administration). 1988a. The National Coastal Pollutant Discharge Inventory: Estimates for San Francisco Bay. Data Summary, Ocean Assessments Division, Office of Oceanography and Marine Assessment National Ocean Service.

NOAA. 1988b. The National Coastal Pollutant Discharge Inventory: Estimates for Santa Monica Bay, San Pedro Bay, and San Diego Bay. Data Summary, Ocean Assessments Division, Office of Oceanography and Marine Assessment National Ocean Service.

NOAA. 1988c. The National Coastal Pollutant Discharge Inventory: Estimates for Humboldt Bay and Monterey Bay. Data Summary, Ocean Assessments Division, Office of Oceanography and Marine Assessment National Ocean Service.

NOAA. 1986. As cited in Chapter 7 of U.S. EPA (1997).

OEHHA (Office of Environmental Health Hazard Assessment). 1991. A Study of Chemical Contamination of Marine Fish from Southern California, Volume II: Comprehensive Study.

Olsen, D., J. Richards, and R.D. Scott. 1991. Existence and Sport Values for Doubling the Size of Columbia River Basin Salmon and Steelhead Runs. *Rivers* 2(1):44–56.

Rand, G.M., and S.R. Petrocelli. 1985. *Fundamentals of Aquatic Toxicology: Methods and Applications*. Washington, D.C: Hemisphere Publishing Corporation.

Roach, B. 1996. Angler Benefits Along Four California Rivers: An Application of Tobit Analysis.

Rockel, M.L. and M.J. Kealy. 1991. The Value of Nonconsumptive Wildlife Recreation in the United States. *Land Economics* 67(4):422-34.

SAIC (Science Applications International Corporation). 1993. Assessment of Compliance Costs Resulting From Implementation of the Proposed Great Lakes Water Quality Guidance. Final Report. Prepared for U.S. Environmental Protection Agency.

SAIC. 1997. Analysis of Potential Costs Related to the Implementation of the California Toxics Rule. Final Draft. Prepared for the U.S. Environmental Protection Agency.

Sanders, L.D., R.G. Walsh, and J.B. Loomis. 1990. Toward Empirical Estimation of the Total Value of Protecting Rivers. *Water Resources Research* 26(7):1345–57.

SFRWQCB (San Francisco Regional Water Quality Control Board). 1994. Contaminant Levels in Fish Tissue from San Francisco Bay, Final Draft Report. San Francisco, CA.

Shacklette, H.T. and J.G. Beorngen. 1984. Element Concentrations in Soils and Other Surficial Materials of the Conterminous United States. U.S. Geological Survey Professional Paper 1270.

Silverman, W.M. 1990. Michigan's Sport Fish Consumption Advisory: A Study in Risk Communication. A thesis submitted to the University of Michigan.

Smith, A.G. 1991. Chlorinated Hydrocarbon Insecticides. In *Handbook of Pesticide Toxicology, Volume 2 Classes of Pesticides*. W.J. Hayers, Jr. and E.R. Laws, Jr. eds. Academic Press, Inc.

Sutherland, R.J. and R.G. Walsh. 1985. Effect of Distance on the Preservation Value of Water Quality. *Land Economics* 61(3):282–91.

U.S. Bureau of Census. 1980. As cited in Chapter 3 of U.S. EPA (1997).

U.S. Bureau of Census. 1995. *1995 Statistical Abstract.* U.S. Department of Commerce, Economics and Statistics Administration.

U.S. Bureau of the Census. 1996. Estimates of Housing Units, Households, Households by Age of Householder, and Persons per Household. ST-96-20R.

U.S. Bureau of Labor Statistics. 1998. U.S. Department of Labor, All Urban Consumers, U.S. City Average. [ftp//ftp.bls.gov].

U.S. EPA (U.S. Environmental Protection Agency). 1980a. Ambient Water Quality Criteria for Silver. EPA 440/5-80-071. United States Environmental Protection Agency. Office of Water, Washington, D.C.

U.S. EPA. 1980b. Ambient Water Quality Criteria for Nickel. EPA 440/5-80-060. Office of Water Regulations and Standards Criteria, Criteria and Standards Division, Washington, D.C.

U.S. EPA. 1982. As cited in Chapter 3 of U.S. EPA (1997).

U.S. EPA. 1985. Ambient Water Quality Criteria for Copper – 1984. EPA 440/5-84-031. Office of Water Regulations and Standards Criteria, Criteria and Standards Division, Washington, D.C.

U.S. EPA. 1989a. Assessing Human Health Risks from Chemically Contaminated Fish and Shellfish: A Guidance Manual. EPA 503/8-89-002. Office of Marine and Estuarine Protection, Washington, D.C.

U.S. EPA. 1989b. Risk Assessment Guidance for Superfund, Volume I: Human Health Evaluation Manual. Office of Emergency and Remedial Response, Washington, D.C.

U.S. EPA. 1991. Technical Support Document for Water Quality-based Toxics Control. EPA 505/2-90-001. Office of Water, Washington, D.C.

U.S. EPA. 1992. National Study of Chemical Residues in Fish, Volumes I and II. EPA 832-R-92-008a,b. Office of Science and Technology, Washington, D.C.

U.S. EPA. 1993. Guidance for Assessing Chemical Contaminant Data for Use in Fish Advisories: Volume 1, Fish Sampling and Analysis. EPA 823-R-93-002. Office of Water, Washington, D.C.

U.S. EPA. 1994. Water Quality Standards Handbook: Second Edition (Update #1). Office of Water, Washington, D.C.

U.S. EPA. 1996. The Metals Translator: Guidance for Calculation of a Total Recoverable Permit Limit From a Dissolved Criteria. EPA 823-B-96-007. Office of Water, Washington, D.C.

U.S. EPA. 1997. Analysis of the Potential Benefits Related to Implementation of the California Toxics Rule. Office of Sustainable Ecosystems and Communities, Office of Policy, Planning and Evaluation and EPA Region 9.

U.S. Department of the Interior, Fish and Wildlife Service and U.S. Department of Commerce, Bureau of the Census. 1998. 1996 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation.

Vena, J.E. 1992. Risk Perception, Reproductive Health Risk and Consumption of Contaminated Fish in a Cohort of New York State Anglers. Research Program in Occupational and Environmental Health, State University of New York at Buffalo.

Verschueren, K. 1983. *Handbook of Environmental Data on Organic Chemicals 2nd. ed*. New York: Van Nostrand Reinhold.

Walsh, R., D.M. Johnson, and J.R. McKean. 1988. Review of Outdoor Recreation Economic Demand Studies with Nonmarket Benefit Estimates, 1968–1988.

Water Resources Control Board. 1988. Nonpoint Source Management Plan.

Water Resources Control Board. 1991. Toxic Substances Monitoring Program, *Data Report (91-1WQ*). California Environmental Protection Agency.

Water Resources Control Board. 1994. Water Quality Assessment. California Environmental Protection Agency.

West, P.C., J.M. Fly, R. Marans, and D. Rosenblatt. 1993. Minorities and Toxic Fish Consumption: Implications for Point Discharge Policy in Michigan.

APPENDIX A. ALTERNATIVES ANALYSIS

In conducting an analysis of the potential costs to point source dischargers as a result of implementing the CTR, EPA made a variety of assumptions. To test the impact of some of these assumptions on the analysis, EPA conducted two alternative analyses. First, EPA estimated the impact on costs associated with changing the human health risk level for carcinogenic pollutants. Second, EPA estimated the impact on costs associated with changing the application of criteria for heavy metals. Sections A.1 and A.2 present the methodology and costs associated with varying the human carcinogenic risks and applying toxic metals criteria in total recoverable form, respectively.

A.1 IMPACT OF HUMAN HEALTH RISK LEVEL

According to EPA's *Water Quality Standards Handbook:* Second Edition (U.S. EPA, 1994), EPA generally regulates carcinogenic toxic pollutants based on a range of assumed risk levels. This range is established based on 1 excess cancer case per 10,000 people (10^{-4}), 1 excess cancer case per 100,000 people (10^{-5}), and 1 excess cancer case per 1,000,000 people (10^{-6}). However, EPA does not recommend a particular risk level as policy.

The State of California historically has protected at a 10^{-6} risk level for carcinogenic pollutants. The CTR follows this history and establishes human health criteria for carcinogens based on a 10^{-6} risk level. The potential costs discussed in Chapter 4 of this report are based on these criteria.

In its readoption of its statewide plans for inland surface waters and enclosed bays and estuaries, however, California may consider other risk levels for carcinogenic pollutants. Again, EPA recommends that states consider minimum risk levels in the range of 10^{-4} to 10^{-6} for carcinogenic priority toxic pollutants to protect public health and welfare. Many states base their human health protection criteria on a 10^{-5} risk level.

The purpose of this analysis is to determine the change in potential costs should the CTR criteria for human health protection from carcinogens be based on a 10^{-5} risk level.

A.1.1 Methodology

EPA used the same methods described in Chapter 4 of this report to derive potential costs related to the use of a lower risk level for carcinogens. The only modification to the methodology is that EPA adjusted the proposed CTR criteria for carcinogens to reflect a lower risk level of 10^{-5}.

A.1.2 Results

Exhibit A-1 summarizes the results of the analysis of lowering the risk level for carcinogens in the proposed CTR. As Exhibit A-1 shows, the changes in estimated costs and pollutant load reductions based on the lower risk level of 10^{-5} are minimal. Under the low scenario, costs decrease by $1.1 million, approximately 11% less than the costs based on the higher risk level. Under the high scenario, annual costs decrease by $5.8 million, also an 11% decrease from the costs based on a 10^{-6} risk level. Pollutant load reductions attributable to use of a lower risk level are estimated to decrease by approximately 4% and 1% under the low and high scenarios, respectively.

Exhibit A-1. Comparison of Estimated Costs if CTR-Based WQBELS Are Calculated Using a Cancer Risk Level of 10^{-5}

Approach	Low Scenario		High Scenario	
	Estimated Annual Costs ($millions)	Load Reductions (10^6 lbs-eq/yr)	Estimated Annual Costs ($millions)	Load Reductions (10^6 lbs-eq/yr)
Baseline Cost Analysis (10^{-6})	$33.5	1.08	$61.0	2.73
Alternative Analysis (10^{-5})	$32.4	1.04	$55.2	2.69

Note: All costs are in first quarter 1998 dollars.

The low sensitivity to the change in risk level primarily is related to the fact that most of the potential costs related to implementing the CTR are being driven by metals. Changes in risk levels for carcinogens primarily affect organic pollutants.

A.2 IMPACT OF METAL TRANSLATORS

The criteria for metals in the proposed rule are expressed in the dissolved form. Where a site specific or theoretical "translator" is used, the use of dissolved metals criteria usually results in permit limits that are less stringent than those derived from total recoverable criteria. The dissolved criteria in the CTR are derived by multiplying the total recoverable criterion by a conversion factor. Permitting regulations, however, require that permit limits be set in terms of total recoverable metals concentrations. Therefore, permit writers must "translate" dissolved criteria to derive total recoverable permit limits which can be done through a variety of methods.

One method employs site-specific information to derive the translator. This is EPA's preferred approach since it is likely to result in the best estimate of actual in-stream partitioning relationships. However, since not all site-specific information was available, the base analysis presented in Chapter 4 used a second method, the theoretical partitioning relationship, to estimate the translator. The theoretical partitioning relationship is based on a partitioning coefficient determined empirically for each metal and, when available, the concentration of total suspended solids in the site-specific receiving water. According to recent EPA guidance on translators (The Metals Translator: Guidance for Calculation of a Total Recoverable Permit Limit From a Dissolved Criteria), this method usually tends to overstate the stringency of the derived permit limit compared to the site-specific method, although it will sometimes understate the stringency (U.S. EPA, 1996). A third method is to simply use the total recoverable criteria that are derived by dividing the dissolved criteria by the conversion factor. This method is very conservative and will, in nearly all cases, result in more stringent permit limits compared to the site-specific method.

Although EPA encourages the use of site-specific translators, some members of the regulated community expressed concern that the state may not choose this approach to derive permit limits. Thus, EPA performed a sensitivity analysis.

A.2.1 Methodology

EPA performed a sensitivity analysis to estimate the effect of the use of total recoverable criteria on CTR-based WQBELs, total costs, and load reductions. EPA calculated CTR-based WQBELs using the same methods described in Chapter 4, except that it used total recoverable criteria in place of dissolved criteria for metals.

A.2.2 Results

The results of this analysis show that costs may be sensitive to the translator chosen by the state. **Exhibit A-2** shows the expected costs and load reductions using conversion factors as the translators.

Exhibit A-2. Comparison of Potential Costs if CTR-based WQBELS Are Calculated Using Criteria Expressed as Total Recoverable

Approach	Low Scenario			High Scenario		
	Estimated Annual Costs ($millions)	Load Reductions (10^6 lbs-eq/yr)	Cost-Effectiveness ($/lb-eq)	Estimated Annual Costs ($millions)	Load Reductions (10^6 lbs-eq/yr)	Cost-Effectiveness ($/lb-eq)
Baseline Cost Analysis	$33.5	1.08	31	$61.0	2.73	22
Alternative Analysis	$62.4	1.25	50	$325.0	2.94	111

Note: All costs are in first quarter 1998 dollars.

As Exhibit A-2 shows, a significant increase in costs can be expected, as compared to the costs of the theoretical partitioning approach used in the base analysis. Potential annual costs under the low scenario are $62.4 million per year, an approximately two-fold increase over the estimates in the low base analysis. Under the high-end scenario, total costs are estimated to be nearly $325 million per year, over five times the cost estimates in the base analysis. Potential load reductions are estimated to increase by approximately 14% over the low base-case scenario, and by nearly 7% under the high scenario. Using conversion factors as translators would result in significantly higher costs per toxic pound-equivalent removed than the base analysis. The cost-effectiveness of the new low scenario is $50 per toxic pound-equivalent removed compared to $31 per toxic pound-equivalent removed in the base analysis. The cost-effectiveness of the new high scenario is $111 per toxic pound-equivalent removed compared to $22 per toxic pound-equivalent removed in the base analysis.

EPA believes that the costs estimated from this analysis greatly overstate true costs. EPA expects that in cases where a facility may incur substantial economic impacts due to an effluent limit for a metal, there will be strong incentives for the facility or the state to develop site-specific data, which will result in more realistic translators, thus reducing potential economic impacts. EPA believes that the cost estimates developed using the theoretical partitioning approach in the base case are more realistic than the cost estimates from this sensitivity analysis.

APPENDIX B. COMPLIANCE COST DECISION MATRIX

I. REASONABLE POTENTIAL

1. Does the pollutant have a reasonable potential to exceed water quality criteria?

No →

No compliance costs.

↓ **Yes**

II. MODIFY/ADJUST EXISTING TREATMENT

1. Is the WQBEL for the pollutant above analytical detection levels?
2. Does the existing wastewater treatment system have the capability to treat/remove the pollutant?
3. Is the WQBEL for the pollutant greater than documented treatable levels?
4. Are modifications/adjustments to the existing wastewater treatment system feasible in light of the pollutant reduction necessary to achieve the WQBEL (i.e., is the reduction less than 10-25% of the current discharge levels)?

Yes →

Facility incurs costs to modify/adjust existing treatment system.

↓ **No**

III. WASTE MINIMIZATION/POLLUTION PREVENTION

1. Is the production process or source generating the pollutant amenable to waste minimization/pollution prevention techniques?

<u>Industrials Only</u>

2. Is the WQBEL for the pollutant above analytical detection levels?
   - If no, is the production process or pollutant source amenable to control techniques expected to reduce pollutant to below analytical detection levels (e.g., product substitution)?
3. Do any of the following conditions apply?:
   - Is the level of pollutant reduction required to meet the WQBEL insignificant (i.e., less than 10-25% of current discharge levels)?
   - Is the pollutant most often in compliance with the projected effluent limit? (i.e., 80% or more of the observations).
   - Are discharge monitoring data inconclusive to assume treatment costs? (e.g., no data exist, are limited or do not reflect existing discharge conditions).

<u>POTWs Only</u>

2. Is the WQBEL for the pollutant above analytical detection levels?
3. Do any of the following conditions apply?:
   - Is the level of pollutant reduction required to meet the WQBEL insignificant (i.e., less than 10-25% of current discharge levels)?
   - Is the pollutant most often in compliance with the projected effluent limit? (i.e., 80% or more of the observations).
   - Are discharge monitoring data inconclusive to assume treatment costs? (e.g., no data exist, are limited, old, or mostly below detection level).
4. Are increased industrial user/source controls feasible?

Yes →

Facility incurs costs to implement waste minimization/pollution prevention.

↓ **No**

IV. NEW/ADDITIONAL TREATMENT SYSTEM		
1. Is the WQBEL for the pollutant above analytical detection levels? 2. Is the effluent concentration for the pollutant sufficiently documented above detection levels and above documented treatable levels? 3. Is the new/additional treatment technology feasible in light of the: - existing treatment process(es)? - production process(es)? - pollutant source(s)? - level of pollutant reductions required to achieve the WQBEL (i.e., is it greater than 10-25% of the current discharge levels)? - cost to add the necessary treatment? [Note: Under the low scenario, the cost trigger is $200/toxic lb-equivalent for a specific facility for a pollutant].	**Yes** →	Facility incurs costs to install additional end-of-pipe treatment (or in-plant treatment).

↓ **No**

V. OTHER CONTROLS		
1. Is the pollutant concentration above analytical detection levels, and treatable levels; and is the WQBEL below analytical detection levels? 2. Is a combination of end-of-pipe treatment and waste minimization/pollution prevention feasible in light of the: - existing treatment process(es)? - pollutant source(s)? - level of pollutant reductions required to achieve the WQBEL (i.e., is it greater than 10-25% of the current discharge levels)? - cost to add the necessary treatment? [Note: Under the low scenario, the cost trigger is $200/toxic lb-equivalent for a specific facility for a pollutant].	**Yes** →	Facility incurs costs for other controls.

↓ **No**

↓ ↓ ↓ ↓ ↓

VIa. Phased TMDL	VIb. Variances from Water Quality Standards	VIc. Site-Specific Criteria	VId. Change Designated Use	VIe. Alternative Mixing Zone
1. Is the discharge to a non-attainment receiving water? 2. Are the other sources of pollutants to the receiving water known?	1. Is the pollutant naturally occurring? 2. Are there natural, ephemeral, intermittent or low flow conditions? 3. Are there human-caused conditions or sources? 4. Are dams, diversions, or other types of hydrologic modifications present? 5. Do the physical conditions related to the natural features of the water body contribute? 6. Would the controls result in substantial and widespread economic and social impact? If yes, will the discharge comply with anti-degradation requirements and cause no increased risk to human health and the environment?	1. Are local environmental conditions not reflected in criteria? 2. Are bio-accumulation factors appropriate?	1. Is the pollutant naturally occurring? 2. Are there natural, ephemeral, intermittent or low flow conditions? 3. Are there human-caused conditions or sources? 4. Are dams, diversions, or other types of hydrologic modifications present? 5. Do the physical conditions related to the natural features of the water body contribute? 6. Would the controls result in substantial and widespread economic and social impact?	1. Does the receiving water body offer a dilution ratio higher than the one presently indicated in the permit?
Facility incurs future cost to comply with TMDL.	Facility incurs costs for preparing variance request and future compliance costs when variance expires.	Facility incurs costs for preparing request for site-specific criteria.	Facility incurs costs associated with preparing a use attainability analysis.	Facility incurs costs to prepare demonstration.

APPENDIX C. DETAILED COST ESTIMATES

Costs, Low, Extrapolated

Pollutant	Total Capital	Annual Capital	Mon	O&M	Total Waste Min.	Annual Waste Min.	Total TPO	Annual TPO	Total TS	Annual TS	Total PT Study	Annual PT Study	Total LL Eval	Annual LL Eval	Total Variance	Annual Variance	Total Other	Annual Other	Totals By Poll.	Percent of Total
Antimony (Sb)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Arsenic (As)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Cadmium (Cd)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Chromium VI (Cr-VI)	0	0	0	0	0	0	1,078,524	153,558	0	0	0	0	0	0	0	0	0	0	153,558	1.5%
Copper (Cu)	0	0	0	0	4,748,214	676,039	8,771,000	1,248,793	0	0	0	0	0	0	1,676,190	238,652	0	0	2,163,484	21.8%
Lead (Pb)	0	0	0	0	0	0	575,000	81,867	0	0	0	0	0	0	0	0	0	0	81,867	0.8%
Mercury (Hg)	0	0	0	0	9,000,000	1,281,398	548,952	78,158	0	0	0	0	0	0	0	0	0	0	1,359,556	13.7%
Nickel (Ni)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Selenium (Se)	0	0	0	0	0	0	616,667	87,799	0	0	0	0	0	0	0	0	0	0	87,799	0.9%
Silver (Ag)	0	0	0	0	1,850,000	263,398	1,165,619	165,958	0	0	0	0	0	0	0	0	0	0	429,356	4.3%
Thallium (Tl)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Zinc (Zn)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1,676,190	238,652	0	0	238,652	2.4%
1,2 Dichlorobenzene	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
1,2 Dichloroethane	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
1,2 Dichloropropane	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
1,2-Trans-Dichloroethylene	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
1,3 Dichlorobenzene	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
1,3-Dichloropropylene	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
1,4 Dichlorobenzene	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
2,4-Dinitrophenol	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
2,4,6 Trichlorophenol	0	0	0	0	462,500	65,850	0	0	0	0	0	0	0	0	0	0	0	0	65,850	0.7%
4,4'-DDD	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
4,4'-DDT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Aldrin	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
alpha-BHC	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
alpha-Endosulfan	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Benzene	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Benzo (a) Anthracene	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Benzo (a) Fluoranthene	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Benzo (k) Fluoranthene	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
beta-BHC	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
beta-Endosulfan	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Bromoform	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Butylbenzyl-pthalate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Carbon Tetrachloride	0	0	0	0	5,533,333	787,822	0	0	0	0	0	0	0	0	0	0	0	0	787,822	7.9%
Chlordane	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Chlorobenzene	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Chlorodibromomethane	0	0	0	0	0	0	4,221,111	600,991	0	0	0	0	0	0	0	0	0	0	600,991	6.1%
Chloroform	0	0	0	0	0	0	4,683,611	666,841	0	0	0	0	0	0	0	0	0	0	666,841	6.7%
Dichlorobromomethane	0	0	0	0	0	0	4,683,611	666,841	0	0	0	0	0	0	0	0	0	0	666,841	6.7%
Dieldrin	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Di-n-Butyl Phthalate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Endosulfan Sulfate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Endrin	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Endrin Aldehyde	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Fluoranthene	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Fluorene	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
gamma-BHC	0	0	0	0	4,462,500	635,360	0	0	0	0	0	0	0	0	0	0	0	0	635,360	6.4%
Heptachlor	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Heptachlor epoxide	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Hexachlorobenzene	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Methylene chloride	0	0	0	0	5,533,333	787,822	0	0	0	0	0	0	0	0	0	0	0	0	787,822	7.9%
PCBs	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Pentachlorophenol	0	0	0	0	462,500	65,850	0	0	0	0	0	0	0	0	0	0	0	0	65,850	0.7%
Phenol	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
TCDD	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Tetrachloroethylene	0	0	0	0	8,000,000	1,139,020	0	0	0	0	0	0	0	0	0	0	0	0	1,139,020	11.5%
Toluene	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Toxaphene	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Trichloroethylene	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%

Costs, High, Extrapolated

Pollutant	Total Capital	Annual Capital	Mon	O&M	Total Waste Min.	Annual Waste Min.	Total TPO	Annual TPO	Total TS	Annual TS	Total PT Study	Annual PT Study	Total LL Eval	Annual LL Eval	Total Variance	Annual Variance	Total Other	Annual Other	Totals By Poll.	Percent of Total
Antimony (Sb)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Arsenic (As)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Cadmium (Cd)	0	0	0	0	7,904,762	1,125,460	0	0	0	0	0	0	0	0	0	0	0	0	1,125,460	2.2%
Chromium VI (Cr-VI)	0	0	0	0	11,854,762	1,687,851	3,312,964	471,692	0	0	0	0	0	0	0	0	0	0	2,159,543	4.2%
Copper (Cu)	4,777,143	680,158	0	418,000	27,486,310	3,913,432	749,302	106,684	0	0	0	0	0	0	0	0	0	0	5,118,273	10.1%
Lead (Pb)	0	0	0	0	11,936,508	1,699,490	958,333	136,445	0	0	0	0	0	0	0	0	0	0	1,835,935	3.6%
Mercury (Hg)	0	0	0	0	33,281,746	4,738,572	365,968	52,106	0	0	0	0	0	0	0	0	0	0	4,790,678	9.4%
Nickel (Ni)	4,777,143	680,158	0	418,000	8,521,429	1,213,260	961,298	136,867	0	0	0	0	0	0	0	0	0	0	2,448,285	4.8%
Selenium (Se)	0	0	0	0	12,163,095	1,731,751	961,298	136,867	0	0	0	0	0	0	0	0	0	0	1,868,618	3.7%
Silver (Ag)	4,777,143	680,158	0	418,000	25,279,762	3,599,269	3,678,933	523,797	0	0	0	0	0	0	0	0	0	0	5,221,224	10.3%
Thallium (Tl)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Zinc (Zn)	4,777,143	680,158	0	418,000	698,413	99,438	0	0	0	0	0	0	0	0	0	0	0	0	1,197,596	2.4%
1,2 Dichlorobenzene	0	0	0	0	25,657,778	3,653,090	0	0	0	0	0	0	0	0	0	0	0	0	3,653,090	7.2%
1,2 Dichloroethane	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
1,2 Dichloropropane	0	0	0	0	0	0	2,470,667	351,767	0	0	0	0	0	0	0	0	0	0	351,767	
1,2-Trans-Dichloroethylene	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
1,3 Dichlorobenzene	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
1,3-Dichloropropylene	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
1,4 Dichlorobenzene	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
2,4-Dinitrophenol	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
2,4,6 Trichlorophenol	0	0	0	0	462,500	65,850	0	0	0	0	0	0	0	0	0	0	0	0	65,850	0.1%
4,4'-DDD	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
4,4'-DDT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Aldrin	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
alpha-BHC	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
alpha-Endosulfan	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Benzene	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Benzo (a) Anthracene	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Benzo (a) Fluoranthene	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Benzo (k) Fluoranthene	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
beta-BHC	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
beta-Endosulfan	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Bromoform	0	0	0	0	326,786	46,527	0	0	0	0	0	0	0	0	0	0	0	0	46,527	0.1%
Butylbenzyl-pthalate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Carbon Tetrachloride	0	0	0	0	18,771,230	2,672,601	0	0	0	0	0	0	0	0	0	0	0	0	2,672,601	5.3%
Chlordane	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Chlorobenzene	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Chlorodibromomethane	0	0	0	0	0	0	5,059,206	720,317	0	0	0	0	0	0	0	0	0	0	720,317	1.4%
Chloroform	0	0	0	0	8,156,786	1,161,343	4,883,611	695,316	0	0	0	0	0	0	0	0	0	0	1,856,659	3.6%
Dichlorobromomethane	0	0	0	0	326,786	46,527	4,683,611	666,841	0	0	0	0	0	0	0	0	0	0	713,368	1.4%
Dieldrin	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Di-n-Butyl Phthalate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Endosulfan Sulfate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Endrin	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Endrin Aldehyde	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Fluoranthene	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Fluorene	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
gamma-BHC	0	0	0	0	20,462,500	2,913,400	0	0	0	0	0	0	0	0	0	0	0	0	2,913,400	5.7%
Heptachlor	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Heptachlor epoxide	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Hexachlorobenzene	0	0	0	0	2,213,333	315,129	0	0	0	0	0	0	0	0	0	0	0	0	315,129	0.6%
Methylene chloride	0	0	0	0	18,444,444	2,626,074	0	0	0	0	0	0	0	0	0	0	0	0	2,626,074	5.2%
PCBs	0	0	0	0	5,728,571	815,620	0	0	0	0	0	0	0	0	0	0	0	0	815,620	1.6%
Pentachlorophenol	0	0	0	0	7,675,833	1,092,866	0	0	0	0	0	0	0	0	0	0	0	0	1,092,866	2.1%
Phenol	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
TCDD	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0%
Tetrachloroethylene	0	0	0	0	40,326,786	5,741,627	0	0	0	0	0	0	0	0	0	0	0	0	5,741,627	11.3%
Toluene	0	0	0	0	10,252,024	1,459,658	0	0	0	0	0	0	0	0	0	0	0	0	1,459,658	2.9%
Toxaphene	0	0	0	0	326,786	46,527	0	0	0	0	0	0	0	0	0	0	0	0	46,527	0.1%
Trichloroethylene	0	0	0	0	326,786	46,527	0	0	0	0	0	0	0	0	0	0	0	0	46,527	0.1%