United States
Environmental Protection
Agency

Office of Water
4301

EPA-820-B-96-001
July 1997

Economic Analysis of the Proposed California Water Quality Toxics Rule

ECONOMIC ANALYSIS
OF THE
PROPOSED CALIFORNIA WATER QUALITY TOXICS RULE

July 15, 1997

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EPA Contract No. 68-C4-0046; Work Assignment No. 2-20
SAIC Project No. 01-0833-07-7502-200

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EXECUTIVE SUMMARY

Under the Clean Water Act (CWA), the State has the primary authority for establishing designated uses for waterbodies 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 approved and disapproved portions of each plan. In December 1992, EPA promulgated the National Toxics Rule (NTR) (57 FR 60848; December 22, 1992) for several States which had not yet met the requirements of CWA section 303(c)(2)(B), including the State of California for those portions of the statewide plans that it had disapproved.

Shortly after the ISWP and EBEP were adopted in 1991, 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 finding the SWRCB had not complied with State law. In September of 1994, the SWRCB took formal action to rescind the plans. Since then, the State of California has been without a complete set of water quality criteria for priority toxic pollutants, as required by section 303(c)(2)(B) of the CWA. 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 for the State of California. The proposed California Toxics Rule (CTR) would apply to 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 proposed in the 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

Under Executive Order (E.O.) 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 E.O., i.e., drafting an Economic Analysis (EA) and review by the Office of Management and Budget (OMB). Pursuant to the terms of the order, EPA has determined that this proposed rule is not "significant". The proposed CTR establishes ambient water quality criteria which, by themselves, do not directly impose economic impacts. When these criteria are combined with the State-adopted designated uses for inland surface waters, enclosed bays and estuaries, water quality standards will be created. EPA acknowledges that there may be a cost to some dischargers for complying with new water quality standards after those standards are translated into specific NPDES permit limits by the State. Consistent with the intent of E.O. 12866, EPA prepared an EA. Since the State has significant flexibility and discretion in how it chooses to implement standards within the NPDES permit program, the EA by necessity includes many assumptions about how the State will implement the water quality standards. These assumptions are based on a combination of EPA guidance and current permit conditions for the facilities examined in this analysis. (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.) A more precise measure of costs and benefits may not be known until the State adopts its implementation provisions. To account for the uncertainty of these assumptions, this analysis estimates a wide range of costs and benefits. By completing the EA, EPA intends to inform the public about how entities might be affected by implementation of CTR-based water quality standards in the NPDES permit program.

SCOPE OF ECONOMIC ANALYSIS

This EA estimates costs to point source dischargers that may be subject to water quality-based effluent limits (WQBELs) calculated using the CTR criteria, and the benefits attributable to those point sources. The point sources included in this study only include those that discharge to waters of the U.S. which also discharge to California inland surface waters and enclosed bays and estuaries. Under the CWA, EPA has direct authority regarding permits issued under the NPDES program. This authority is delegated to the State of California. EPA did not calculate costs for any program for which it does not have enforceable authority, such as agricultural nonpoint sources. In addition, EPA did not calculate costs for NPDES sources which are not typically subject to numeric WQBELs, including sources required to hold NPDES stormwater permits and other wet weather dischargers.

EPA acknowledges that the water quality criteria in this rule may also have an indirect effect on sources not permitted under the NPDES program or not subject to numeric water quality-based effluent limits. Any potential indirect effect on these sources is unknown at this time. These nonpoint sources and wet weather discharges can include, but are not limited to, runoff from farms, urban areas, and abandoned mines, and contaminated sediments. The State may ask or require these sources to implement best management practices or participate in a comprehensive watershed management planning approach. Control strategies for wet weather discharges and of nonpoint sources are an important part of EPA and California's overall strategy to improve water quality.

DESCRIPTION OF BASELINES

In order to estimate the costs and benefits, EPA established an appropriate baseline. The baseline was the starting point for measuring incremental costs and benefits of a regulation. The baseline for the CTR was established by assessing what would occur in the absence of a proposed regulation. EPA estimated the incremental costs and benefits of potential State implementation of the CTR using two different baselines to account for the uncertainty of how the State might implement water quality standards through NPDES permits in the absence of the proposed CTR.

Model 1: Negligible Costs and Benefits

The baseline used in this model assumes that the State will rely on its narrative toxicity standard and implementation of the standard using "best professional judgment" (BPJ) to place numeric water quality-based effluent limits for toxic pollutants into 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, under one scenario, permit writers could comply with the permitting regulations by basing numeric effluent limits on narrative standard that incorporate the latest scientific information. This approach would likely 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.

Thus, under this scenario, the costs and benefits of the CTR would be negligible since implementation of permits under the CTR would not differ significantly from how the State may implement permits under current law.

Model 2: Most Likely Estimate of Costs and Benefits

The baseline used in Model 2 assumes that in the absence of the rule, current permit requirements and current effluent concentrations would continue in the future. The baseline generally uses current permit limits to develop a high scenario cost estimate and current effluent concentrations to develop a low scenario cost estimate.

Permits in the State are written and issued by nine autonomous Regional Water Quality Control Boards. Therefore, the derivation of effluent limits varies from Board to Board. Some of the effluent limits used in this baseline were derived from criteria in California's statewide plans, since some permits were issued during the time the plans were in effect. However, since the plans have been revoked, permit writers no longer use the criteria contained in the plans. Regional Boards have been using a variety of methods to derive numeric permit limits including the use of numeric criteria adopted in the mid-1980s and drinking water maximum contaminant levels (MCLs) developed under the Safe Drinking Water Act. Given this diversity, this baseline best represents the current variable situation in California, and will yield a comparative result that best represents actual incremental costs and benefits that will occur when the CTR is implemented. The methodology described below uses Model 2 as the operating assumption for the estimation of costs.

METHOD FOR ESTIMATING POTENTIAL COMPLIANCE COSTS AND POLLUTANT LOADING REDUCTIONS UNDER MODEL 2

In general, the method used by EPA to estimate costs and pollutant load reductions attributable to implementation of the proposed CTR was to develop detailed cost estimates for a selected subset (the sample) of facilities from the 184 major point source dischargers and 1,057 permitted minor point source dischargers to California's inland waters, enclosed bays, and estuaries, and then use the sample results to extrapolate to the universe of potentially affected facilities. For this analysis, 20 sample facilities were selected for analysis; 17 major dischargers and 3 minor dischargers. These sample facilities were selected to represent the range of discharger categories and geographic distribution of all permitted dischargers to inland waters, enclosed bays, and estuaries. Some simplifying assumptions were made to facilitate analysis and to overcome data limitations, where necessary. These assumptions, for the most part, were designed to be "conservative," that is, to err on the side of more stringent and costly controls, rather than less.

The actual impact of the CTR after water quality standards are translated into specific NPDES permit limits will vary depending upon the procedures used to implement the CTR. These procedures typically specify the methods to determine the need for water quality-based effluent limits (WQBELs) and, if WQBELs are required, how to derive WQBELs from applicable numeric water quality criteria. The implementation procedures used to derive WQBELs for this study were based on the methods recommended in the EPA *Technical Support Document for Water Quality-based Toxics Control* (or TSD) (EPA/505/2-90-001; March 1991). Since States are not required to use the methods in the TSD, implementation procedures can vary, and may result in more or less stringent WQBELs. Due to the uncertainty of the State's approach to implementation at this time, a range of costs were developed to represent a potential range of impact based on certain implementation assumptions. The following generally describes how the low- and high-end of the range of costs were developed for this study:

Low-end Cost Scenario

- If the CTR-based permit limit was more stringent than existing effluent concentrations, then costs were estimated for the incremental pollutant reductions required to achieve the CTR-based limit.

- In the absence of any monitoring data, it was assumed that no impacts would occur, even if a permit limit exists that is less stringent than the CTR-based limit. (It was assumed that if a facility is not monitoring for a pollutant, then it is not expected to be present in the effluent.)

- If monitoring data were available, but all values were reported as below analytical detection levels, it was assumed that no impacts would occur.

- If the estimated annualized cost for removal of a pollutant exceeded $200 per toxic pounds-equivalent then it was assumed that dischargers would explore the use of alternative regulatory approaches (e.g., variances, site-specific criteria, etc.). When it was assumed that facilities would pursue alternative regulatory approaches, no treatment cost was estimated for a facility; however, costs were estimated for pursuing the alternative.

High-end Cost Scenario

- If the CTR-based permit limit was more stringent than the existing permit limit (or detectable effluent concentration in the absence of a permit limit), then costs were estimated for the incremental pollutant reductions required to achieve the CTR-based limit.

- If only monitoring data were available (i.e., no permit limit), but all values were reported as below analytical detection levels, it was assumed that no impacts would occur.

- Acknowledging that opportunities for the use of alternative regulatory approaches may be limited depending upon the particular circumstances for a "facility," costs were also estimated under a higher cost scenario that assumes alternative regulatory approaches would be pursued only when the cost for the particular "category of dischargers" exceeds a cost trigger. Particularly, if the estimated annualized cost for a "category of dischargers" exceeded $500 per toxic pounds-equivalent then it was assumed that dischargers within the "category" would be pursue alternative regulatory approaches. However, this situation did not occur in our high-end analysis.

Prior to estimating compliance costs, an engineering analysis of how each sample facility could comply with the CTR-based effluent limitations was performed. The costs were then estimated based on the decisions and assumptions made in the analysis. To ensure consistency in estimating the general types of controls that would be necessary for a sample facility to comply with the CTR (assuming that implementation of the CTR resulted in more stringent requirements), as well as to integrate into the cost analysis the other alternatives available to regulated facilities, a costing decision matrix was developed that was used for each sample facility. The underlying assumption of the decision matrix was that a facility will examine lower-cost alternatives prior to incurring the expense and potential liabilities associated with constructing end-of-pipe treatment facilities.

Once the low-end and the high-end estimates of costs were generated, corresponding loadings reductions of pollutants were estimated for each scenario. For the low- and the high-end scenarios, the baseline loadings represent the existing pollutant discharge level and the existing permit limit for a pollutant, respectively. The difference between the baseline and the potential load from CTR-based WQBELs represent the reduction from implementation of the CTR. Both baseline loads and load reductions are expressed in toxic pounds-equivalent, which is a normalized measurement of pollutants that accounts for relative toxicities among pollutants.[1]

Once the costs, toxic-weighted baseline loads, and toxic-weighted load reductions were computed for the sample, the costs were extrapolated (for both scenarios) to encompass all direct dischargers to bays, estuaries, and inland waters. The extrapolated costs represent the total annual estimated compliance cost resulting from implementation of the CTR.

Cost estimates were also prepared to account for the potential impact on indirect dischargers. Cost estimates for indirect dischargers were prepared by using estimates from earlier studies by two major POTWs in the State and an assumed range of potentially impacted indirect dischargers.

POTENTIAL COMPLIANCE COSTS RESULTS

The cost of the implementation of the CTR will vary depending upon which baseline is used. Model 1 assumes that in the absence of the CTR, the State of California would require effluent limits in permits based on the existing narrative water quality standard which would be essentially equivalent to numeric water quality standard based on criteria proposed in the CTR. Therefore the use of Model 1 results in negligible costs.

Alternatively, using Model 2, the potential annual cost for implementing the CTR is estimated to range from about \$15 million to \$87 million for the low- and high-end scenarios, respectively. As shown in **Exhibit ES-1**, indirect dischargers are estimated to incur slightly more of the potential cost burden in the low-end scenario than direct dischargers. However, under the high-end scenario, direct dischargers are expected to incur most of the potential costs. EPA believes that costs incurred after implementation of CTR-based permit limits will approach the low-end of the cost range. Costs are unlikely to reach the high-end of the range because State authorities are likely to choose implementation options that provide some degree of flexibility to dischargers. Furthermore, cost estimates for both scenarios, but especially for the high-end scenario, may be overstated because the analysis tended to use conservative

[1] Incidental removals of conventional and non-conventional pollutants were not taken into account in this analysis.

assumptions in calculating CTR-based permit limits and in establishing baseline loadings. The baseline loadings for the high-end were generally based on current effluent limits rather than actual pollutant discharge data. Most facilities discharge pollutants in concentrations below current effluent limits.

EXHIBIT ES-1. SUMMARY OF ANNUALIZED COST ESTIMATES FOR THE TWO COST MODELS (IN $ MILLIONS)

Cost Category	Model 1	Model 2	
		Low-End Scenario	High-End Scenario
Direct Dischargers	$0	$5.2	$83.4
Indirect Dischargers	$0	$9.7	$3.2
TOTAL	**$0**	**$14.9**	**$86.6**

Note: Costs are in first quarter 1996 dollars.

Low-end Cost Scenario

Under the low-end cost analysis, major publicly owned treatment works (POTWs) are expected to incur the largest percentage (67 percent) of the total projected annualized costs. However, distributed among 128 POTWs discharging to inland water, bays and estuaries within the State, the average cost per plant would be just over $27,000 per year. Chemical and petroleum industries are estimated to incur the highest cost of the industrial categories: 18 percent of the total annual costs of implementation of the CTR, with an annual average of $47,500 per facility. Facilities within the metals and transportation equipment category make up less than 8 percent of the total costs. However, the average cost per plant for facilities within the metals and transportation equipment category is the highest for the direct dischargers at almost $57,000 per year. The mining and lumber and paper categories are estimated to incur no costs related to the State implementing the CTR.

Major permitted dischargers account for 100 percent of the annual costs under the low-end scenario. None of the minor sample facilities were projected to incur costs related to implementation of the CTR.

Consistent with the intent of the low-end to reflect the State's flexibility in implementing the CTR, annualized costs for alternative regulatory approaches comprise just over 74 percent of the total costs. Regulatory relief was assumed necessary in the low-end cost analysis when the total cost for a sample facility exceeded $200 per toxic pounds-equivalent of pollutant reduced. Annualized costs for developing and implementing waste minimization plans account for the bulk of the remaining costs (20 percent), primarily because of the smaller increment of pollutant to be removed (as compared to the increment required for the high-end), minimizing the need to expand existing treatment systems or install new treatment.

Pollution control of five pollutants (i.e., mercury, silver, chromium VI, aldrin, and chlorodibromomethane) account for just over 50 percent of all annual costs. Under the low-end scenario, almost 60 percent of potential annual costs are to control toxic organics; costs to control metals and mercury account for approximately 40 percent of all annual costs.

High-end Cost Scenario

Under the high-end cost analysis, major POTWs may incur almost 74 percent of the total projected annualized cost, a total of just over $61 million. However, distributed between the 128 major POTWs in the State, the average cost per plant would be approximately $480,000 per year. Chemical and petroleum industries will incur the highest cost of the industrial dischargers: just over 16 percent of the total estimated annual cost, and averaging almost $678,000 per plant. Facilities within the metals and transportation equipment category constitute less than 7 percent of the total costs. However, the average cost per plant for facilities within the metals and transportation equipment category is the highest for the direct dischargers at almost $817,000 per year. As with the low-end scenario, the mining and lumber and paper categories are estimated to incur no costs related to the State implementing the CTR.

Major permitted dischargers account for 100 percent of the annual costs under the high-end scenario. None of the minor sample facilities were projected to incur costs related to the State implementing the CTR.

Due to the conservative nature of the high-end scenario, annualized operation and maintenance (O&M) and capital costs comprise almost 90 percent of the annual costs. Annualized O&M costs comprise almost 60 percent of the total annual costs. This result is driven primarily by the fact that the increment of pollutant removal is relatively small (i.e., less than 25 percent of current effluent levels), and that many of the sample facilities already possessed treatment processes that could achieve CTR-based effluent limits. Therefore, increased O&M was assumed adequate to comply with CTR-based effluent limits (as opposed to installing new treatment equipment). Capital costs constitute 29 percent of the total annual costs.

Alternative regulatory approaches were assumed to be necessary in the high-end cost analysis when the total cost for a discharger exceeded $500 per toxic pound-equivalent of pollutant reduced. No categories/facilities exceeded this $500 cost trigger, and therefore, no such costs were incurred under the high-end scenario.

Pollution control for three pollutants (i.e., mercury, silver, and chromium VI) contribute more than one-third of the estimated annual implementation costs. Costs to control metals and mercury account for almost 60 percent of all annual costs; costs to control toxic organic pollutants account for just over 40 percent.

Costs to Indirect Dischargers

According to EPA Region 9, there are approximately 2,100 significant industrial users (SIUs) that discharge to POTWs located on inland surface waters, enclosed bays, or estuaries in California. POTWs, faced with more stringent limits, may choose to control toxic pollutant discharges into their plant by requiring SIUs and other indirect dischargers to comply with stricter local limits. Based on studies at two major POTWs in California, the estimated compliance costs for SIUs are $9.6 million (representing an assumed impact on 30 percent of the population of SIUs) and $3.2 million (10 percent impact) for the low- and the high-end scenarios, respectively.

APPORTIONMENT OF COSTS BETWEEN FEDERAL AND STATE ACTIONS

EPA is proposing this rule to fill a gap in California's water quality standards that resulted from litigation rescinding its water quality control plans, which contained water quality criteria for toxic pollutants. Consequently the State is not meeting the requirements of section 303(c)(2)(B) of the CWA, necessitating this federal action. However, the State is moving forward with its own rulemaking effort to restore the rescinded water quality criteria at some future date. Once the State adopts its own water quality criteria and EPA approves these criteria, EPA will take action to stay or withdraw the federal criteria.

If the federal action turns out to be only temporary, the implementation costs of the criteria should not be solely attributed to the federal action. If the implementation costs of the criteria are apportioned between the federal action and the State action, depending on when the State criteria replace the federal criteria, the high-end costs estimated using Model 2 ($87 million annually) could be shared based on the time period over which costs are expected to be incurred.[2] For example, if the State restores criteria 5 years after promulgation of the federal rule, it would assume approximately 42 percent of the implementation costs.[3]

POLLUTANT LOADING REDUCTIONS AND COST-EFFECTIVENESS

In addition to the estimated costs, the overall pollutant loading reductions under both the low- and the high-end scenarios were estimated. Load reductions were computed on a toxic pounds-equivalent basis. Toxic pounds-equivalent represent a unit of measurement that permit uniform comparison among pollutants based on their relative toxicity. For example, reducing the discharge of aluminum by 10 pounds will have a different effect on the environment as compared to reducing mercury by 10 pounds. Expression of pollutant loads and reductions in terms of a pollutant's relative toxicity allows direct comparisons among pollutants for the purposes of cost-effectiveness analysis.

The toxicity-weighted baseline loading under the low- and the high-end scenarios are approximately 3.6 and 23.8 million toxic pound-equivalents per year, respectively. The large difference between the baseline loads is the result of the method between the scenarios to compute loads and loading reductions. Under the low-end scenario, the existing discharge levels are used while under the high-end scenario the existing permit limits are used. Generally, facilities discharge at concentrations well below existing permit limits and therefore baseline loadings under the low- and high-end scenarios differ considerably.

Approximately 0.63 million toxic pounds equivalent are expected to be reduced under the low-end scenario. Of this reduction, chromium VI accounts for just over 40 percent; mercury, 28 percent; and silver, 21 percent. Overall, toxic organic pollutant reductions account for just over 7 percent of the total reductions under the low-end scenario.

A similar pattern exists for the high-end scenario, where approximately 7 million toxic pound-equivalents are expected to be reduced. Over 80 percent of the total projected toxic-weighted annual reductions will come from reducing metals, including mercury, while less than 15 percent of expected reductions are for organic pollutants. Of the metals that will be reduced, silver accounts for almost 60 percent of the total annual reductions and mercury accounts for almost 16 percent. Of the organics, toluene accounts for 10.5 percent of the total annual reductions, while several other organic pollutants are reduced at relatively small percentages.

Under both scenarios, the POTW category accounts for almost all of the loading reductions. Among industrial categories in the low-end scenario, electric utilities account for the largest reductions at 0.2 percent. A similar pattern is evident under the high-end scenario with POTWs accounting for most of the reductions.

Cost-Effectiveness

Cost-effectiveness is defined as the cost in dollars to remove one toxic pound equivalent. The overall cost-effectiveness is estimated to be $8 per toxic pound-equivalent and $12 per toxic pound-equivalent for the low- and high-end scenarios, respectively. In the low-end scenario, the highest cost per toxic pound-equivalent removed was observed

[2] High-end costs were used in order to provide a conservative estimate of the highest possible costs that could be attributed to the State or federal government.

[3] This apportionment exercise assumes that the State will adopt criteria that are similar to the federal criteria.

for the miscellaneous category ($134 per toxic pound equivalent), which includes industrial categories such as museums, airports, and national security, while the lowest is for POTWs at about $6 per toxic pound-equivalent. In the high-end scenario, the highest cost per toxic pound-equivalent removed was for the metals and transportation category at $122 per toxic pound-equivalent, while the lowest was for POTWs at $9 per toxic pound-equivalent.

POTENTIAL BENEFITS

The benefits analysis for this EA is intended to provide insight into both the types and the potential magnitude of the benefits expected to result from implementation of point source controls associated with the rule. 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. Conceptually, 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).

Benefit Categories Applicable to the Analysis

To implement a benefits analysis, the types or categories of benefits that apply need to be defined. EPA relied on a set of benefits categories that typically applies to changes in the water resource environment. Benefits typically are categorized according to whether or not they involve some form of direct use of, or contact with, the resource. *Use* benefit categories can include both direct and indirect uses of the impacted waters, such as human health and recreational values. Improved environmental quality can be valued by individuals apart from any past, present, or anticipated future use of the resource in question. Such *passive use* (nonuse) values have been categorized in several ways in the economics literature, typically embracing the concepts of existence, bequest, and stewardship.

Ecologic Resources and Toxic Impairment of California Waters

EPA's analysis of benefits (U.S. EPA, 1997) revealed that California is one of the most biologically diverse areas in the world with more unique plants and animals than any other State in the nation. However, the analysis also revealed that, on average, over 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. 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. Current concentrations of toxics in California's aquatic systems may pose risk to resident and migratory biota through direct and indirect pathways of exposure in the surface waters, diets, or sediments. Human health may also be at risk, as evidenced by fish consumption advisories for San Francisco Bay and nine inland waterbodies, including reservoirs, rivers, and creeks in Santa Clara County and the Grassland Area of the Kesterson National Wildlife Refuge in Merced County.

EPA analyzed the extent of toxic impairment of California waters (U.S. EPA, 1997) based on water quality data for the State of California. For that analysis, EPA defined "impaired" waters as those monitored and rated by the State of California as medium or poor quality for at least one toxic pollutant or group of toxic pollutants. EPA assumed for this EA (maybe conservatively) that only half of the waters that were not monitored by the State are impaired in the same proportion as assessed waters. The resulting estimates of toxic impairment of California waters are shown in **Exhibit ES-2**. To the extent that more than half of the unmonitored waters are impaired by toxics, the potential benefits of reducing toxic impairments described below will be understated.

EXHIBIT ES-2. ESTIMATED TOXIC-IMPAIRMENT OF CALIFORNIA WATERS[1]

	Estimated Toxic Impairment (Percent)
Freshwater Fishing	
Lakes and Reservoirs	15%
Ponds	15%
Rivers and Streams	10%
Saltwater Fishing	
San Francisco Bay	69%
Other California Bays	38%
Estuaries	35%
Saline Lakes	52%

[1] "Impaired" waters are defined as those rated by the State of California as medium or poor quality for at least one toxic pollutant or group of pollutants.

Contribution of Point Sources to Toxic-related Impairment

EPA's analysis of benefits (U.S. EPA, 1997) assessed the contribution of point sources to the toxic impairment of California waters based on available information regarding loadings of toxic pollutants from different sources. EPA modified the analysis for this EA to account for a general lack of data. The resulting estimates of the share of toxic-related impairment due to point sources alone are presented in **Exhibit ES-3**, and represent the toxic-weighted average across a range of toxic pollutants. There are a number of uncertainties and limitations in the estimates, and EPA recognizes a need for additional data in this area.

EXHIBIT ES-3. ESTIMATED SHARE OF TOXIC LOADINGS TO CALIFORNIA SURFACE WATERS ATTRIBUTABLE TO POINT SOURCES

	Estimated Share of Toxic 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.

Potential Human Health Benefits

EPA quantified and monetized three categories of potential benefits: (1) human health, (2) recreational angling, and (3) passive use. To calculate human health benefits, EPA assessed the potential risks to San Francisco Bay and freshwater anglers from the consumption of contaminated fish, and the potential reductions in these risks expected to result from implementation of the CTR.

Baseline Health Risks

EPA assessed baseline human health risks (cancer and systemic effects) based on current contaminant levels in fish tissue samples collected from San Francisco Bay and freshwater fisheries throughout California. Average consumers face a baseline excess lifetime cancer risk of 1.8×10^{-4} from consumption of fish from San Francisco Bay, and 1.5×10^{-4} from consumption of freshwater fish, resulting in 5 excess cancer cases per year. (High end consumers face somewhat higher baseline risk levels of 9.2×10^{-4} for San Francisco Bay and 7.6×10^{-4} for freshwaters.) EPA assessed systemic risks by means of a hazard quotient. A hazard quotient of one or greater implies that chronic chemical exposures exceed EPA-established thresholds of toxicity, and are indicative of potential for adverse health effects. Hazard quotients for average and high end consumers in San Francisco Bay and freshwaters exceed one for several pollutants, including PCBs, mercury, and dioxin.

Potential Risk Reductions Due to the Rule

EPA estimated the potential reductions in fish tissue contaminant concentrations, and thus human health risks, that would result from the rule based on: (1) the estimated reduction in loadings for the relevant pollutants contributing to baseline risks, and (2) the estimated share of toxic-weighted loadings to California surface waters attributable to point sources. The reduction in cancer cases and associated monetized benefits are shown in **Exhibit ES-4** (there is no means of monetizing reductions in noncancer health risks). The potential reductions in human health risks are primarily attributable to reduced concentrations of DDT. However, reductions attributable to the rule are small because the estimated loadings reductions for the contaminants responsible for the majority of the baseline risks (PCBs, dioxin, and mercury) are small.

EXHIBIT ES-4. POTENTIAL HUMAN HEALTH BENEFITS TO RECREATIONAL ANGLERS[1]

	Annual Reduction in Cancer Cases	Annual Monetized Benefits (millions of 1996 dollars)
San Francisco Bay	0.0	$0.0
Freshwater Resources	0.0–0.6	$0–$5.3

[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. Range based on estimate of reductions in fish tissue concentration contamination resulting from the rule. Does not include potential reductions in noncancer health risks.

Recreational Angling Benefits

Concerns about the health effects of eating contaminated fish 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 consist of two components: fewer fishing trips are taken because of the health concerns and advisories, and the value of trips that continue to be taken is reduced. EPA estimated the potential value of these components that would result from meeting the water quality standards of the CTR, and than estimated the portion that could be attributable to implementation of point source controls.

Increased Value of Existing Trips

EPA found no available studies of the value to California anglers of reducing toxic contamination of surface waters. However, the potential significance of the contamination problem in terms of how present anglers value the fishery is illustrated by a 1992 study of the Wisconsin Great Lakes open water sport fishery (Lyke, 1993). Lyke estimated the value of the fishery to Great Lakes trout and salmon anglers if it were "completely free of contaminants that may threaten human health" to be between 11 to 33 percent of the current value of the fishery. EPA estimated the current value of California fisheries that are impaired by toxic contamination based on: (1) the estimated number of fishing days in California waters, (2) the estimated toxic impairment of California waters, and (3) the estimated value of a fishing day. Then, EPA increased this baseline value by 11 to 33 percent and multiplied it by the estimated reduction in toxic-weighted loadings and the estimated share of loadings to California surface waters attributable to point sources. The potential benefits that may result from the rule range from $0.6 million to $8.6 million per year.

Value of Increased Participation

Reduced toxic contamination may improve perceptions of water quality and thus have a positive impact on participation. In addition, reduced toxic contamination may increase the stability, resilience, and overall health of numerous ecosystems, which may translate into higher catch rates and increased angling effort. As a result, even if good substitute sites exist for the toxic-affected areas that anglers are aware of, some minimal increase in participation may result from implementation of the CTR on a statewide basis. EPA assumed a 5 percent increase in angler participation (10 percent for San Francisco Bay) would result from meeting the water quality standards of the CTR.

EPA estimated the potential value of increased participation based on (1) the estimated number of fishing days in California waters and the potential increase due to the rule, (2) the estimated toxic impairment of California waters, and (3) the estimated value of a fishing day. Then, EPA multiplied this value by estimated reduction in toxic-weighted loadings and by the estimated share loadings to California surface waters attributable to point sources. The potential benefits that may result from the rule range from $0.3 million to $1.5 million per year ($1996). However, because of the uncertainties inherent in the analysis (e.g., an inability to account for substitute sites), EPA used zero as a lower bound estimate.

Passive Use Benefits

To estimate passive use benefits attributable to the rule, EPA utilized an extensive literature review of the economics literature providing empirical evidence of the use and passive use values associated with improved water quality and fisheries (Fisher and Raucher, 1984). This review indicated that nonuse values have been estimated to be *at least* half as great as recreational values, and concluded that if passive use values were potentially applicable to a policy action, using a 50 percent approximation was, with proper caveating, preferred to omitting passive use values from a benefit-cost analysis. This "rule of thumb" implies that passive use benefits applicable to the CTR may amount to between $0.3 million and $5.0 million per year.

Since the rule of thumb estimate is based on recreational angling values, it likely provides a lower bound on passive use values because it does not address nonanglers. EPA therefore estimated passive use values held by nonangling households, which includes other water recreators such as boaters, and swimmers, as well as nonusers. EPA based this estimate on (1) the estimated number of angling and nonangling households in California, (2) the implied passive use value per household for angling households based on the "rule of thumb", and (3) the relationship between total willingness to pay for users and nonusers, as revealed by the literature. EPA's approach yielded an estimate of passive use values for nonangling households of between $0.6 million and $31.3 million per year.

Summary of Monetized Benefits

A summary of the potential benefits is provided in **Exhibit ES-5**. However, a number of potential or likely benefits that have been omitted from the quantified and monetized estimates. For example, the omission of potential motorized and nonmotorized 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 (notably, swimming). Other of these 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 observation, were also omitted. 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, it is conceivable that these benefits could be appreciable.

EXHIBIT ES-5. SUMMARY OF ANNUAL BENEFITS (MILLIONS OF 1996 DOLLARS)

Benefit Category	Annual Value
Human Health (cancer risk)	
San Francisco Bay	$0.0
Other Saltwater Resources	+
Freshwater Resources	$0.0–5.3
Recreational Angling	
Increased Value of Existing Trips	$0.6–$8.6
Increased Participation [1]	$0.0–$1.5
Passive Use	
Households with Recreational Anglers	$0.3–$5.0
Other Households	$0.6–$31.3
Omitted Benefits [2]	+
Total	$1.5–$51.7

[1] A lower bound of zero is used because of difficulties in accounting for substitute sites at the statewide level.

[2] 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 mentioned.

COMPARISON OF BENEFITS AND COSTS

A direct comparison of the estimated annualized cost of the CTR to the potential annual benefits under Model 2 shows that the benefits range overlaps the range of costs. As shown in **Exhibit ES-6**, annualized costs range from $14.9 million to $86.6 million (1996, first quarter), and the portion of annual benefits that can be monetized amounts to between $1.5 million and $51.7 million. Discounted benefits also overlap discounted costs. Discounting applies a present value social accounting in which the stream of future benefits and costs are discounted to their present values to reflect society's rate of time preference. EPA calculated the streams of discounted benefits and costs assuming discount rates of 3 percent and 7 percent. EPA also considered two different phase-in scenarios for benefits (10-year and 20-year) to account for the potential delay in realizing benefits because many of the pollutants addressed by the CTR are persistent in the environment.

Benefits and costs are of similar magnitude in the comparison of both annual and discounted benefits and costs. However, since EPA used a number of assumptions that may have overstated costs, and omitted several benefits categories, benefits and costs may be more commensurate than shown by **Exhibit ES-6**.

**EXHIBIT ES-6. COMPARISON OF POTENTIAL MONETIZED BENEFITS AND COSTS UNDER MODEL 2
(MILLIONS OF 1996 [FIRST QUARTER] DOLLARS)**

Method	Monetized Benefits Range		Cost Range	
Direct Annual Comparison[1]	\$1.5–\$51.7		\$14.9–\$86.6	
Discounted Benefits and Costs[2]				
Discount Rate	3%	7%	3%	7%
10-Year Phase-In of Benefits	\$23–\$807	\$14–\$473	\$260–\$1,430	\$182–\$996
20-Year Phase-In of Benefits	\$18–\$611	\$10–\$333	\$260–\$1,430	\$182–\$996

[1] These monetized costs and benefits are not directly comparable. Since EPA used a number of assumptions that may have overstated costs (especially at the high-end of the range) and omitted several benefits categories, benefits and costs may be more commensurate than shown in this table.

[2] Present values over 30 years. Reflects capital costs in years 1 and 16, a 7 percent opportunity cost of capital, and O&M and monitoring costs in years 2 through 30. Benefits are phased in proportionately over 10 and 20 years, and have their full value in the remaining years.

EVALUATION OF REGULATORY OPTIONS

Impact of Human Health Risk Level

The proposed CTR criteria for the protection of human health for carcinogenic toxic pollutants are based on an assumed risk equal to one excess cancer case per 1 million individuals (10^{-6} risk level). An alternative analysis was performed to determine the potential impact if the risk level was changed to one excess cancer case per 100,000 individuals (10^{-5} risk level) using the same methodology as the original analysis.

Changes in costs and pollutant loading reductions under both the low- and the high-end scenarios based on lowering the risk level were minimal. The insensitivity of the results to a change in risk level is due to the fact that most of the costs and attendant load reductions are driven by toxic metals, and most carcinogens are toxic organic pollutants, which account for only a small percentage of costs and load reductions under the overall analysis.

Impact of Metals Translators

The criteria for metals in the proposed rule are in the dissolved form. The use of dissolved 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 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 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. 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. A third method is to simply use the total recoverable criteria which 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.

EPA performed a sensitivity analysis to estimate the effect of this third method of translation on total costs and load reductions. Although EPA encourages the use of site-specific translators, some members of the regulated community expressed concern that the State may choose this conservative approach to deriving permit limits, and asked EPA to perform this sensitivity analysis.

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-end scenario are just over $35 million per year, over a six-fold increase over the estimates in the low-end base analysis. Under the high-end scenario, total costs are estimated to be almost $154 million per year, almost double the cost estimates in the base analysis. Potential load reductions are estimated to increase by 0.4 million toxic pounds-equivalent per year (approximately 60 percent) over the low-end base case (0.63 million toxic pound-equivalents per year). Under the high-end scenario, load reductions increase by over 2 million toxic pounds-equivalent per year over the base case (7.02 million toxic pounds-equivalent per year). Using conversion factors as translators would result in higher costs per toxic pounds-equivalent removed than the base analysis.

1. INTRODUCTION

This document has been prepared to support the California Toxics Rule (CTR), a regulatory action that proposes, for the State of California, numeric water quality criteria for priority toxic pollutants necessary in the State of California to meet the requirements of section 303(c)(2)(B) of the Clean Water Act (CWA) (33 U.S.C. 1251 et seq.).

1.1 PURPOSE

Under Executive Order (E.O.) 12866 (58 FR 51735, October 4, 1993), the Agency must determine whether the regulatory action is "significant" and therefore subject to the requirements of the E.O., i.e., drafting an Economic Analysis (EA) and review by the Office of Management and Budget (OMB). Under section 3(f), the order defines "significant" as those actions likely to lead to a rule: (1) 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"); (2) creating serious inconsistency or otherwise interfering with an action taken or planned by another agency; (3) materially altering the budgetary impacts of entitlement, grants, user, fees, or loan programs; or (4) raising novel legal or policy issues arising out of legal mandates, the President's priorities, or the principles set forth in the order.

Pursuant to the terms of the order, EPA has determined that this proposed rule is not "significant". The proposed CTR establishes ambient water quality criteria which, by themselves, do not directly impose economic impacts. When these criteria are combined with the State-adopted designated uses for inland surface waters, enclosed bays and estuaries, water quality standards will be created. EPA acknowledges that there may be a cost to some dischargers for complying with new water quality standards after those standards are translated into specific NPDES permit limits by the State. Consistent with the intent of E.O. 12866, EPA prepared an Economic Analysis (EA). Since the State has significant flexibility and discretion in how it chooses to implement standards within the NPDES permit program, the EA by necessity includes many assumptions about how the State will implement the water quality standards. These assumptions are based on a combination of EPA guidance and current permit conditions for the facilities examined in this analysis. (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.) A more precise measure of costs and benefits may not be known until the State adopts its implementation provisions. To account for the uncertainty of these assumptions, this analysis estimates a wide range of costs and benefits. By completing the EA, EPA intends to inform the public about how entities might be affected by implementation of CTR-based water quality standards in the NPDES permit program.

This document identifies the need for the proposed regulation, assesses potential costs and benefits, and analyzes alternative approaches. Wherever possible, the costs and benefits are expressed in monetary terms.

1.2 REPORT ORGANIZATION

The remainder of this 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 proposed rule.

Chapter 3 ("Scope of Economic Analysis") describes the baselines chosen for analysis and the specific types of costs and benefits that were estimated.

Chapter 4 ("Analysis of Costs and Cost-Effectiveness") presents: (1) the potential costs of State implementation of water quality criteria, and (2) an assessment of the cost-effectiveness of potential State implementation.

Chapter 5 ("The Benefits Associated with the CTR: Methods and Concepts") describes the approach used to estimate the potential benefits of State implementation of water quality criteria.

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 three categories of quantified and monetized benefits resulting from State implementation of the CTR.

Chapter 9 ("Comparison of Potential Benefits to Costs") compares costs and benefits estimated in the previous chapters.

Chapter 10 ("Alternatives Analysis") describes alternative approaches considered in setting and implementing water quality criteria in California.

2. NEED FOR THE REGULATION

This chapter discusses: (1) the statutory requirements that dictate the necessity for the proposed rule; and (2) the environmental factors that indicate that water quality criteria for toxic pollutants are necessary for California inland surface waters and enclosed bays and estuaries.

2.1 STATUTORY REQUIREMENT

This rule proposes, for the State of California, numeric water quality criteria for priority toxic pollutants necessary to fulfill the requirements of section 303(c)(2)(B) of the Clean Water Act (CWA) (33 U.S.C. 1251 et seq.).

EPA is proposing this rule based on the Administrator's determination that criteria are necessary in the State of California to meet the requirements of CWA section 303(c)(2)(B). This section of the CWA requires states to adopt numeric water quality criteria for priority toxic pollutants for which EPA has issued CWA section 304(a) criteria guidance and whose presence could reasonably be expected to interfere with designated uses. Priority toxic pollutants are identified in CWA section 307(a).

EPA is proposing this rule to fill a gap in California water quality standards. This gap is the result of litigation by several municipal entities and one industry in California who sued the California State Water Resources Control Board (SWRCB) over whether the SWRCB's statewide water quality control plans for inland surface waters and for enclosed bays and estuaries were adopted in compliance with State law. The California Superior Court for the County of Sacramento issued its final decision in favor of the plaintiffs in March of 1994. In July of 1994, the Court ordered the SWRCB to rescind the two water quality control plans, and the SWRCB formally did so in September of 1994. The SWRCB's water quality control 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 proposed federal criteria take effect, they will create legally applicable water quality standards in the State of California for inland surface waters, enclosed bays and estuaries for all purposes and programs under the CWA.

This proposed rule does not change or supersede any criteria previously promulgated for the State of California in the National Toxics Rule, as amended (Water Quality Standards; Establishment of Numeric Criteria for Priority Toxic Pollutants, 57 FR 60848, December 22, 1992, as amended by Stay of federal Water Quality Criteria for Metals; Water Quality Standards; Establishment of Numeric Criteria for Priority Toxic Pollutants; States' Compliance – Revision of Metals Criteria; Final Rules, 60 FR 22228, May 4, 1995 [the National Toxics Rule (NTR), as amended]).

2.2 AN OVERVIEW OF ENVIRONMENTAL CONCERNS

This rule, which will establish federal ambient water quality criteria for certain priority toxic pollutants for the State of California, is necessary to address 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 on California water bodies indicate that elevated levels of toxic pollutants exist in fish tissue; these discoveries have resulted in fishing advisories or bans.

This analysis indicates that toxic pollutants covered by the proposed rule impair many of California's surface water resources. ("Impaired" waters are defined as those that have been assessed and are rated by the State of California as medium or poor water quality for at least one toxic water quality pollutant or groups of pollutants. "Impaired" is further defined 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, over 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 waterbodies, 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 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 waterbody types, particularly estuaries and wetlands.

- Currently, there are 12 fish consumption health advisories in waters covered by the CTR (nine inland waterbodies and three 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 Grasslands area, Suisun Bay, San Pablo Bay, and San Francisco Bay based on elevated selenium levels in duck, greater and lesser scaup, and scoters.

Water quality standards for toxic pollutants are important to State and EPA efforts to address these problems. Clearly established water quality goals enhance the effectiveness of many of the State's and EPA's water programs including permitting, enforcement, coastal water quality improvement, fish tissue quality protection, nonpoint source controls, drinking water quality protection, and ecological protection. Numeric criteria for toxic pollutants allow the State and EPA to evaluate the adequacy of existing and potential control measures to protect aquatic ecosystems and human health. Numeric criteria also provide a legal basis for including water quality-based effluent limitations in NPDES permits to control toxic pollutant discharges. Congress recognized these issues when it enacted section 303(c)(2)(B) to the CWA. In order to protect human health, aquatic ecosystems, and successfully implement toxic pollutant controls, EPA believes that actions which are available to the Agency must be taken to ensure that all necessary numeric criteria for priority toxic pollutants are established in a timely manner.

As States, including California, and EPA continue the transition from an era of primarily technology-based controls to an era in which technology-based controls are integrated with water quality-based controls, it is important that EPA ensures timely compliance with CWA requirements. An active federal role is essential to assist California in adopting a complete set of toxic pollutant criteria as part of its pollution control programs.

3. SCOPE OF ECONOMIC ANALYSIS

This chapter describes the baselines chosen for analysis and discusses the specific types of costs and benefits that would result from State implementation of the proposed California Toxics Rule (CTR).

3.1 DESCRIPTION OF BASELINES USED FOR ANALYSIS

In order to estimate costs and benefits, an analysis must have established an appropriate baseline. The baseline is the starting point for measuring incremental costs and benefits of a regulation. The baseline is determined by assessing what would occur in the absence of a regulation. EPA estimated the incremental costs and benefits of potential implementation of the CTR using two different models to account for the uncertainty of how the might implement water quality criteria through NPDES permits in the absence of the proposed CTR.

3.1.1 Model 1: Negligible Costs and Benefits

The baseline used in this model assumes that the State will rely on its narrative toxicity criteria and implementation of the criteria using "best professional judgment"(BPJ) to place numeric water quality-based effluent limits for toxic pollutants into 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 criterion. The basis for such limits could include EPA's 304(a) criteria documents or other appropriate scientific information. Therefore, under one scenario, permit writers could comply with the permitting regulations by basing numeric effluent limits on narrative criteria that incorporate the latest scientific information. This approach would likely result in permit limits that are nearly identical to those that would result from implementation of the CTR criteria, which are also based on the latest available scientific information.

Thus, under this scenario, the costs and benefits of the CTR would be negligible since implementation of permits under the CTR would not differ significantly from State implementation of permits under current law.

3.1.2 Model 2: Most-Likely Estimate of Costs and Benefits

The baseline used in Model 2 assumes that in the absence of the rule, current permit requirements and current effluent concentrations would continue in the future. The baseline generally uses current permit limits to develop a high scenario cost estimate and current effluent concentrations to develop a low scenario cost estimate.[4] Using this baseline, the incremental annual costs incurred owing to implementation of the CTR were estimated to range from $15 to $87 million, and corresponding annual monetized benefits were estimated to range from $1.5 to $51.7 million; additional non-monetized benefits were also expected to accrue.

Permits in the State are written and issued by nine autonomous Regional Water Quality Control Boards. Therefore the derivation of effluent limits varies from Board to Board. Some of the effluent limits used in this baseline were derived from criteria in California's statewide plans, since some permits were issued during the time the plans were in effect. However, since the plans were revoked, permit writers no longer use the criteria contained in the plans. Regional Boards have been using a variety of methods to derive numeric permit limits including the use of numeric criteria adopted in the mid-1980s and drinking water maximum contaminant limits (MCLs) developed under the Safe

[4] For a more detailed explanation of how the baseline was established for the high and low scenarios see Chapter 4.

Drinking Water Act. Given this diversity, this baseline best represents the current variable situation in California, and will yield a comparative result that represents actual incremental costs and benefits that will occur when the CTR is implemented. The methodology described in Chapter 4 uses Model 2 as the operating assumption for the estimation of costs.

3.2 POINT SOURCE COSTS AND BENEFITS

3.2.1 Rationale

This Economic Analysis (EA) estimates costs to point source facilities that are typically subject to numeric water quality-based effluent limits (WQBELs) calculated using CTR criteria and the benefits attributable to those point sources. The point sources included in this study only include those that discharge to waters of the United States which also discharge to California inland surface waters and enclosed bays and estuaries. Under the Clean Water Act (CWA), EPA has direct authority regarding permits issued under the National Pollutant Discharge Elimination System (NPDES). EPA did not calculate costs for any program for which it does not have enforceable authority, such as agricultural nonpoint sources. In addition, EPA did not calculate costs for NPDES sources which are not typically subject to numeric WQBELs, including sources required to hold NPDES stormwater permits and other wet weather dischargers.

EPA predicted how water quality standards based on CTR criteria may be implemented by the State through numeric effluent limits for NPDES facilities and attempted to predict the actions these facilities may need to take in order to comply with effluent limits based on the new criteria. EPA envisions that some of these costs may involve efforts to defer new effluent limits until studies are undertaken to allocate pollutant reductions throughout the watershed and, where appropriate, EPA has included the costs of these studies in the analysis. Although EPA has focused on calculating costs to NPDES permitted facilities, EPA believes that a comprehensive watershed approach that addresses all significant sources of problem pollutants will often present more cost-effective approaches. However, the total costs of actions necessary to implement a watershed approach in a given area can only be adequately estimated 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 waterbodies. Accordingly, the benefits estimated include only those that may occur as a result of loadings reductions from NPDES point sources typically subject to numeric WQBELs.

The methodology and results of this cost analysis are described further in Chapter 4. The methodology and results of this benefits analysis are described further in Chapters 5, 6, 7, and 8.

3.3 NONPOINT SOURCES AND WET WEATHER DISCHARGES

3.3.1 Potential Impacts Unknown

The proposed rule does not have a direct effect on sources not permitted under the NPDES program (e.g., nonpoint sources) or NPDES sources not typically subject to numeric water quality-based effluent limits (e.g., wet weather discharges). Therefore, these sources were not analyzed quantitatively in this EA. Any potential indirect effect on these sources is unknown at this time.[5] These nonpoint sources and wet weather discharges can include, but are not

[5] Agricultural nonpoint source discharges and wet weather discharges are technically difficult to model and evaluate for costing purposes because they are intermittent, highly variable, and occur under different hydrologic or climatic conditions than continuous discharges from industrial and municipal facilities, which are evaluated under critical low flow or drought conditions. Thus, the evaluation of agricultural nonpoint source discharges and storm

limited to, runoff from farms, urban areas, and abandoned mines, and contaminated sediment. The State may ask or require these sources to implement best management practices or 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. Wet weather discharge problems, nonpoint source problems, and the current strategies under way to control them are discussed below. Many of the programs developed to control wet-weather discharges and nonpoint sources are already in place. Costs due to these programs have already been incurred or will soon be incurred owing to existing federal, State, and local environmental programs.

The following is an attempt to qualitatively describe the types of nonpoint sources and wet weather discharges that contribute to toxic impairment of waterbodies and those programs that need to continue if all California waters are to ultimately meet water quality standards.

The categories of sources that may not always be subject to numeric effluent limits and which are likely to contribute to toxic impairment of water bodies include:

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 washed away by 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 waterbodies, or reintroduced to the local irrigation system.

As a result of existing federal and State law, much research and time has been spent attempting to alleviate the difficult problems caused by agricultural runoff. Unlike point source discharge, 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 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. An example of existing efforts to reduce agricultural runoff is a Central Valley Regional Water Quality Control Board program which has developed a total maximum monthly load (TMML) allocation for selenium in the San Joaquin River which has identified the source of selenium and determined limits throughout the watershed to reduce the load.

2) **Inactive and Abandoned Mines**

California has over 15,000 inactive, abandoned mines. Costs for mines (both active and inactive) with major NPDES permits were estimated and are included in the point source cost analysis. However, only five 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 are not currently permitted. Acid mine drainage results in the discharge of metals such as cadmium, copper, lead, mercury, and zinc. Clean-up technology for mine drainage can be costly and labor intensive. Treatment methods vary depending on the site and extent of pollution. 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 under water to eliminate reaction with air, impounding mine

water discharges and their effects on the environment are highly site-specific and data intensive.

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 involves control of the mine drainage before it enters surface waters. Treatment techniques include chemical precipitation, ion exchange, construction of wetlands, and evaporating mine discharge in surface impoundments.[6]

Efforts are already under way to clean up mine sites under existing State and Clean Water Act requirements (stormwater regulations) and under Superfund. However, in order to reach full compliance with water quality standards, additional assessment and treatment may be necessary for some California waterbodies.

3) **Urban Runoff**

Most cities in California have a separate storm drainage system for diverting storm water to prevent flooding. When rainfall flushes out pollutants, such as toxic metals and pesticides, that accumulate on the ground, stormwater drains can carry harmful amounts of these pollutants into rivers, lakes, and bays. Urban runoff in California has been shown to be a significant contributor to water quality problems. Urban runoff is regulated as an NPDES point source for large towns and as a nonpoint source for medium and small towns. Approaches to control of stormwater pollution stress BMPs. Programs designed to use BMPs often emphasize pollution prevention (e.g., street cleaning, 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 stormwater permits and through nonpoint source planning. For example, under its existing NPDES stormwater 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.[7]

State Implementation of Nonpoint Source Management Program

Since some of the sources discussed above are exempt from federal permitting requirements, the State must develop alternative strategies and controls to protect or restore water quality.

The following description of California programs to control nonpoint sources was extracted from a State of California document entitled, *Nonpoint Source Management Plan, November 1988*. The State of California uses three general management approaches to address nonpoint source problems. In most cases, the Regional Water Quality Control Boards decide the mix of options that will be used to address any given nonpoint control problem.

1. Voluntary Implementation of Best Management Practices

Property owners or managers may voluntarily implement BMPs. BMPs include but are not limited to structural and nonstructural controls and operation and maintenance procedures. BMPs could be applied before, during, or after pollution producing activities to reduce or eliminate the introduction of pollutants into receiving waters. A voluntary approach would take advantage of the expertise and incentives offered by a variety of existing State and federal

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[6] *California's Rivers and Streams: Working Toward Solutions*, California State Water Resources Control Board for California Rivers and Streams, January 1995, pp. 1–4.

[7] California 305(b) Report on Water Quality, State Water Resources Control Board, August 1996, p.4.

programs. Lead agencies for these programs include the U.S. Soil Conservation Service, the U.S. Agricultural Soil Stabilization and Conservation Service, and the Resource Conservation Districts.

2. Regulatory-Based Encouragement of Best Management Practices

There are two ways in which Regional Boards can use their regulatory authorities to encourage implementation of BMPs.

First, Regional Boards may encourage BMPs by waiving adoption of waste discharge requirements on condition that dischargers comply with BMPs.

Alternatively, the State Board and the Regional Boards may enforce BMPs indirectly by entering into management agency agreements (MAAs) with other agencies which have the authority to enforce. Such authority derives either from the agency's regulatory authority or its management responsibility for publicly owned or controlled land. MAAs will include (or reference) specific, acceptable BMPs and their means of implementation.

3. Waste Discharge Limitations

Regional Boards can adopt and enforce requirements on the nature of any proposed or existing waste, including discharges from nonpoint sources. Although Regional Boards are precluded from specifying the manner of compliance with waste discharge limitations, in appropriate cases limitations may be established at a level which, in practice, requires implementation of BMPs.[8]

3.3.2 Benefits

The analysis of costs conducted for this EA encompasses only costs incurred by point source dischargers as part of the State of California's implementation of the CTR. Similarly, the potential benefits estimated here represent only the potential benefits from implementing point source controls.[9] Estimation of the point source portion of "total" benefits was accomplished by first evaluating baseline water resource value and the potential increase in value that would result from meeting water quality standards (e.g., removing toxic-related use impairments). Then, a portion of this value was attributed to the point source controls included in the cost analysis based on the estimated reduction in toxic-weighted pollutant loadings and the estimated contribution of point source loadings to the toxic-related use impairments. Much of the benefits work is drawn directly from the original benefits report (U.S. EPA, 1997). The methodology used to estimate potential benefits is described in more detail in Chapters 5 through 8.

[8] Violations of State waste discharge limitations are not subject to federally enforceable actions.

[9] In comparison, the original benefits report (U.S. EPA, 1997) presented an estimate of the "total" benefits of meeting the water quality standards embodied in the CTR. Since controlling point sources alone may not in all cases be sufficient to meet water quality standards, "total" benefits represent potential benefits attributable to both point source and nonpoint source controls.

4. ANALYSIS OF COSTS AND COST-EFFECTIVENESS

This chapter presents the analysis of potential costs resulting from implementation of the California Toxics Rule (CTR) to point sources that discharge to inland surface waters, enclosed bays, and estuaries. The cost estimates were derived from SAIC (1997). A description of the methodology and the limitations of the cost analysis is presented in Section 4.1. The results of the compliance analysis, including costs, pollutant loading reductions, and cost-effectiveness by industrial sector, are presented in Section 4.2.

4.1 OVERVIEW OF METHODOLOGY AND LIMITATIONS OF ANALYSIS

To estimate potential costs and pollutant load reductions attributable to implementation of the proposed CTR, EPA developed detailed cost estimates for a selected subset (a sample) of facilities from the point source dischargers to California's inland waters, enclosed bays, and estuaries, and then used the sample results to extrapolate to the universe of potentially affected facilities. The population of National Pollutant Discharge Elimination System (NPDES)-permitted facilities that discharge into California's enclosed bays, estuaries, and inland surface waters includes 184 major dischargers and 1,057 minor dischargers. For the analysis of costs, 17 major dischargers and 3 minor dischargers were selected as sample facilities to represent the range of discharger categories and geographic distribution of all permitted dischargers to enclosed bays, estuaries, and inland surface waters.

Some simplifying assumptions were made to facilitate analysis and to overcome data limitations, where necessary. These assumptions were designed to be "conservative," that is, to err on the side of more stringent and costly controls, rather than less. **Exhibit 4-1** presents these technical assumptions and the potential impact on the analysis.

The actual impact of the CTR after water quality standards are translated into specific NPDES permit limits will vary depending upon the procedures that will be used to implement the CTR. These procedures typically specify the methods to determine the need for water quality-based effluent limits (WQBELs) and, if WQBELs are required, how to derive WQBELs from applicable water quality criteria. The implementation procedures used to derive WQBELs for this study were based on the methods recommended in the EPA *Technical Support Document for Water Quality-based Toxics Control* (or TSD) (U.S. EPA 1991).

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

A projected effluent quality (PEQ) was calculated and compared to the projected CTR-based WQBEL. A PEQ is an effluent value statistically adjusted for uncertainty to estimate a maximum value. The PEQ for each selected pollutant was compared to the most stringent projected CTR-based WQBEL (the most stringent WQBEL based on protection of aquatic life and human health). If the PEQ exceeded the projected CTR-based WQBEL, a reasonable potential existed to exceed a CTR-based WQBEL. Pollutants with a reasonable potential to exceed then were analyzed to determine potential costs to achieve the CTR-based WQBEL.

EXHIBIT 4-1. SUMMARY OF MAJOR TECHNICAL ASSUMPTIONS USED TO DERIVE POTENTIAL COSTS ASSOCIATED WITH IMPLEMENTATION OF THE PROPOSED CALIFORNIA TOXICS RULE

Assumption	Potential Impact on Analysis
Existing facilities that contain effluent limits for toxic pollutants were selected as representative facilities and used as the basis for extrapolation to the universe of potentially affected facilities	Tends to bias the sample in terms of possibly overstating the number and types of pollutants that may require control; may tend to overestimate the need for WQBELs and costs when extrapolated to the universe
The methods used to determine reasonable potential and calculate CTR-based WQBELs were 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 overstate costs; if State methods are comparable to the TSD, costs are neither over- nor understated
Use of human health criteria based on the consumption of water and organisms (for fresh water discharges only)	Applies the most stringent criteria for human health protection; tends to result in more stringent effluent limits; tends to overestimate potential cost
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
Effluent flow used in calculating CTR effluent limits was assumed to be the maximum treatment plant design flow	Restricts dilution available for the discharge; tends to make resultant water quality-based effluent limits more stringent; tends to overestimate the costs required to achieve CTR-based limits
In the absence of data or information related to critical low flow for the receiving water, zero dilution was assumed	Restricts dilution available for the discharge; tends to make resultant water quality-based effluent limits more stringent; tends to overestimate the costs required to achieve CTR-based limits
The highest reported ambient receiving water concentration was used to represent the background concentration when calculating CTR-based WQBELs In the absence of ambient receiving water data, zero was used as the background concentration	Using the highest reported value 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 Assuming zero in the absence of background data allows the discharger a larger portion of the assimilative capacity of the receiving water; tends to underestimate costs
For discharges to estuaries and enclosed bays, derived metals translators using stream (as opposed to lake) partitioning coefficients; applied the lowest theoretical partitioning factor for metals without empirically derived partitioning coefficients	Results in lower translators and more stringent criteria; tends to overestimate costs
The maximum pollutant effluent concentrations observed during the monitoring period were 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
The existing permit limit, or maximum pollutant effluent concentration in the absence of a permit limit, was used for estimating costs if CTR-based WQBELs were exceeded (High-end 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
Capital costs were amortized over 10 years	The useful life of most equipment currently is more than 10 years; tends to overestimate the annual costs to a facility

4.1.2 Method for Estimating Potential Costs

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 of the State's approach to implementation at this time, EPA developed a range of costs to represent the potential range of impact based on certain implementation assumptions.

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

- If the CTR-based permit limit was more stringent than existing effluent concentrations,[10] then costs were estimated for the incremental pollutant reductions required to achieve the CTR-based limit.

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

- If monitoring data were available, but all values were reported as below analytical detection levels, it was assumed that no costs would be incurred.

- If the estimated annualized cost for removal of a pollutant exceeded $200 per toxic pound-equivalent then it was assumed that dischargers would explore the use of other remedies or controls. When it was assumed that facilities would pursue alternative regulatory approaches, no treatment cost was estimated for a facility.

The high-end cost scenario was developed to reflect a higher baseline loadings estimate and a less flexible State implementation approach than the low-end scenario. The assumptions used for the high-end scenario resulted 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 (as compared to the low-end scenario). The following significant assumptions were used for the high-end analysis:

- If the CTR-based permit limit was more stringent than the existing permit limit (or detectable effluent concentration in the absence of a permit limit), then costs were incurred for the incremental pollutant reductions between the current permit limit and the CTR-based limit.

- If only monitoring data was available (i.e., no permit limit), but all values were reported as below analytical detection levels, it was assumed that no costs would be incurred.

- Acknowledging that opportunities for the use of alternative regulatory approaches may have been limited depending upon the particular circumstances for a "facility," costs were also estimated under a higher cost

[10] If existing effluent concentration exceeded the existing permit limit, the permit limit rather than the existing effluent concentration was compared to the CTR-based limit to estimate costs. This method prevents taking credit for costs attributable to correcting current violations of existing State permit limits.

scenario that assumed alternative regulatory approaches would be granted only when the cost for the particular "category of dischargers" exceeded a cost trigger. Particularly, if the estimated annualized cost for a "category of dischargers" exceeded $500 per toxic pounds-equivalent, then it was assumed that dischargers within the "category" would pursue alternative regulatory approaches.

Prior to estimating compliance costs, an engineering analysis of how each sample facility could comply with the CTR-based effluent limitations was performed. The costs were then estimated based on the decisions and assumptions made in the analysis. A costing decision matrix was developed and was used for each sample facility 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. Specific rules were established in the matrix to provide reviewing engineers with guidance in consistently selecting options.

Under the decision matrix, costs for minor treatment plant operation and facility changes were considered first. Minor, low-cost modification or adjustment of existing treatment was determined to be feasible where literature indicated that the existing treatment process could achieve the revised WQBEL and where the additional pollutant reduction was relatively small (e.g., 10 to 25 percent of current discharge levels).

Where it was not technically feasible to simply adjust existing operations, the next most attractive control strategy was determined to be waste minimization/pollution prevention controls. However, costs for these controls were estimated only where they were considered feasible based on the reviewing engineer's understanding of the process(es) at a facility. The practicality of techniques was determined based on several "rules of thumb" established in the decision matrix. Decision considerations included the level of pollutant reduction achievable through waste minimization/pollution prevention techniques, 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 best professional judgment using general knowledge of industrial and municipal operations.

If waste minimization/pollution prevention alone was deemed not feasible to reduce pollutant levels to those needed to comply with the CTR-based WQBELs, as calculated in this analysis, a combination of waste minimization/pollution prevention, simple treatment, and/or process optimization was considered. If these relatively low-cost controls could not achieve the CTR-based WQBELs, more expensive controls (e.g., end-of-pipe treatment) were considered.

Development of end-of-pipe treatment cost estimates began with a review of the existing treatment systems at each facility. Decisions to add new treatment systems or to supplement existing treatment systems were based on this initial evaluation. For determining the need for additional or supplemental treatment, sources of performance information included the EPA Office of Research and Development (ORD), Risk Reduction Engineering Laboratory's "RREL Treatability Database" (Version 4.0). 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, unit processes that would achieve compliance with the CTR-based effluent limits were chosen using the same documentation.

Following the calculation of end-of-pipe treatment costs, the relationship between the cost of adding the treatment and other types of remedies or controls was considered. Specifically, if the estimated annualized cost for removing a pollutant exceeded a cost trigger (i.e., $200 per toxic pounds-equivalent for a facility under the low-end, and $500 per toxic-pounds-equivalent for a category of dischargers under the high-end scenario), it was assumed that dischargers would explore the use of alternative regulatory approaches to comply with CTR-based effluent limits. When it was assumed that facilities would pursue an alternative regulatory approach, no treatment cost was estimated for a facility. In addition, pollutant load reductions were not calculated or credited for any pollutant for which an alternative was assumed.

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.

4.1.3 Method for Estimating Pollutant Load Reductions

Pollutant loading reductions were calculated by EPA using major municipal and industrial sample facility data to indicate the decrease in pollutant discharge owing to more stringent WQBELs resulting from the State implementing the CTR. In order to factor in the uncertainty related to estimating the incremental costs as well as pollutant load reductions, EPA calculated pollutant load reductions under both a low- and high-end scenario.

Low-end Scenario

Under the low-end scenario, pollutant load reductions were taken for the increment between the maximum effluent concentrations[11] (which serves as the baseline) and the CTR-based WQBELs. If no monitoring data were provided for a pollutant, the pollutant load reduction was not calculated.

Low-end pollutant reductions based on compliance with CTR-based WQBELs were determined by first calculating the difference between the baseline value and the CTR-based effluent limitation. Several assumptions were made in determining the reduction for a pollutant:

- If the difference between the baseline value and the CTR limitation was negative, zero reduction was assumed.

- If the existing effluent concentration was above the method detection level (MDL), but the CTR 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 limit for calculating pollutant load reductions.

- For purposes of calculating a baseline value, it was assumed that facilities were discharging at the maximum reported effluent concentration. If facilities were discharging at levels below this maximum concentration, this assumption may have overestimated pollutant reductions.

High-end Scenario

[11] If the effluent concentration exceeded the existing permit limit, the permit limit rather than the existing effluent concentration was compared to the CTR-based limit to estimate load reduction. This method prevented taking credit for pollutant removals attributable to correcting current violations of existing State permit limits.

Under the high-end scenario, pollutant loading reductions were calculated by finding the difference between the existing permit limitation (considered the baseline value) and the CTR-based effluent limitation.[12] In the absence of a permit limit for a pollutant, the maximum effluent concentration was used as the surrogate for the permit limit.

High-end pollutant reductions based on implementing CTR-based WQBELS were determined by first calculating the difference between the baseline value and the CTR-based effluent limitation. Several assumptions were made in determining the reduction for a pollutant:

- If the difference between the baseline value (existing permit limit) 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.

- For purposes of calculating a baseline value, it was assumed that facilities were discharging at the level of the existing permit limitation. If facilities were discharging at levels below the permit limitation, this assumption may overestimate pollutant reductions.

Estimating Annual Baseline Loads and Pollutant Load Reductions

To determine the annual baseline load for both the low- and high-end scenarios, the baseline value, expressed in concentration units (micrograms per liter), was multiplied 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.

To determine the pollutant loading reduction for a facility, 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) was converted 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. Annual pollutant loading reductions were calculated for each of the pollutants analyzed at each sample facility for which costs were estimated.

Extrapolation of facility-specific (cost-specific extrapolations were conducted similarly) baseline pollutant reductions to the universe of facilities in the State (discharging to inland surface waters, enclosed bays, and estuarine waters) was conducted by averaging the pollutant loading reduction across all facilities in each discharge category and then multiplying by the number of facilities in the category.

Toxic Weighting of Baseline Loads and Pollutant Reductions

[12] Using the permit limitations of a facility as the baseline for pollutant loadings may tend to overstate pollutant removals since some facilities may be discharging at concentrations below permitted amounts.

Baseline and pollutant loading reductions were weighted using EPA toxic weights. Toxic weight factors were derived for toxic pollutants primarily from EPA's chronic freshwater aquatic life criteria and toxicity values. However, EPA human health criterion also was used in cases where a human health criterion had been established for the consumption of fish. Generally, toxic weights were derived by EPA through standardizing these criteria using copper as the standard pollutant (the original EPA criterion for copper, 5.6 ug/L, was used as the water quality criterion and the standardization factor).

Toxic weights for pollutants were taken from the *Assessment of Compliance Costs Resulting from Implementation of the Final Great Lakes Water Quality Guidance* (SAIC, 1995). The toxic weights used in the Great Lakes analysis represent toxic weights calculated by EPA's Office of Science and Technology (OST) in 1988 using pollutant criteria that have been used in various EPA regulatory efforts.[13] Toxic weights for the pollutants evaluated in this study are listed in **Exhibit 4-2**.

[13] National water quality criteria have changed over the years, resulting in corresponding changes in toxic weights. Also, because of California point source discharges to both freshwater and saltwater bodies of water, two different sets of toxic weights would need to be covered. To maintain a general level of consistency between the CTR and previous rules, this study used previously calculated toxic weights.

EXHIBIT 4-2. U.S. ENVIRONMENTAL PROTECTION AGENCY OFFICE OF SCIENCE AND TECHNOLOGY TOXIC WEIGHTS FOR POLLUTANTS ANALYZED

Pollutant	Toxic Weight
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
Zinc	0.051
1,2-Dichlorobenzene	0.011
1,3-Dichlorobenzene*	1
1,4-Dichlorobenzene*	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
beta-BHC	100
Bromoform*	1
Chlordane	2,300
Chlorodibromomethane*	1
Chloroform	0.0021
Dichlorobromomethane*	1
Dieldrin	57,000
Endosulfan	100
Endrin	98
Fluoranthene	0.92
gamma-BHC	70
Heptachlor	4,100
Heptachlor Epoxide*	1
Hexachlorobenzene	720
Methylene Chloride	0.00042
Polychlorinated Biphenyls (PCBs)	7,490
Pentachlorophenol	0.5
Phenol	0.028
TCDD Equivalents	4.20e+08
Toluene	0.0056

*Value was not provided in the source document; a toxic weight of 1 was assumed.

Source: EPA/OST 1988 Cost Effectiveness Criteria and Weights

To calculate toxic pounds-equivalent for each pollutant, the pollutant loading reduction extrapolated to the universe of facilities in the State was multiplied by the appropriate toxic weight for that pollutant. Toxic pounds-equivalent were determined for each pollutant for the baseline pollutant loadings, as well as the pollutant reductions based on low- and high-end scenarios.

Determining Cost-effectiveness

The determination of cost-effectiveness, defined as the incremental annualized cost of a pollution control option per incremental pound-equivalent of pollutant removed by that control option, was based on the methodology EPA used in developing National effluent guidelines limitations and standards under the CWA. The pollutant limits used in permits based on the National effluent guidelines program are established on the best available technology that is economically achievable. However, permit limits for pollutants based on water quality standards are founded on the water quality criteria required to protect the receiving water for a given use, such as fishable/swimmable. For the CTR, cost-effectiveness values were calculated by dividing the total estimated annual costs by the projected annual toxic-weighted pollutant reductions attributable to implementation of the water quality criteria established in the CTR. The cost-effectiveness then estimated the "dollar-per-toxic-pound-removed" resulting from implementing the CTR.

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, it was assumed that many POTWs will select the option of controlling discharges to their collection system as a cost-effective means to comply with permit limits. If POTWs were to select this method of control, the dischargers to the POTWs would be affected. Therefore, an estimate of the potential costs to dischargers to POTWs (i.e., indirect dischargers) was developed.

EPA's estimate was based in part on 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 (MAS) for copper and nickel. These mass audit studies estimated the total costs of implementing various combinations of copper and nickel reduction projects (see City of San Jose, 1994; EOA, 1994).

For this analysis, pretreatment programs administered by other sample POTWs to reduce pollutants were assumed to result in per discharger investment costs and O&M costs equal to the average per-discharger capital and O&M costs for the combined copper and nickel reduction programs planned at the San Jose-Santa Clara and Sunnyvale POTWs. Average per-facility investment costs for industrial participants were estimated from the San Jose-Santa Clara and Sunnyvale MASs for copper and nickel pollution prevention projects. The average cost per indirect discharger was estimated to be $61,526 or $15,000 per year at an interest rate of 7 percent and conservatively annualized over a period of 5 years. The total annual costs to the indirect discharger population in California then were estimated by multiplying the annualized cost ($15,000) by the total number of potentially affected indirect dischargers.

In the *Assessment of Compliance Costs Resulting from Implementation of the Final Great Lakes Water Quality Guidance* (SAIC, 1995), between 10 and 30 percent of the significant industrial user (SIU) population within the Great Lakes Basin would be required to implement control measures to comply with control requirements. The Great Lakes estimates were based on analysis of indirect dischargers at nine industrialized POTWs. The results showed that between 8 and 44 percent of indirect dischargers could be affected by new permit limits on POTWs and concluded that a range of between 10 and 30 percent was a reasonable measure of potentially affected SIUs.

For purposes of this analysis, the same range of affected indirect dischargers was assumed (between 10 and 30 percent). It was assumed that indirect discharger costs could be highest under the low-end scenario, where more implementation flexibility was assumed for direct discharges and there was less dependence upon the use of additional treatment systems. Therefore, 30 percent of all SIUs were assumed to be affected under the low-end cost scenario. Under the high-end scenario, where less implementation flexibility was assumed and the use of additional treatment systems or components was more likely for direct discharges, it was assumed that 10 percent of all SIUs could be affected.

4.2 POTENTIAL COSTS

The cost of the implementation of the CTR will vary depending upon which baseline is used. As described in Chapter 3, Model 1 assumed that in the absence of the CTR, the State of California would impose water quality standards based on the narrative criteria that would be essentially equivalent to numeric water quality criteria proposed in the CTR. Therefore the use of Model 1 resulted in negligible costs.

Under Model 2, the potential annual cost for implementing the CTR was estimated to range from about $15 million to $87 million for the low- and high-end scenarios, respectively. As shown in **Exhibit 4-3**, direct and indirect dischargers would share the potential cost burden in the low-end scenario. However, under the high-end scenario, direct dischargers were expected to incur most of the potential costs. EPA believes that costs incurred after implementation of CTR-based permit limits will approach the low-end of the cost range. Costs are unlikely to reach the high-end of the range because State authorities are likely to choose implementation options that provide some degree of flexibility to dischargers. Furthermore, cost estimates for both scenarios, but especially for the high-end scenario, may be overstated because the analysis tended to use conservative assumptions in calculating CTR-based permit limits and in establishing baseline loadings. The baseline loadings for the high-end were generally based on current effluent limits rather than actual pollutant discharge data. Most facilities discharge pollutants in concentrations below current effluent limits.

4.2.1 Low-end Cost Scenario

Under the low-end cost analysis, POTWs were expected to incur the largest percentage (67 percent) of the total projected annualized costs. However, distributed among 128 major POTWs in the State, the average cost per plant would be just over $27,000 per year. Chemical and petroleum industries were estimated to incur the highest cost of the industrial categories: 18 percent of the total annual costs, with an annual average of $47,500 per plant. Facilities

EXHIBIT 4-3. SUMMARY OF ANNUALIZED COST ESTIMATES FOR THE TWO COST MODELS (IN $ MILLIONS)

		Model 2	
Cost Category	**Model 1**	**Low-end Scenario**	**High-end Scenario**
Direct Dischargers	$0	$5.2	$83.4
Indirect Dischargers	$0	$9.7	$3.2
Total	**$0**	**$14.9**	**$86.6**

Note: Costs are in first quarter 1996 dollars.

within the metals and transportation equipment category made up less than 8 percent of the total costs. However, the average cost per plant for facilities within the metals and transportation equipment category was the highest for the direct dischargers at almost $57,000 per year. The mining and lumber and paper categories were estimated to incur no costs related to implementation of the CTR (see **Exhibit 4-4**).

EXHIBIT 4-4. SUMMARY OF ANNUAL COST ESTIMATES BY DISCHARGER CATEGORY: LOW-END SCENARIO

Discharger Category	Number of Plants	Total Costs	Category Cost as % of Total Cost	Average Cost per Plant
Major Dischargers				
Publicly Owned Treatment Works	128	$3,489,913	66.9%	$27,265
Chemicals/Petroleum Products	20	$949,183	18.2%	$47,459
Electric Utilities	13	$185,091	3.6%	$14,238
Metals and Transportation Equipment	7	$398,657	7.6%	$56,951
Miscellaneous	6	$195,261	3.7%	$32,543
Mining/Construction	6	$0	0.0%	$0
Lumber and Paper	4	$0	0.0%	$0
Total	**184**	**$5,218,105**	**100%**	**$28,359**

Major permitted dischargers accounted for 100 percent of the annual costs under the low-end scenario. None of the minor sample facilities were projected to incur costs related to implementation of the CTR.

Consistent with the intent of the low-end scenario to depict flexibility in implementing the CTR, over 70 percent of the total costs consisted of annualized costs for pursuing alternative regulatory approaches. Alternative regulatory approaches were assumed necessary in the low-end cost analysis if the total cost for a sample facility exceeded $200 per pound of pollutant reduced. Annualized costs for developing and implementing waste minimization plans accounted for the bulk of the remaining costs (20 percent), primarily because of the smaller increment of pollutant to be removed (as compared to the increment required for the high-end), minimizing the need to expand existing treatment systems or install new treatment.

Five pollutants (mercury, silver, chromium IV, aldrin, and chlorodibromomethane) accounted for almost 50 percent of all annual costs. Under the low-end scenario, almost 60 percent of annual costs were for the control of toxic organics; costs to control metals and mercury accounted for about 40 percent of all annual costs.

4.2.2 High-end Cost Scenario

Under the high-end, POTWs were expected to incur approximately 74 percent of the total projected annualized cost, a total of about $61 million. However, distributed among the 128 major POTWs in the State, the average cost per plant would be almost $480,000 per year. Chemical and petroleum industries incurred the highest cost of the industrial dischargers: 16 percent of the total estimated annual cost, and averaging just over $678,000 per plant. Facilities within the metals and transportation equipment category made up only 6.9 percent of the total costs. However, the average cost per plant for facilities within the metals and transportation equipment category was the highest for the direct dischargers at almost $817,000 per year. The mining and lumber and paper categories were estimated to incur no costs related to implementation of the CTR (see **Exhibit 4-5**).

EXHIBIT 4-5. SUMMARY OF ANNUAL COST ESTIMATES BY DISCHARGER CATEGORY: HIGH-END SCENARIO

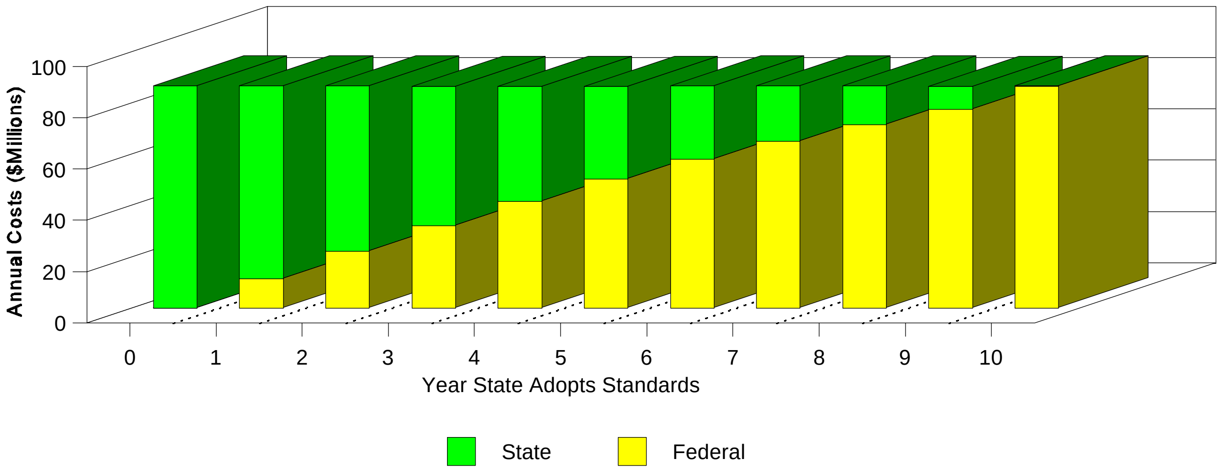
Discharger Category	Number of Plants	Total Costs	Category Cost as % of Total Cost	Average Cost per Plant
Major Dischargers				
Publicly Owned Treatment Works	128	$61,426,718	73.7%	$479,896
Chemicals/Petroleum Products	20	$13,555,706	16.3%	$677,785
Electric Utilities	13	$185,091	0.2%	$14,238
Metals and Transportation Equipment	7	$5,717,524	6.9%	$816,789
Miscellaneous	6	$2,467,389	3.0%	$411,232
Mining/Construction	6	$0	0.0%	$0
Lumber and Paper	4	$0	0.0%	$0
Total	**184**	**$83,352,428**	**100%**	**$453,002**

Note: Totals may not add up due to rounding.

Major permitted dischargers accounted for 100 percent of the annual costs under the high-end scenario. None of the minor sample facilities were projected to incur costs related to implementation of the CTR.

Almost 90 percent of the annual costs was composed of annualized operation and maintenance (O&M) and capital costs. Annualized O&M costs formed over 60 percent of the total annual costs. This result was driven primarily by the fact that the increment of pollutant removal was relatively small (i.e., less than 25 percent of current effluent levels), and that many of the sample facilities already possessed treatment processes that could be enhanced potentially to achieve CTR-based effluent limits. Therefore increased O&M was assumed adequate to comply with CTR-based effluent limits (as opposed to installing new treatment equipment). Capital costs made up 28 percent of the total annual costs.

EXHIBIT 4-6. POTENTIAL FEDERAL VS. STATE IMPLEMENTATION COSTS ATTRIBUTABLE TO THE CTR



Alternative regulatory approaches were assumed necessary in the high-end cost analysis if the total cost for a discharger category exceeded $500 per pound of pollutant reduced. There were no discharger categories that exceeded this $500 cost trigger, and therefore there were no costs related to alternative regulatory approaches incurred under the high-end scenario.

Three pollutants (mercury, silver, and chromium VI) contributed over one-third of the estimated annual implementation costs. Costs to control metals and mercury accounted for over 55 percent of all annual costs; costs to control toxic organic pollutants accounted for under 45 percent.

4.2.3 Costs to Indirect Dischargers

According to EPA Region 9, there are 2,144 significant industrial users (SIUs) that discharge to POTWs located on inland surface waters, enclosed bays, or estuaries in California. POTWs, faced with more stringent limits, may choose to control toxic pollutant discharges into their plant by requiring SIUs and other indirect dischargers to comply with stricter local limits. Based on studies at two major POTWs in California, the estimated compliance costs for SIUs are $9.6 million (representing an assumed impact on 30 percent of the population of SIUs) and $3.2 million (10 percent impact) for the low- and the high-end scenarios, respectively.

4.3 APPORTIONMENT OF COSTS BETWEEN FEDERAL AND STATE ACTIONS

As discussed in Section 1, EPA is proposing this rule to fill a gap in California's water quality standards that resulted from litigation requiring the State to rescind its water quality control plans, which contained water quality criteria for toxic pollutants. Consequently the State is not meeting the requirements of section 303(c)(2)(b) of the CWA, necessitating this federal action. However, the State is moving forward with its own rulemaking effort to restore the rescinded water quality criteria at some future date. Once the State adopts its own water quality criteria and EPA approves these criteria, EPA will take action to stay or withdraw the federal criteria.

If the federal action turns out to be only temporary, the implementation costs of the criteria would be shared with the State action. **Exhibit 4-6** shows how implementation costs of the criteria would be apportioned between the federal action and the State action depending on when the State objectives replace the federal criteria. The high-end costs estimated using Model 2 ($87 million annually) were shared based on the time period over which costs are expected to be incurred.[14] For example, if the State restores criteria 5 years after promulgation of the federal rule, it would assume approximately 42 percent of the implementation costs.[15]

4.4 POLLUTANT LOADING REDUCTIONS AND COST-EFFECTIVENESS

Exhibits 4-7 and **4-8** present the annual unweighted and toxic-weighted baseline loadings and the annual reductions in pounds for each pollutant, respectively. As shown in Exhibit 4-8, under the low-end scenario, where the baseline represents existing effluent concentrations, the expected pollutant reduction due to the State implementing the CTR is approximately 17 percent of the baseline load of 3.6 million toxic-pounds equivalent per year. The total expected pollutant load reduction under the low-end scenario is approximately 0.6 million toxic pound-equivalents per year.

Under the high-end scenario, the expected pollutant reduction due to the State implementing CTR-based WQBELs is approximately 30 percent of the baseline load of 23.8 million toxic pounds-equivalent per year. The total expected pollutant load reduction under the high-end scenario is 7.0 million toxic pound-equivalents per year.

Exhibit 4-9 ranks the top 10 pollutants by the reduction of the expressed individual pollutant as a percent of total reduction. Under the low-end scenario, chromium VI is anticipated to be reduced by over 40 percent as a percent of the total reductions, and mercury and silver account for reductions of 28.6 and 21.5 percent, respectively. Overall, organic removals account for just over 7 percent of the total reductions. The top two organics, alpha-BHC and aldrin, are estimated to be reduced by 3.5 and 3.0 percent, respectively. The number of pollutants for which pollutant

[14] High-end costs were used in order to provide a conservative estimate of the highest possible costs that could be attributed to the State or federal government.

[15] This apportionment exercise assumes that the State will adopt criteria that are similar to the federal criteria.

EXHIBIT 4-7. EXTRAPOLATED BASELINE AND REDUCTION EXTRAPOLATED BASELINE LOADS AND REDUCTIONS (Not Toxicity Weighted)

Pollutant	High-end Scenario			Low-end Scenario		
	Existing	Reductions	(%)	Existing	Reductions	(%)
Arsenic (As)	489,257	0	0.0%	45,671	0	0.0%
Cadmium (Cd)	32,082	0	0.0%	3,687	0	0.0%
Chromium VI (Cr-VI)	99,064	15,482	15.6%	34,872	7,401	21.1%
Copper (Cu)	329,374	83,153	25.2%	251,813	11,207	4.5%
Lead (Pb)	226,995	96,131	42.3%	25,182	0	0.0%
Mercury (Hg)	2,731	2,197	80.4%	697	360	51.7%
Nickel (Ni)	1,607,363	474,446	29.5%	211,290	0	0.0%
Selenium (Se)	57,135	7,535	13.2%	3,531	0	0.0%
Silver (Ag)	170,667	86,138	50.5%	28,705	2,887	10.0%
Zinc (Zn)	1,608,040	254,452	15.8%	602,919	0	0.0%
1,2-Dichlorobenzene	45,416,985	2,152,889	4.7%	0	0	
1,3-Dichlorobenzene	4,412,408	0	0.0%	0	0	
1,4-Dichlorobenzene	166,058	0	0.0%	1,084	0	0.0%
2,4,6-Trichlorophenol	991	0	0.0%	0	0	
4,4'-DDD	8	6	78.0%	8	0	0.0%
4,4'-DDT	11	7	68.8%	11	0	0.0%
Aldrin	394	386	98.2%	393	379	96.5%
alpha-BHC	342	223	65.1%	315	222	70.6%
alpha-Endosulfan	7	2	24.3%	0	0	
Benzene	54,160	0	0.0%	7,306	0	0.0%
beta-BHC	95	0	0.0%	0	0	
Bromoform	15,549	2,766	17.8%	13,369	0	0.0%
Butylbenzyl-phthalate	0	0		0	0	
Chlordane	0	0	0.0%	0	0	
Chlorobenzene	0	0		0	0	
Chlorodibromomethane	36,745	6,199	16.9%	36,607	5,388	14.7%
Chloroform	1,461,716	27,011	1.8%	224,842	0	0.0%
delta-BHC	0	0		0	0	
Dichlorobromomethane	66,404	259	0.4%	52,283	0	0.0%
Dieldrin	0	0	0.0%	0	0	
Endosulfan	3	1	18.3%	0	0	
Endrin	6	1	9.3%	1	0	0.0%
Fluoranthene	158,588	902	0.6%	0	0	
Fluorene	0	0		0	0	
gamma-BHC	211	40	18.9%	130	0	0.0%
Heptachlor	0	0	0.0%	0	0	0.0%
Heptachlor epoxide	0	0	0.0%	0	0	
Hexachlorobenzene	291	280	96.1%	0	0	
Methylene Chloride	2,116,908	4,901	0.2%	6,332	0	0.0%
PCBs	0	0	0.0%	0	0	
Pentachlorophenol	10,700	1,359	12.7%	0	0	
Phenol	0	0		0	0	
TCDD equivalents	0	0	0.0%	0	0	
Tetrachloroethylene	0	0		0	0	
Toluene	396,224,940	131,979,884	33.3%	0	0	
Total	454,766,230	135,196,649		1,550,548	27,836	

Note: Loadings rounded to the nearest whole number. NC = not calculated.
Units are in pounds per year (lb/yr).

EXHIBIT 4-8. EXTRAPOLATED BASELINE AND REDUCTION EXTRAPOLATED BASELINE LOADS AND REDUCTIONS (Toxic Weighted)

Pollutant	High-end Scenario			Low-end Scenario		
	Existing	Reductions	(%)	Existing	Reductions	(%)
Arsenic (As)	1,957,026	0	0.0%	182,683	0	0.0%
Cadmium (Cd)	166,828	0	0.0%	19,174	0	0.0%
Chromium VI (Cr-VI)	3,516,784	549,616	15.6%	1,237,973	262,746	21.2%
Copper (Cu)	154,806	39,082	25.2%	118,117	5,267	4.5%
Lead (Pb)	408,591	173,035	42.3%	45,327	0	0.0%
Mercury (Hg)	1,365,636	1,098,425	80.4%	348,491	180,189	51.7%
Nickel (Ni)	57,865	17,080	29.5%	7,606	0	0.0%
Selenium (Se)	62,848	8,289	13.2%	3,884	0	0.0%
Silver (Ag)	8,021,354	4,048,471	50.5%	1,349,148	135,218	10.0%
Zinc (Zn)	82,010	12,977	15.8%	30,749	0	0.0%
1,2-Dichlorobenzene	499,587	23,682	4.7%	0	0	
1,3-Dichlorobenzene	4,412,408	0	0.0%	0	0	
1,4-Dichlorobenzene	166,058	0	0.0%	1,084	0	0.0%
2,4,6-Trichlorophenol	347	0	0.0%	0	0	
4,4'-DDD	5,784	4,511	78.0%	5,784	0	0.0%
4,4'-DDT	68,978	47,488	68.8%	68,699	0	0.0%
Aldrin	19,679	19,325	98.2%	19,669	18,974	96.5%
alpha-BHC	34,208	22,259	65.1%	31,522	22,246	70.6%
alpha-Endosulfan	725	176	24.3%	0	0	
Benzene	975	0	0.0%	132	0	0.0%
beta-BHC	9,477	0	0.0%	0	0	
Bromoform	15,549	2,766	17.8%	13,369	0	0.0%
Butylbenzyl-phthalate	0	0		0	0	
Chlordane	461	0	0.0%	0	0	
Chlorobenzene	0	0		0	0	
Chlorodibromomethane	36,745	6,199	16.9%	36,607	5,388	14.7%
Chloroform	3,070	57	1.8%	472	0	0.0%
delta-BHC	0	0		0	0	
Dichlorobromomethane	66,404	259	0.4%	52,283	0	0.0%
Dieldrin	19,924	0	0.0%	0	0	0.0%
Endosulfan	304	56	18.3%	0	0	
Endrin	563	52	9.3%	137	0	0.0%
Fluoranthene	145,901	830	0.6%	0	0	
Fluorene	0	0		0	0	
gamma-BHC	14,802	2,791	18.9%	9,078	0	0.0%
Heptachlor	1,690	0	0.0%	424	0	0.0%
Heptachlor epoxide	0	0	0.0%	0	0	
Hexachlorobenzene	209,728	201,623	96.1%	0	0	
Methylene Chloride	889	2	0.2%	3	0	0.0%
PCBs	1,459	0	0.0%	0	0	
Pentachlorophenol	5,350	679	12.7%	0	0	
Phenol	0	0		0	0	
TCDD Equivalents	5,562	0	0.0%	0	0	
Tetrachloroethylene	0	0		0	0	
Toluene	2,218,860	739,087	33.3%	0	0	
Total	**23,760,180**	**7,018,818**		**3,582,415**	**630,029**	

Note: Totals rounded to nearest whole number. NC = not calculated.
Units are in toxic pounds-equivalent per year (lbs-eq/yr).

EXHIBIT 4-9. RANKING OF POLLUTANT REDUCTIONS AS A PERCENT OF TOTAL REDUCTIONS

Low-end Scenario		High-end Scenario	
Pollutant	**Reduction as a Percent of Total**	**Pollutant**	**Reduction as a Percent of Total**
Chromium VI	41.7%	Silver	57.7%
Mercury	28.6%	Mercury	15.7%
Silver	21.5%	Toluene	10.5%
alpha-BHC	3.5%	Chromium VI	7.8%
Aldrin	3.0%	Hexachlorobenzene	2.9%
Chlorodibromethane	0.8%	Lead	2.5%
Copper	0.8%	4,4'-DDT	0.7%
ND	ND	Copper	0.6%
ND	ND	1,2-Dichlorobenzene	0.3%
ND	ND	alpha-BHC	0.3%
Total	100%		100%

Note: Totals are rounded. ND = no data.

reductions were observed was primarily because under the low-end scenario, alternative regulatory approaches were assumed for a number of pollutants and facilities, for which no pollutant load reduction credit was taken.

Under the high-end scenario, over 80 percent of the total projected toxic-weighted annual reductions will come from reducing metals, including mercury, while less than 15 percent of expected reductions are for organic pollutants. Of the metals that will be reduced, silver accounts for more than 60 percent of the total annual reductions and mercury accounts for another 16 percent. Of the organics, toluene accounts for 10.5 percent of the total annual reductions, while several other organic pollutants are reduced at relatively small percentages.

The key highlight of the preceding exhibits is the fact that metals and mercury reductions account for a significant percentage of the total pollutant reductions under both scenarios. The two possible reasons for this result are as follows:

- It was observed that the State of California has implemented permit limits for organics that are generally equal to or more stringent than the limits projected to be implemented because of the CTR. Many of the sample facilities will not be affected by State implementation of the CTR-based WQBELs, because they have already absorbed or soon will absorb the financial and regulatory burden.

- There was a lack of data for organic pollutants. A more comprehensive data set may have shown some organic pollutants to be present, particularly under the low-end scenario, where effluent concentrations only were used to determine reasonable potential to exceed CTR-based WQBELs. However, even with a more expanded data set, similar results under the low-end scenario may be expected because for many of the facilities that did have data, most of the data points were reported below detection limits.

Exhibit 4-10 presents the costs, loading reductions, and cost-effectiveness for each discharge category evaluated in this study. As described earlier in this section, the cost-effectiveness is a measure, expressed in annual dollars per annual toxic-weighted pound removed. In order to preserve consistency, toxic weights used in previous water quality standard rulemakings were used in this study.

EXHIBIT 4-10. ANNUAL BASELINE LOADS, LOAD REDUCTIONS, AND COST-EFFECTIVENESS BY INDUSTRIAL CATEGORY

Category	Low-end Scenario			High-end Scenario		
	Annual Costs	Loading Reductions	Cost-effective-ness	Annual Costs	Loading Reductions	Cost-effective-ness
POTWs	$3.5	0.62	6	$61.4	6.72	9
Mining/Construction	$0	0	NC	$0	0	NC
Chemicals/Petroleum Products	$1.0	0	NC	$13.6	0.21	65
Electric Utilities	$0.2	0.01	17	$0.2	0.01	17
Metals/Transport Equipment	$0.4	0	NC	$5.7	0.05	122
Miscellaneous	$0.2	0.001	134	$2.5	0.03	73
Lumber/Paper	$0	0	0	$0	0	0
TOTAL	**$5.2**	**0.63**	**8**	**$83.4**	**7.02**	**12**

Notes: Totals (costs and load reductions) and cost-effectiveness are rounded values.
Loadings in millions of annual toxic-weighted pounds.
Costs are in millions of 1996 first quarter dollars.
Capital costs are annualized at 7% over 10 years.
Cost-effectiveness in $/toxic lbs-equivalent.
NC = not calculated.

Exhibit 4-10 shows that the overall cost-effectiveness was estimated to be $8 per toxic pounds-equivalent and $12 per toxic pounds-equivalent for the low- and high-end scenarios, respectively. In the low-end scenario, the highest cost per toxic pounds-equivalent removed was observed for the miscellaneous category ($134 per toxic pounds-equivalent), while the lowest was for POTWs at $6 per toxic pound-equivalent. In the high-end scenario, the highest cost per toxic pounds-equivalent removed was for the metals and transportation category at $122 per toxic pounds-equivalent, while the lowest was for POTWs at $9 per toxic pounds-equivalent.

Exhibit 4-11 presents the cost-effectiveness values from previous EPA rulemaking efforts. In comparison to these previous rulemakings, implementing the proposed CTR criteria appears to be cost-effective at both the low and high ends. Even when the higher category-based cost-effectiveness numbers from Exhibit 4-10 are used, the comparison still appears favorable.

EXHIBIT 4-11. INDUSTRY COMPARISON OF INCREMENTAL COST-EFFECTIVENESS FOR DIRECT DISCHARGERS (TOXIC AND NONCONVENTIONAL POLLUTANTS ONLY) COPPER-BASED WEIGHTS (FIRST QUARTER 1996 DOLLARS)

Industry	Incremental Cost-effectiveness for Selected Technology Option(s) ($/lbs-eq removed)[1]
Aluminum Forming	167.71
Battery Manufacturing	2.77
Coil Coating – Canmaking	13.86
Coal Mining	None
Coil Coating	67.91
Copper Forming	37.42
Electronics I	559.94
Electronics II	Not Available
Foundries	116.42
Inorganic Chemicals I	<1.39
Inorganic Chemicals II	8.32
Iron and Steel	2.77
Leather Tanning	None
Metal Finishing	16.63
Nonferrous Metals Forming	95.63
Nonferrous Metals Manufacturing I	5.54
Nonferrous Metals Manufacturing II	8.32
OCPSF	6.93[2]
Pharmaceuticals	1.39
Plastics Molding and Forming	None
Porcelain Enameling	8.32
Petroleum Refining	None
Pulp and Paper (PCB control for De-ink	24.95
Textile Mills	None

[1] Updated from 1981 dollars. Reflects incremental cost-effectiveness to proceed from current levels to levels represented by best available technology economically achievable.

[2] Reflects costs and removals of both air and water pollutants.

Source: *Cost-Effectiveness Analysis of Proposed Effluent Limitations Guidelines and Standards for the Pesticide Manufacturing Industry*. (EPA/821/R-92-004, April 1992)

5. 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 the State of California implementing CTR-based water quality criteria into California water quality standards and NPDES permit limits.[16] This chapter presents economics concepts and analytical issues associated with defining benefit categories and developing quantified and monetized benefits estimates. Section 5.2 describes the economic concepts used in the benefits analysis; Section 5.3 discusses the limitations of the analysis; and Section 5.4 describes the baseline and attribution issues relevant to the CTR.

5.1 ECONOMIC CONCEPTS APPLICABLE TO THE BENEFITS ANALYSIS

This economic analysis 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 describes 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.

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

[16] Hereafter, references to the benefits resulting from the CTR, are referring 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.

Use Benefits	
In-Stream	• Commercial fisheries, shellfisheries, and aquaculture; navigation • Recreation (fishing, boating, swimming, etc.) • Subsistence fishing • Human health risk reductions
Near Stream	• Water-enhanced noncontact recreation (picnicking, photography, jogging, camping, etc.) • Nonconsumptive 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
Ecological	• 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 can include both direct and indirect uses of the impacted waters, and the direct use category embraces both consumptive and nonconsumptive activities. In most applications to pollutant reduction scenarios, the most prominent use benefit categories are those related to recreational fishing, boating, and swimming.

Whether or not recreational use benefits reflect society's prime motivation for environmental protection measures is unclear. Many benefits analyses focus on recreational values however, 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 are 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 potential use benefit.

Passive Use (Nonuse or Intrinsic) 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 use (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 nonrival (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).[17]

Such passive use values may be significant, but it is difficult to assign a benefit value to these motivations. Whereas human uses of a resource can be observed directly and valued with a range of technical economic techniques, passive use values can only be ascertained 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.[18] 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 ecosytem health. The species include, but are not limited to, bald eagles, other 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, bird watching). But of greater relevance is the applicability of ecologic benefits under the traditional passive use categories of existence (stewardship or preservation) 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 policy making.

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.

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[17] 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.

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

5.2 LIMITATIONS INHERENT IN THE BENEFITS ANALYSIS

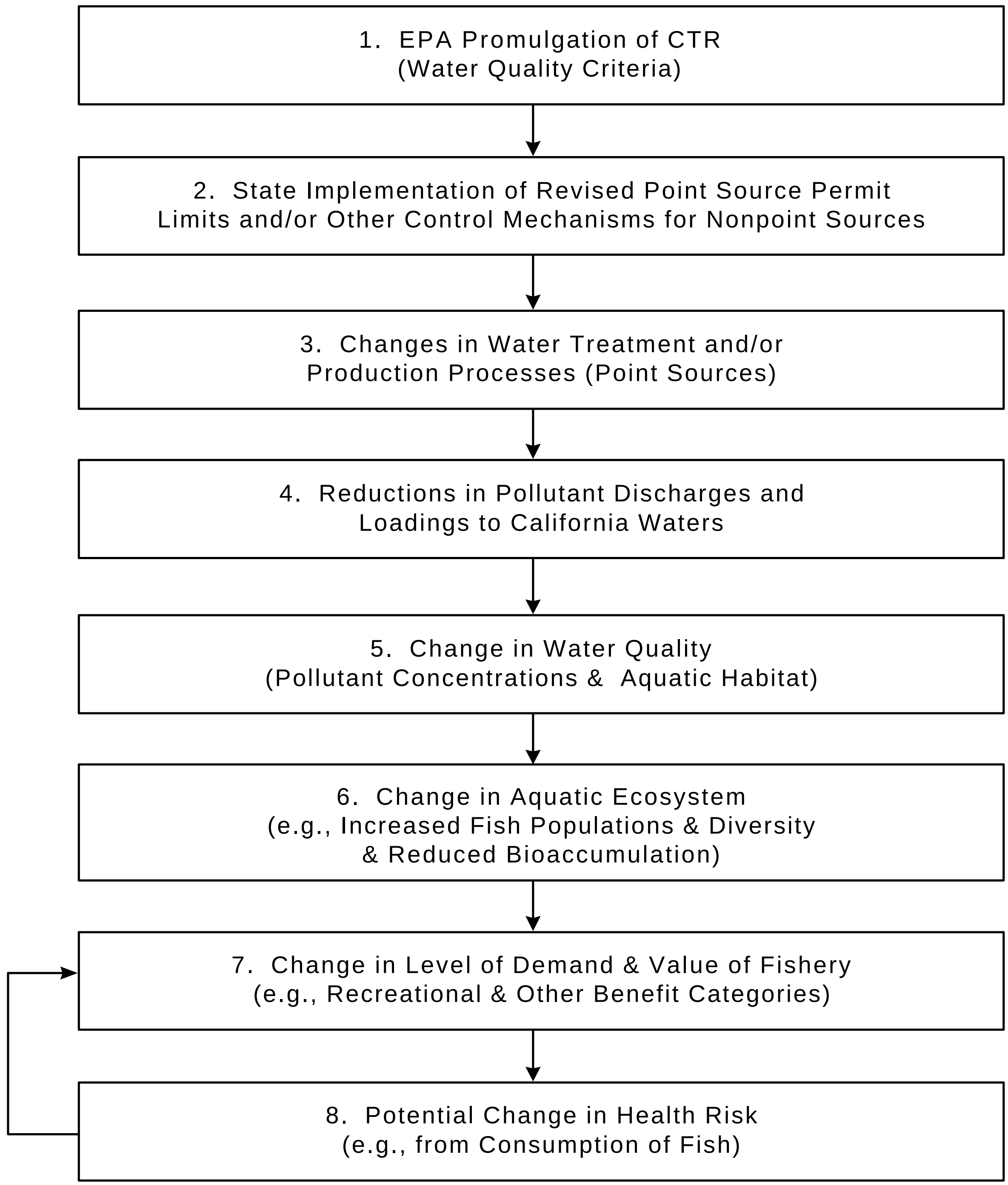
5.2.1 Causality: Linking the CTR to Beneficial Outcomes

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 policy making; 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.

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 regulatory bodies 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 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 feasible 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

EXHIBIT 5-2. CHAIN OF EVENTS IN CTR BENEFITS ANALYSIS



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 timeframe 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 forecast accurately 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 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. The areal extent of contamination (e.g., PCBs or dioxins bound in sediment) typically is very widespread (even if it originates from a well defined source at a specific location). The 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 Conclusions

It is important that the inherent limitations of the benefits analysis be recognized and appreciated. The numerical results are based on limited assessments of the extent to which the CTR may contribute to improvements beyond baseline levels. The benefits analysis is geared toward indicating: (1) the types of benefits to be anticipated; (2) a general approach for describing and, as feasible, estimating these benefits; (3) the general magnitude of the monetized worth of several categories of benefits; and (4) an indication of how benefits compare to costs.

It also is important to recognize that the principal benefits of the CTR are likely to include long-term (i.e., delayed) ecologic benefits. The persistence and toxicity of the compounds to be controlled under the CTR imply that, in essence, the principal benefits will exhibit characteristics that make them less amenable to empirical evaluation than the benefits of most other programs: (1) temporally, most of the direct benefits are likely to be delayed for many years; and (2) structurally, the benefits are largely of the ecologic or passive use variety. For example, reduced toxic loadings under the CTR may result, after a recovery period of several years, in the improved health of aquatic ecosystems.

5.3 BASELINE AND BENEFITS ATTRIBUTION ISSUES

Benefits estimates were derived in this study using an approach in which the benefits of discrete, large-scale changes in water quality beyond present day conditions are estimated (wherever feasible), and then a share of those benefits is apportioned to the CTR. This implies a need to recognize and establish the appropriate baseline, and then to provide a means for assigning to the CTR an appropriate share (fraction) of benefits of moving from the baseline to the larger-scale water quality improvement goals. These two issues are discussed below.

5.3.1 The Relevant Water Quality Baseline for the CTR

One of the most important, practical analytic problems faced in attributing benefits to the CTR is the need to account for the appropriate water quality baseline. A benefits analysis is, for the most part, only able to measure improvements from *current* (i.e., observable) conditions. However, the appropriate baseline is to account for water quality as it is *anticipated* after current compliance with the numeric criteria embodied in section 303(c)(2)(B) of the Clean Water Act, and also accounting for other regulatory and remedial actions in progress, or anticipated, absent the CTR. Therefore, there is an important distinction between *current* conditions, and the conditions that reflect the relevant *baseline* for the CTR.

To estimate the benefits relevant strictly to the CTR, the benefits of improvements from current conditions need to be apportioned—the share of benefits anticipated absent the CTR must be separated from the share for which the rule is responsible. An empirical approach for basing such an attribution would be to discern how toxic loadings reductions are allocated from current conditions to the CTR-relevant baseline, and then from this baseline to the post-CTR loadings. In estimating pollutant loadings and reductions for the CTR, EPA adjusted the actual permit limit, where necessary, to reflect the relevant baseline. The loadings reduction estimates indicate significant reductions in the toxic-weighted loadings of pollutants from baseline (not current) conditions. Thus, there is reason to anticipate that the CTR may have a significant impact relative to loadings at its baseline, but we have no empirical information with which to discern how this reduction compares to the difference between current conditions and the CTR-relevant baseline.

5.3.2 Attribution of Benefits to the CTR

The share of larger-scale water quality improvement benefits attributed (apportioned) to the CTR was based on a three-stage process:

- 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 are estimated and 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 percent of the toxic-weighted pollutant loadings that contributed to baseline impairments, then one would estimate (absent better or more refined data) that perhaps $200 million of the potential water quality benefits would be attributable to the potential elimination (100 percent reduction) of all point source discharges. If the CTR was expected to achieve a 50 percent reduction in the offending point source discharges, one would then develop an estimate of $100 million as a rough approximation of CTR-related benefits.[19] 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 percent of the total loadings problem (reducing by 50 percent the 40 percent of total loadings due to point sources).

5.3.3 Point versus Nonpoint Loads, and Current versus Historical Loads

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 (NPS) 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, NPS load estimates are probably highly imprecise and very incomplete (likely to be underestimated because of omitted sources). Even point source estimates of 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 attributable to historical (i.e., past) 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 attributable to historical discharges rather than current loadings. For example, a recent investigation found that current PCB levels in parts of Green Bay on Lake Michigan were more than 90 percent attributable to past discharges, which were now being continuously re-released to the water column and biota through the sediment and the food web. The degree to which historical loads contribute to present day concentrations will vary according to many complex contaminant- and site-specific factors (e.g., the Green Bay data may reflect a worse case attribution scenario for assigning benefits to controlling ongoing point source loadings). 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.

[19] Forty percent of $500 million equals $200 million, and 50 percent of the $200 million equals $100 million.

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. QUALITATIVE ASSESSMENT OF POTENTIAL ECOLOGICAL BENEFITS

This chapter describes the types of ecological benefits anticipated to result from implementation of the NPDES permit program to meet water quality standards. 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, and 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, 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. Sixty-three types of freshwater fish are unique in that a high percentage of them are endemic to the State; that is, they are found nowhere else (Moyle, 1994, 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, over 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).[20] Impairments of ambient water quality in California have been summarized in the *Analysis of the Potential Benefits Related to Implementation of the California Toxics Rule* (U.S. EPA, 1997); they 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, 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.

- Trace elements (especially selenium) may be responsible for water quality impairment in 52 percent of all bays, 55 percent of rivers and streams assessed, and 16 percent 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 waterbodies, 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.

[20] 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.

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 Waterbodies Impaired	Ecological Resources Potentially Affected
Region 1: North Coast Region	55% of bays (16,500 acres); minor impairment of other waterbodies	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 (200,00 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

[1]Based on 1994 assessment of water quality. Some key waterbodies 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.

EXHIBIT 6-1. SUMMARY OF BASELINE CALIFORNIA REGIONAL WATER QUALITY ASSESSMENTS (Continued)

Region	Areal Extent of Toxics Impairment	Pollutants of Concern	Primary Pollutant Sources	Key Waterbodies 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 waterbodies; 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 waterbodies 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

Source: U.S. EPA (1997).

- 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 waterbody 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 (nine inland waterbodies and three 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 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.

Thus, 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 either 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 may also be bioaccumulated in organisms, making them available to terrestrial predators (e.g., fish-eating wildlife) 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 (biological organisms, including threatened and endangered species), 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 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, and the specific tissues/organs affected depend on the exposure pathways, the exposure concentrations, and the ability to metabolize or excrete the accumulated contaminants. Metabolic degradation of the contaminants occurs through enzymatic pathways, and the rate/ability of metabolic degradation 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, which results in four identified biotic responses (**Exhibit 6-2**). The four biotic responses—lethal toxicity, sublethal toxicity, bioaccumulation, and habitat alteration— provide broad categorization for a multitude of specific biotic responses.

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 results in death of the organism. The death of individuals from populations can influence the future reproductive viability of populations, and in turn may influence higher levels of biological organization (e.g., communities). 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, reproduction; increase 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 fishes, 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. The potential adverse effects of toxics present in California's aquatic systems on the various levels of biological organization include a list of quantifiable endpoints ranging from lethality (death due to direct exposure to acutely toxic concentrations or indirect exposure to sublethal concentrations that eventually cause death) to sublethal endpoints owing to direct or indirect exposures that cause physiological or behavioral abnormalities.

Toxics present in California's aquatic systems can affect many aspects of cellular metabolism and physiology that can reduce normal growth in organisms. Exposure to the toxics can affect biological activities, such as feeding, respiration, and enzymatic pathways, that are necessary for physiological maintenance (homeostasis) and growth. Reduced physiological fitness or reduced growth in organisms will affect the ability of an organism to survive and reproduce in the environment.

Exposure to certain toxics present in California's aquatic systems, including aromatic hydrocarbons and heavy metals, can increase the rate of genetic mutations by impairing DNA synthesis, increasing DNA-strand exchanges, and altering chromosome number. 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.

Effects on Ecosystems

In addition to adverse effects on biota, toxics may also adversely affect ecosystem function and integrity through direct and indirect effects on biota. The effects of toxics in ecosystems are complex because of the diversity of species assemblages and trophic interactions (e.g., direct and indirect interaction between populations in aquatic food webs) (Barron and Woodburn, 1995). Examples of the effects of toxics on ecosystem function and integrity include alteration of system processes such as impaired decomposition of organic matter and disruption of predator-prey interactions.

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
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

[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.

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
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

The effects of toxics on ecosystem function and integrity are complex and difficult to estimate. Functional endpoints (e.g., community metabolism) may be less sensitive to toxic effects than are structural properties (e.g., loss of sensitive species) of ecosystems (Kersting, 1994). For example, predators may switch to alternate prey (Eaton et al., 1985), or phytoplankton abundance may be maintained by changes in the dominant algal species (Brock et al., 1992). Also, several species may perform similar functions, with sensitive species replaced by more resistant species following contaminant exposure. Thus functional redundancy in ecosystems may confer resistance, despite major changes in ecosystems structure, such as species losses (Barron and Woodburn, 1995). However, contaminant effects on ecosystem structure and species diversity are likely to be specific to the aquatic system (e.g., pond vs. river) and exposure scenario (e.g., chemical mixture, duration, and extent of exposure).

6.3 TOXIC REDUCTIONS AND ECOLOGICAL HEALTH

When the State implements the NPDES permit program to achieve water quality standards, it is anticipated that ambient water quality will improve through reductions in the concentrations of toxics in California's aquatic systems. As discussed in the Draft *Analysis of the Potential Benefits Related to Implementation of the California Toxics Rule* (U.S. EPA, 1997), California's aquatic ecosystems include food webs of phytoplankton, invertebrates, fish, birds, mammals, and other organisms that interact with each other through a complex flow of matter and energy. Because of this linkage, toxic reductions in ambient water that result in increased survival, growth, and reproductive fitness of individuals and populations should contribute to the health of other ecosystem components, including other species, populations, and communities.

Toxics reductions are anticipated for a variety of metals, pesticides, aromatics, and chlorinated organic contaminants, as discussed by U.S. EPA (1997). Exposure of biota to these pollutants at levels exceeding the resulting water quality standard based on the proposed criteria may result in adverse ecological effects that will change or alter the structural or functional ability of the organism to survive, grow, and reproduce. Adverse effects of many toxics include increased susceptibility to disease, reduced growth and development, altered physiology and behavior, impaired reproductive health and behavior, and if concentrations are high enough, death. Any one of these adverse effects can ultimately affect the survival, reproductive success, and overall health of a population, which may affect ecosystem health (U.S. EPA, 1997).

Because all components of ecosystems are linked to each other, improved survival, growth, productivity, and reproductive capacity of aquatic and terrestrial organisms should translate to improved ecosystem stability, resilience, and overall health. Improvements in ecosystem health following toxics reductions will depend on the interaction of a variety of factors:

- The timing, duration, and magnitude of baseline toxics exposure
- The rate of toxics reduction attributable to achieving CTR-based water quality standards
- The species assemblages present
- The trophic structure of the system
- The reproductive capacity and growth rate of susceptible and tolerant populations
- The availability of colonizing organisms.

6.4 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 (e.g., recreational fishing), passive use (nonuse) values (e.g., existence values) and, to the extent not reflected elsewhere, ecologic benefits (improved ecosystem health apart from or in addition to any anthropocentric value).

This section provides a qualitative description of potential ecologic benefits resulting from improvements in ecosystem health under the proposed rule. Only a qualitative description of ecologic benefits is provided because of: (1) the complexity and diversity of California aquatic systems, and the diversity of ecological receptors (Section 6.1); (2) the multitude of contaminants and exposure conditions (Section 6.2); (3) the complexity of ecosystem structure and function, and uncertainty in the interaction between factors involved in ecosystem recovery and responses (Section 6.3); and (4) uncertainty regarding the extent to which the CTR will result in toxics loading reductions significant enough (relative to the contribution of historic and on-going point and nonpoint loadings) to generate appreciable changes in ambient concentration and ecosystem health.

EPA performed a qualitative assessment of the ecologic benefits of the proposed rule (IEC, 1996), rather than a contaminant specific quantitation of the magnitude and extent of benefits accruing for each affected aquatic system. A quantitative analysis of the ecologic benefits anticipated to accrue from the CTR would include the following:

- Inventories describing resident resources of ecological, commercial, and recreational value.

- An assessment of migratory biota utilizing California ecosystems and an assessment of the ecological interactions between resident and migratory species.

- Development of fate and transport models to describe exposure, uptake, distribution, transformation, accumulation, and dissipation of toxics.

- Knowledge of the magnitude and composition of CTR-induced toxic loadings reductions relative to all impairment-generating contaminant loadings.

- Knowledge of the toxicity and associated responses caused by exposure of ecologically important species and habitats to toxics to be regulated by the proposed rule.

Baseline data (before CTR-related toxics reduction) would need to be gathered to make well informed assessments of the potential benefits. Once collected, it would be necessary to estimate the change in ecological quality and values (i.e., accrued benefits) resulting from toxics reductions under the proposed rule.

An additional approach to estimating ecologic benefits could use existing case studies documenting the benefits of restored or remediated ecological resources. For example, an appropriate case study might include a published study (if available) documenting the benefits accruing from prior restoration of a formerly contaminated aquatic system. A quantitative case study approach would involve: (1) determining the availability and appropriateness of existing studies for remediated sites; (2) assessing the ecological benefits accruing from the change in baseline (pre-remediation) ecological services; (3) comparing the benefits to the estimated costs of the remedial actions; and (4) evaluating the representativeness of the case study(s) to the aquatic systems to be regulated by the proposed rule.

Under the proposed rule, reductions in toxics may be appreciable enough to provide substantial improvements in the ambient water quality of California's aquatic ecosystems (U.S. EPA, 1997). 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 could not be quantified 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 are expected to contribute to improved conditions for California fish spawning and/or migration in bays/harbors and estuaries, lakes, rivers and streams, and saline lakes.

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

- Reductions in toxics may contribute to improved 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 may reduce 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 may contribute to improved 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 may contribute to the increased stability, resilience, and overall health of numerous ecosystems throughout California, and may contribute to protecting, restoring, and maintaining California's ecological diversity.

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

It is difficult to estimate the benefits of implementation of the NPDES permit program to meet water quality standards based on criteria in the CTR because of the contribution of point sources to 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 on society's values for water quality improvements. However, there is typically less information available as to the exact source of contamination in many waterbodies. Indeed, there are limited data available on the contribution of point sources to the toxic-related problems in California waters.

To estimate the potential benefits of the proposed rule, 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).

7.1 SAN FRANCISCO BAY

EPA found two sources describing the relative contributions of point and nonpoint sources of toxic loadings in San Francisco Bay: Davis et al. (1991) and NOAA (1988a).

Davis et al. (1991) 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.

Davis et al. (1991) estimated nonpoint source loadings based on estimates of urban and nonurban runoff, riverine inputs, atmospheric deposition, oil spills, and contributions from dredging activities. They used loadings data for urban runoff and dredging from Gunther et al.(1987)[21]. They estimated nonurban runoff using a NOAA model that factors in sediment loss from nonurban lands and average trace metal concentrations in soil. Riverine inputs were based on pollutants from the Sacramento and San Joaquin rivers, and 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, and the loading from the San Joaquin River was based on 1985–87 water quality data collected by the U.S. Geological Survey. Atmospheric deposition loadings were based on measurements in other parts of the United States, as reported in Gunther et al. (1987).

The second study on the relative contribution of point and nonpoint sources of toxic loadings in San Francisco Bay is the National Coastal Pollutant Discharge Inventory developed by NOAA's Strategic Assessment Branch (1988a). NOAA 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. They estimated nonpoint source loadings to include urban and nonurban runoff, and riverine inputs.

[21] Other studies suggest that Gunther et al. (1987) may underestimate the contribution from dredging activities.

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 stormwater 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 U.S. Geological Survey'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 by 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 Davis et al. (1991) and NOAA (1988a). 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 percent for the NOAA data based on the median weight for the class of chlorinated hydrocarbon pesticides,[22] and a range of 1.5 to 7.1 percent for the Davis et al. (1991) data. Because of the uncertainties in the estimates, EPA

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

EXHIBIT 7-1. TOXIC-WEIGHTED POINT SOURCE CONTRIBUTIONS TO SAN FRANCISCO BAY[1]

Pollutant	NOAA[1]	Davis et al.[2]
Zinc	5.2%	4.0–18.9%
Copper	5.5%	4.0–11.9%
Nickel	N/A	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	N/A	28.4–28.4%
Mercury	6.9%	26.4–51.8%
Chlorinated Hydrocarbon Pesticides	51.5%	N/A

[1] Toxic weighting based on loadings data from NOAA (1988a).

[2] Source: U.S. EPA (1997); range based on data from Davis et al. (1991).

assumed thatpoint sources contribute between 1 and 10 percent of total toxic loadings to San Francisco Bay for the purposes of estimating the potential benefits of the point source controls of the CTR. EPA is soliciting additional data and information on the relative contribution of point and nonpoint sources of toxic pollutants to San Francisco Bay.

EPA's analysis is subject to the following uncertainties and limitations:

- The concentration of pollutants below the detection limit is unknown. Davis et al. (1991) and NOAA (1988a) relied on assumptions about concentrations below the detection limit to estimate pollutant loadings.

- The available studies do not include estimates for some point and nonpoint sources of pollutants (e.g., "historic" loadings from contaminated sediment or point source mine drainage).

- Davis et al., (1991) and NOAA (1988a) 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.

- 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.

- Davis et al. (1991) did not estimate the contribution from pollutants, other than selenium, in the Sacramento River, and did not have local data for the estimates 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 few air sources of toxics that might reach the bay.

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[23]. **Exhibit 7-2** presents the estimated contribution of point sources to toxic loadings in each bay.

EXHIBIT 7-2. POINT SOURCE CONTRIBUTION OF OTHER CALIFORNIA BAYS

Pollutant	Nonurban Bays[1]		Urban Bays[1]		
	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	*	*	*

[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.

* Point source contribution varies within group of pesticides.

Source: U.S. EPA (1997).

Based on NOAA (1988b and 1988c), 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 percent of loadings in the nonurban bays (Monterey and Humboldt Bays, respectively), and 91.1, 87.9, and 82.6 percent in the urban bays (San Diego, Santa Monica, and San Pedro Bays, respectively).

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 showed the contribution of point sources to be much higher in these urban bays than EPA estimated for San Francisco Bay. The reason for this discrepancy is not readily apparent. EPA is soliciting additional data and information on the relative contribution of point and nonpoint sources to total toxic loadings in urban bays and estuaries.

On average, EPA estimated that point sources account for 67 percent of toxic-weighted loadings to urban bays and 28 percent of toxic-weighted loadings to nonurban bays.[24] These results are used to attribute a portion of the

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

[24] The average for urban bays includes 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 =

recreational angling benefits described in Chapter 8 to the point source controls of the CTR. However, EPA did not know the proportion of recreational angling days (and thus benefits) that occur in urban bays versus nonurban bays. Therefore, EPA developed a weighted average assumption of the point source contribution to other 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,[25] and obtained total population living within 10 miles of each bay.[26] 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 estimate of 64 percent for point source loadings to urban and nonurban bays. 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 Water Quality Assessment database (State Water Resources Control Board, 1994) (U.S. EPA, 1997). This approach yielded a land area-weighted average estimate of 42 percent for point source loadings to urban and nonurban bays.

The limitations and uncertainties noted in Section 7.1 also apply to the results for other bays and estuaries.

7.3 FRESHWATER RESOURCES

Because of data and resource limitations, EPA could not assess the relative source contribution for each freshwater resource in California. Consequently, EPA relied on a limited set of data to develop a relatively broad range of point source contributions to freshwater. EPA used data for the Sacramento and San Joaquin rivers (each of which have a different relative influence of permitted mining discharges), and information on the influence of permitted mines on freshwaters, to develop a statewide estimate of the relative contribution of point sources to toxic loadings in freshwater. Because of the limited data reflected in the analysis, EPA is soliciting additional data and information on this issue.

EPA estimated the relative point source contributions to toxic loadings for the Sacramento and San Joaquin Rivers based on data from the Central Valley Regional Water Quality Control Board (CVRWQCB). 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.

66.7). The average for nonurban bays is the mean of the toxic-weighted average for the two nonurban bays (23.2 + 33.1 = 28.2).

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

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

EXHIBIT 7-3. TOXIC-WEIGHTED POINT SOURCE CONTRIBUTIONS OF SELECTED CONTAMINANTS TO FRESHWATER RIVERS

Pollutant	Sacramento River[1]	San Joaquin River[1]
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 Regional Water Quality Control Board, *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 than for the San Joaquin River. Using a toxicity-weighted average across all five pollutants, EPA estimated that about 46.3 percent of loadings to the Sacramento River, and only 3.4 percent 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 using 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.[27] 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), and calculated a weighted average point source contribution of 3 percent for lakes and 3 percent for rivers. The 3 percent for freshwater lakes is also applied to saline lakes.

EPA's analysis for freshwater is subject to the following uncertainties and limitations:

- 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.
- Only a subset of cities in the Central Valley region were incorporated for the estimate of urban runoff.
- The use of effluent concentration data from the Sacramento County POTW may not be representative of effluent from other facilities.
- Historic loadings in sediments may not be accounted for.

7.4 SUMMARY

[27] A very small percentage of all 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).

Exhibit 7-4 summarizes EPA's estimate of the relative contribution of point sources to total loadings 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 is generally not known, it is difficult to assess their overall impact on the estimates.

EXHIBIT 7-4. ESTIMATED SHARE OF TOTAL LOADINGS ATTRIBUTABLE TO POINT SOURCES FOR CALIFORNIA WATERBODIES

Waterbody	Total 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); Central Valley Regional Water Quality Control Board; and California 1994 Water Quality Assessment 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, PCBs, dioxin, and certain metals (e.g., silver).	High		✔	
Detection limits for some key pollutants (e.g., PCBs and dioxin) too high to reliably estimate the relative contribution of point and nonpoint sources.	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. QUANTIFIED AND MONETIZED BENEFITS ESTIMATES

EPA quantified and monetized three categories of potential benefits from implementation of the NPDES permits program to meet water quality standards based on criteria established in 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.

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 have also been adversely affected by toxic pollution. Fish consumption advisories have been issued for nine inland waterbodies, 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.

EPA assessed baseline human health risks (cancer and systemic effects) based on current 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 on recreational anglers. Consequently, this analysis does not include health risks to nonangler family members that consume fish obtained from recreational

EXHIBIT 8-1. FISH CONSUMPTION HEALTH ADVISORIES IN CALIFORNIA

Waterbody/Location	Advisory for General Population		Advisory for Sensitive Populations[1]		Contaminants of Concern
	Avoid Consumption	Limit Consumption[2]	Avoid Consumption	Limit Consumption[2]	
Inland Surface Waters					

EXHIBIT 8-1. FISH CONSUMPTION HEALTH ADVISORIES IN CALIFORNIA (Continued)

	Advisory for General Population		Advisory for Sensitive Populations[1]		
Waterbody/Location	**Avoid Consumption**	**Limit Consumption[2]**	**Avoid Consumption**	**Limit Consumption[2]**	**Contaminants of Concern**
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
Guadalupe Reservoir Calero Reservoir Alamaden Reservoir Guadalupe River Guadalupe Creek Alamital Creek Plus associated ponds along these rivers and creeks (Santa Clara County)	All fish		All fish		Mercury
Harbor Park Lake (Los Angeles County)	Goldfish Carp		Goldfish Carp		DDT Chlordane
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 over 35"	Max. of 2 meals per month ▸ All sport fish	Striped bass over 27" Shark over 24"	Max. of 1 meal per month ▸ All sport fish	Mercury PCBs Dioxins Pesticides
Belmont Pier/Pier J (Los Angeles Harbor)		Max. of 2 meals per month ▸ Surf perch			DDT, PCBs
Los Angeles/Long Beach Harbors (esp. Cabrillo Pier)	White Croaker	Max. 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, or who are breast-feeding, and children under six 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).

angling.[28] Nor does this analysis 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 effected 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 percent of the trips taken near the bay (the area north of Stinson Beach to south of Davenport) (NMFS, 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 may also participate in freshwater angling. The *National Survey of Fishing, Hunting, and Wildlife-Associated Recreation* (U.S. FWS, 1993) indicates that 50 percent of the saltwater anglers (adults and children) in California fish exclusively in saltwater, while the remaining saltwater anglers fish in both saltwater and freshwater. EPA assumed that each angler that splits his or her time spends half of his or her time using each resource. Therefore, of the 166,000 anglers that use San Francisco Bay for saltwater angling, EPA estimated that 83,000 anglers fish exclusively in the bay, and the other 83,000 split their time between the bay and freshwater resources. Thus, EPA estimated that 125,000 full-time equivalent anglers use San Francisco Bay.

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 percent of anglers take fewer trips in response to fish consumption advisories (Fiore et al., 1989;

[28] Approximately 45 percent of anglers share the catch with family members (SSFBA, 1995).

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 percent 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 have fallen by 22 percent from 1987 to 1994.) EPA's adjusted estimate of full-time equivalent anglers for San Francisco Bay is 112,500.

Freshwater Resources

EPA estimated the number of freshwater anglers in California based on the *National Survey of Fishing, Hunting, and Wildlife-Associated Recreation* (U.S. FWS, 1993) and license sales reported by the California Department of Fish and Game for 1991. The survey indicated that there were 3.6 million resident anglers in California (including adults and children): 61 percent of adult anglers fish exclusively in freshwater, 20 percent split their angling activity between fresh and saltwater, and the remaining 19 percent of anglers fish exclusively in saltwater.[29] Assuming that children apportion their angling time in the same way as adults, EPA estimated that 2.2 million anglers fish exclusively in freshwater, and 710,700 split the time between fresh and saltwater resources. EPA assumed that each angler that splits the time spends half of the time using each resource. Based on this information, EPA calculated that 2.5 million full-time equivalent anglers use freshwater resources in California.

EPA reduced this estimate of freshwater anglers by the number of one-day license sales in 1991 (304,328) 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 one-day licenses between salt and freshwater resources. Using this approach, EPA estimated a potentially exposed population of 2.2 million full-time equivalent anglers using freshwater resources in California.

[29] Because these data may have been collected before a fish consumption advisory for several freshwater resources, potentially exposed populations may be overestimated.

EXHIBIT 8-2. LOCATIONS OF FISH CONSUMPTION HEALTH ADVISORIES

EXHIBIT 8-2. LOCATIONS OF FISH CONSUMPTION HEALTH ADVISORIES (continued)

8.1.2 Fish Consumption

EPA estimated fish consumption rates for both San Francisco Bay and freshwater anglers based on 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 grams/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**).

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).

To the extent that SMBSCS (1994) does not accurately characterize the fish consumption of anglers using freshwater resources, it will lead to an over- or under-estimate of risks. EPA is soliciting data on the consumption of freshwater fish by California anglers.

8.1.3 Fish Tissue Contaminant Concentrations

EPA calculated the arithmetic mean of fish tissue contaminant concentrations for San Francisco Bay and for statewide freshwater resources based on available data (see Appendix F in U.S. EPA, 1997). In calculating the concentrations, EPA used one-half the method detection level (MDL) for samples in which contaminants were reported as nondetects (U.S. EPA, 1993).

San Francisco Bay

EPA obtained fish tissue contaminant levels from a study conducted by the San Francisco Regional Water Quality Control Board (SFRWQCB, 1994). 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 frequent means of consumption (croaker and surf perch fillets with skin, and shark and striped bass without skin).

EPA developed species-weighted fish tissue contaminant concentrations for San Francisco Bay by relying 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 weights, EPA assumed that keep rates are comparable across the four species in the analysis.[30]

EXHIBIT 8-4. SPECIES WEIGHTS FOR SAN FRANCISCO BAY FISH CONSUMPTION

Species	Number of Fish Caught	Percent of Catch
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.

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 waterbodies 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)**.

8.1.4 Baseline Risk Levels

EPA calculated exposure based on the assumption that each fish contained all contaminants listed at the concentrations used in the benefits report (this information is provided in the public record). **Exhibit 8-6** reports the assumed toxicity values for cancer and systemic effects. Standard EPA assumptions were used regarding length of residence (70 years) and body weight (70 kg) (U.S. EPA, 1989b).

EXHIBIT 8-5. SPECIES WEIGHTS FOR FRESHWATER FISH CONSUMPTION

Species	Annual Fishing Activity[1] (number of days)	Keep Rate[2]	Keep Rate Weighted Days	Consumption[3] Weighting Factors

[30] 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 over- or under-estimate bay angler risks (U.S. EPA, 1997).

Trout	10,641	25%	2,660	28.0%
Bass	6,164	25%	1,541	16.2%
Catfish	4,098	80%	3,278	34.6%
Panfish	1,866	90%	1,679	17.7%
Other	1,315	25%	329	3.5%
Total	24,084	—	9,487	100.0%

[1] Fishing activity obtained from *National Survey of Fishing, Hunting, and Wildlife-Associated Recreation, California, 1993,* USFWS.

[2] Keep rate estimated from *Sacramento River Sport Fish Catch Inventory Project, Final Performance Report,* CDFG, 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 dividing the keep rate weighted days for each species by the total keep rate weighted days.

San Francisco Bay

Exhibit 8-7 presents estimated 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 that statistical excess cancer cases per year at baseline are less than 1. For anglers consuming a mixed species fish diet at the 90th percentile consumption rate, EPA estimated that the individual excess lifetime cancer risk is 9.2×10^{-4}.[31] These risks are dominated by PCBs and dioxin, which contribute 49 and 41 percent, respectively, to the cancer risk for an average angler.

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 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.[32]

EXHIBIT 8-6. TOXICITY VALUES FOR 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}	✔	✔

[31] 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.

[32] 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.

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}	1.0×10^{-9}	✔	
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 and 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 and dioxin RfDs, 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.

Source: U.S. EPA (1997).

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
	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%
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	1.8×10^{-4}	9.2×10^{-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).

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

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.

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
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).

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
	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%
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. 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**
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).

8.1.5 Potential Risk Reductions Due to the Rule

To estimate the potential reductions in fish tissue contaminant concentrations attributable to implementation of the NPDES permit program to achieve water quality standards based on the CTR, EPA multiplied baseline risks by the reduction in loadings expected from the rule (U.S. EPA, 1997), and by the assumed contribution of point sources to total loadings as developed in Chapter 7 (**Exhibit 8-11**). EPA then calculated the potential reductions in human health risks attributable to implementation of the CTR.

The health risk reductions attributable to the rule are small because the loadings reductions for the contaminants that account for the majority of the baseline risks are small. EPA estimated no reduction in loadings of the two contaminants (PCBs and dioxin) that account for 90 percent of the cancer risks for San Francisco Bay anglers. And, of the four contaminants that account for 90 percent of the cancer risks for freshwater anglers, EPA estimated a loadings reduction for only one (DDT, which accounts for 17 percent of baseline risks). The exact reason for these small loadings reductions is not clear; however, one possible explanation may be related to the methods used to account for pollutant load reductions when concentrations are below the analytical method detection level (MDL). As described in Section 4.1.3, if both the CTR-based effluent limit and the existing effluent permit limit (high-end scenario) or effluent concentration (low-end scenario) were below the MDL for a pollutant, then zero reduction was assumed. The instances where existing permit limits were more stringent than CTR-based effluent limits for a pollutant would also contribute to the small loadings reductions.

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	0.0	0.0	NA
Chlordane	0.0	0.0	0.0
Copper	4.5–25.2	<0.1–2.5	0.1– 0.8
DDT	0.0–68.8	0.0–6.9	0.0–2.1
Dieldrin	0.0	0.0	0.0
Dioxin	0.0	0.0	NA
Endosulfan	0.0–18.3	NA	0.0–0.6
Endrin	0.0–9.3	NA	0.0–0.3
Fluoranthene	0.0–0.6	0.0–0.1	NA
Fluorene	0.0	0.0	NA
HCH-alpha	NA	NA	NA
HCH-beta	NA	NA	NA
HCH-gamma	NA	NA	NA
Heptachlor Epoxide	0.0	0.0	NA
Heptachlor	0.0	0.0	NA
Hexachlorobenzene	0.0–96.1	0.0–9.6	0.0–2.9
Mercury	51.7– 80.4	0.5–8.0	1.6–2.4
Nickel	0.0–29.5	NA	0.0–0.9
PCBs	0.0	0.0	0.0
Pyrene	NA	NA	NA
Selenium	0.0–13.2	NA	0.0–0.4
Silver	10.0–50.5	0.1–5.1	NA
Toxaphene	0.0	NA	NA
Zinc	0.0–15.8	0.0–1.6	0.0–0.5

[1] Source: U.S. EPA (1997). Range based on low- and high-end cost scenarios.
[2] Calculated by multiplying column 1 by the estimated point source attribution (1–10 percent).
[3] Calculated by multiplying column 1 by the estimated point source attribution (3 percent).

Similarly, of the contaminants with HQs that exceed one (PCBs, dioxin, and mercury for high consumption San Francisco Bay anglers, and PCBs and mercury for high consumption freshwater anglers), EPA estimated a loadings reduction for only one (mercury).

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. The lower bound estimate of reductions in statistical cancer cases is zero because the lower bound estimates of statewide loadings reductions for all carcinogens is zero. Based on an estimated value of a statistical life of $2.5 million to $9.0 million (American Lung Association, 1995) and assuming all cancers are fatal, potential human health benefits (cancer) to recreational anglers attributable to the rule are $0.0 to $5.3 million per year.

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}	1.80×10^{-4} – 1.84×10^{-4}	9.02×10^{-4} – 9.20×10^{-4}
Freshwater Resources				
Total[3]	1.51×10^{-4}	7.60×10^{-4}	1.33×10^{-4} – 1.51×10^{-4}	6.65×10^{-4} – 7.60×10^{-4}

[1] Range based on estimate of reductions in fish tissue concentration contamination.
[2] Total for 11 contaminants listed on Exhibit 8-7.
[3] Total for 8 contaminants listed on Exhibit 8-9.

EXHIBIT 8-13. POTENTIAL HUMAN HEALTH BENEFITS OF REDUCING CANCER AFTER IMPLEMENTATION OF THE CTR TO RECREATIONAL ANGLERS[1]

Waterbody	Annual Reduction in Cancer Cases	Annual Monetized Benefits (millions of 1996 dollars)
San Francisco Bay	0.0	$0.0
Freshwater Resources	0.0–0.6	$0–$5.3

[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). Range based on estimate of reductions in fish tissue concentration contamination.

Exhibit 8-14 presents the potential effect of the CTR on systemic risks for recreational anglers. For anglers with average consumption rates, EPA estimated that the HQ for mercury (which is below 1 at baseline) will be reduced for both San Francisco Bay and freshwater anglers. For anglers with high consumption rates (90th percentile), the HQ for mercury will be reduced to between 0.67 and 1.80 for San Francisco Bay, and to between 0.59 and 1.46 for freshwater resources. Thus, for high consumption anglers, the HQ remains less than 1 under the high-end cost scenario. EPA also estimated that the HQs for PCBs and dioxin will remain less than 1 for both average and high end consumers.

8.1.6 Uncertainties and Limitations

EPA's human health risk assessment is subject to the following limitations:

- 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

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**	**2.26**	**11.31**
Mercury	0.75	**3.77**	0.13–0.36	0.67–**1.80**
Dioxin	0.51	**2.54**	0.51	**2.54**
Freshwater Resources				
PCBs	**1.40**	**7.02**	**1.40**	**7.02**
Mercury	0.62	**3.12**	0.12–0.29	0.59–**1.46**

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

(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 percent of Asians and 44 percent of Hispanics.

- Risks were based on tissue contaminant levels measured in raw fish fillets. One study (OEHHA, 1991) found that DDT concentrations may decrease by 20 to 80 percent 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. The 90th percentile consumption rate covers people consuming higher than average consumption.

8.2 RECREATIONAL ANGLING BENEFITS

The above section described the potential for human health benefits from implementing the NPDES permit programs to achieve the water quality standard based on criteria established under the CTR. Concerns about the health effects of eating contaminated fish may also 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 consist of two components: fewer fishing trips are taken because of the health concerns and advisories, and the value of trips that continue to be taken is reduced. In addition, as described in Chapter 6, reduced toxic contamination may increase stability, resilience, and overall health of numerous ecosystems, which may translate into higher catch rates and increased angling effort in California. Thus, the potential benefits of implementation of the CTR include an increase in value of the fishing experience and an increase in participation in fishing.

This section provides estimates of these two components of value. Because the analysis is conducted at the statewide level and does not take into account numerous site-specific considerations that will affect the level of benefits from the rule, the results are intended to provide only 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 waterbody 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 DDT, chlordane, dioxin, mercury, PCBs, and selenium (see Exhibit 8-1). These advisories, and knowledge of toxic contamination in other waterbodies, may affect anglers' enjoyment of the fishing experience. Reductions in DDT, mercury, and selenium discharges to California waters from NPDES facilities are expected to result from implementation of the CTR. Thus, the rule may result in an increase in value to recreational anglers as a result of reducing concentrations of toxics in fish tissue.

EPA found no available studies of the value to California anglers of reducing toxic contamination of surface waters. However, the potential significance of the contamination problem in terms of how present anglers value the fishery is illustrated by a 1992 study of the Wisconsin Great Lakes open water sport fishery (Lyke, 1993). Lyke estimated the value of the fishery to Great Lakes trout and salmon anglers if it were "completely free of contaminants that may threaten human health" to be between 11 to 31 percent of the current 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 (e.g., the fish consumption advisory for San Francisco Bay was issued in 1994; 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 proposed rule in California. In adaptation, EPA looked at what the research might indicate about potential benefits for all California waters affected by toxics, not just those waters under fish consumption advisories.[33]

To transfer the Lyke results, EPA first estimated the number of fishing days in California that occur in toxic-impaired waters, distinguishing between waterbody type (e.g., freshwater river versus saltwater bay fishing days). 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 11 to 31 percent (from Lyke) 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

[33] 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.

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; as described in the EPA, 1997. EPA then analyzed the extent of toxic impairment of California waters based primarily on the State of California's Water Quality Assessment 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 should also be noted that EPA defined "impaired" waters as those monitored and rated by the State of California as medium or poor quality for at least one toxic pollutant or group of toxic pollutants.[34] The State of California has monitored 9 percent of river and stream miles; 54 percent 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 percent of river and stream miles, 19 percent of lake and reservoir acreage, 69 percent of San Francisco Bay, 51 percent of other California bays, 47 percent of estuaries, and 69 percent of saline lakes are "impaired" (U.S. EPA, 1997). EPA assumed for this analysis, maybe conservatively, that only half (50 percent) of the waters that have not been monitored are impaired in the same proportion as monitored waters.[35] EPA also assumed that California has monitored half of bays, estuaries, and saline lakes and then assumed half (50 percent) of the waters that have not been monitored were impaired similar to monitored waters. 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 results in estimates of the number of toxics-affected fishing days. EPA estimated a total of 4.4 million fishing days in toxic-impaired waters in California, of which 2.3 million are associated with freshwater fishing and 2.1 million are associated with saltwater fishing.

EXHIBIT 8-15. BASELINE VALUE OF FISHING DAYS OCCURRING IN TOXIC-IMPAIRED WATERS[1] IN CALIFORNIA ($1996)

	Fishing Days per Year[1]	Percent of Assessed Waters Toxic-Impaired[1,3]	Toxic-Affected Fishing Days[4]	Consumer Surplus per Day	Baseline Value ($ millions)
Freshwater Fishing					
Lakes and Reservoirs	9,678,800	15%	1,416,008	$25–$35	$35.4–$49.6
Ponds	1,534,100	15%	224,439	$25–$35	$5.6–$7.9
Rivers and Streams	6,002,900	10%	621,600	$25–$35	$15.5–$21.8
Subtotal	17,215,800	—	2,262,048	—	$56.6–$79.2
Saltwater Fishing					
Bays					
San Francisco Bay	697,500[3]	69%	486,275	$50–$100	$24.1–$48.1
Other California Bays	1974,350[3]	38%	755,189	$50–$100	$37.8–$75.5

[34] The California Water Quality Assessment database categories of medium and poor translate to the U.S. EPA categories of not 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.

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

Estuaries	1,649,610[3]	35%	581,488	$50–$100	$29.1–$58.1
Saline Lakes	549,870[3]	52%	284,558	$50–$100	$14.2–$28.5
Subtotal	4,871,330	—	2,102,509	—	$105.1–$210.3
Total	22,087,130	—	4,364,557	—	$161.7–$289.4

[1] Source: Based on U.S. FWS (1993) and U.S. EPA (1997).

[2] "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.

[3] Based on a total of 5,498,700 total saltwater fishing days. Assumes 50 percent in bays (e.g., pier fishing), 30 percent on estuaries, and 10 percent 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 (125,000) multiplied by the average days per angler (6.2) from Huppert (1989).

[4] Calculation of toxic-affected fishing days may not be duplicated exactly due to rounding or percent of assessed waters that are toxic-impaired.

Baseline Fishery Value

To estimate the consumer surplus associated with the estimated 4.4 million fishing days, 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 a surplus value for 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

EXHIBIT 8-16. ESTIMATES OF CONSUMER SURPLUS PER FISHING DAY ($1996)

Study	Location/Species	Consumer Surplus Estimate
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

species sought. However, most of the results fall in the range of $50 to $100. This range is also consistent with the average surplus value reported by Walsh et al. for saltwater fishing ($95 per day).

Multiplying toxic-impaired fishing days by the relevant range of consumer surplus per day results in estimates of the baseline value of the fishery (Exhibit 8-15). EPA estimated that the baseline value of these waters in California is currently between $161.7 million and $289.4 million per year.

Potential Benefits

Multiplying the baseline fishery value ($161.7 million to $289.4 million per year) by the increase in value estimated by Lyke (11 to 31 percent) results in potential benefits of between $17.8 million to $89.7 million per year from achieving a "toxic-free" fishery. To estimate the portion of these benefits that might reasonably be attributable to the CTR, EPA multiplied the benefits by the reduction in toxic-weighted loadings expected from the rule (17.6 to 29.5 percent), and by the assumed contribution of point sources to total loadings as developed in Chapter 7.

For San Francisco Bay, saline lakes, and freshwater, EPA applied the attribution assumptions (1 to 10 percent, 3 percent, and 3 percent, respectively) directly, and for other bays and estuaries EPA applied 42 to 64 percent based on the population and land area weighting described in Chapter 7 **(Exhibit 8-17)**. The approach results in potential benefits attributable to the CTR of between $0.6 million and $8.6 million per year.

EXHIBIT 8-17. POTENTIAL RECREATIONAL ANGLING BENEFITS FROM A "TOXIC-FREE" FISHERY ATTRIBUTABLE TO IMPLEMENTATION OF THE CTR (MILLIONS, 1996 DOLLARS PER 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 and Reservoirs	$35.4–$49.6	$3.9–$15.4	17.6%–29.5%	3%	$0.0–$0.1
Ponds	$5.6–$7.9	$0.6–$2.4	17.6%–29.5%	3%	$0.0–$0.0
Rivers and Streams	$15.5–$21.8	$1.7–$6.7	17.6%–29.5%	3%	$0.0–$0.1
Saltwater					
San Francisco Bay	$24.1–$48.1	$2.6–$14.9	17.6%–29.5%	1%–10%	$0.0–$0.4
Other Bays	$37.8–$75.5	$4.2–$23.4	17.6%–29.5%	42%–64%	$0.3–$4.4
Estuaries	$29.1–$58.1	$3.2–$18.0	17.6%–29.5%	42%–64%	$0.2–$3.4
Saline Lakes	$14.2–$28.5	$1.6–$8.8	17.6%–29.5%	3%	$0.0–$0.1
Total	**$161.7–$289.4**	**$17.8–$89.7**	—	—	**$0.6–$8.6**

8.2.2 Value of Increased Participation

In addition to increasing the value of existing angling days, reduced toxic loadings may also increase fishing participation. 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 for two reasons. First, little is known about how decreases in participation vary given different levels of contamination. Where toxic contamination is not publicized or a fish consumption advisory is not posted, toxic-impaired waters may experience no decrease in effort since anglers will 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. Unfortunately, it is difficult to account for substitute sites in estimating benefits since the availability of substitute sites may vary greatly depending on geographical location and the economic status of the affected anglers. However, participation on unaffected waters may actually decrease by shifting participation to the waters improved under implementation of the CTR, thus decreasing existing congestion at unimpaired sites. EPA did not estimate the benefits of reduced congestion or account for the effect of substitute sites in estimating benefits from 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 waterbodies 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 increased stability, resilience, and overall health of numerous ecosystems, which may translate into higher catch rates and increased angling effort. As a result, even if good substitute sites exist for the toxic-affected areas that anglers are aware of, some minimal increase in participation may result from implementation of the CTR on a state-wide basis.

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 of those aware of fish consumption advisories, 17 percent took 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 percent 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 percent 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 percent reduction in trips may be reasonably expected. Because public knowledge of toxic contamination varies across waterbodies, EPA conservatively assumed a 5 percent 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 percent increase in angler participation for the Bay.

Potential Benefits Attributable to the CTR

EPA multiplied the number of toxic-affected fishing days (estimated in Section 8.2.1) by 5 percent to estimate the expected increase in participation, and valued these days using the estimated consumer surplus values shown above. To estimate the portion of these benefits attributable to implementation of the CTR, EPA multiplied by the reduction in loadings and the attribution assumptions developed in Chapter 7. As shown in **Exhibit 8-18**, benefits due to increased participation that may be attributable to California implementing the NPDES permits program to achieve water quality standards based on criteria established in the CTR range from $0.3 to $1.5 million per year ($1996). Because of the uncertainties inherent in the analysis (e.g., not accounting for substitute sites), EPA used zero as a lower bound estimate.

8.3 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 absent carefully designed and executed primary research (i.e., using the contingent valuation method). However, "benefits transfer" techniques can be used to develop a rough approximation of the potential magnitude of these passive use values.

EXHIBIT 8-18. POTENTIAL BENEFITS FROM INCREASED ANGLING PARTICIPATION

	Baseline Toxic-impaired Fishing Days	Additional Fishing Days (5% of Baseline)[1]	Consumer Surplus (per day)	Value of Additional Days (millions of 1996 dollars)	Reduction in Toxic-Weighted Loadings due to the CTR	Assumed Point Source Contribution to Total Loadings	Potential Benefits Attributable to Implementation of the CTR (millions of 1996 dollars)[2]
Freshwater							
Lakes and Reservoirs	1,416,008	70,800	\$25–\$35	\$1.8–\$2.5	17.6%–29.5%	3%	\$0.01–\$0.02
Ponds	224,439	11,222	\$25–\$35	\$0.3–\$0.4	17.6%–29.5%	3%	\$0.00–\$0.00
Rivers and Streams	621,600	31,080	\$25–\$35	\$0.8–\$1.1	17.6%–29.5%	3%	\$0.00–\$0.01
Subtotal	2,262,048	113,102	\$25–\$35	\$2.8–\$4.0	—	—	\$0.01–\$0.04
Saltwater							
San Francisco Bay	481,275	48,128	\$50–\$100	\$2.4–\$4.8	17.6%–29.5%	1%–10%	\$0.00–\$0.14
Other Bays	755,189	37,759	\$50–\$100	\$1.9–\$3.8	17.6%–29.5%	42%–64%	\$0.14–\$0.71
Estuaries	581,488	29,074	\$50–\$100	\$1.5–\$2.9	17.6%–29.5%	42%–64%	\$0.11–\$0.55
Saline Lakes	284,558	14,228	\$50–\$100	\$0.7–\$1.4	17.6%–29.5%	3%	\$0.00–\$0.01
Subtotal	2,102,509	129,189	\$50–\$100	\$6.5–\$12.9	—	—	\$0.26–\$1.42
Total	**4,364,557**	**242,292**	—	**\$9.3–\$16.9**	—	—	**\$0.27–\$1.45**

[1] Angling participation in San Francisco Bay is estimated to increase by 10%.
[2] Totals may not add due to rounding.

8.3.1 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. This review indicated that nonuse values have been estimated to be *at least* half as great as recreational values, and concluded that if passive use values were potentially applicable to a policy action, using a 50 percent approximation was 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 would lead, 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).

In applying a benefits transfer-based rule of thumb such as described above to estimate passive use values from estimates of recreational use benefits, it is important to consider the extent to which the primary research efforts have evaluated resources, and changes in resource conditions, 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 is a change in toxics-related water quality parameters that will reduce the likelihood or severity of impairments in the future.

The studies reviewed in the Fisher and Raucher analysis generally apply to this context. For example, the Mitchell-Carson study examines the potential change in the use attainment of waters nationwide, starting from a baseline in which the predominant share of waters were meeting designated uses, and valued environmental quality changes that brought most remaining waters up to fishable or swimmable goals. 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 for the CTR. For example, the Sanders et al. results (implying a ratio of approximately 1.8 or 1.9, depending on the scenario and results applied) 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 scenario evaluated, coupled with the irreversibility associated with the resource quality change being evaluated, one would anticipate relatively higher ratios of existence and bequest values to direct use values than in a CTR-like setting in which less drastic (though still important) environmental improvements are expected.[36]

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

[36] 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).

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 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 rule of thumb 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 50 percent rule of thumb 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 is between $0.3 million and $5.0 million per year.

Applying the 50 percent 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. Thus, this approach omits passive use values held by households that do not have active recreational anglers of the waters. 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 nonangling households.

8.3.2 Passive Use Value for Nonangling Households

To account for the passive use values held by nonangling households, which includes other water recreators such as boaters, and swimmers, as well as 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 nonangling households. (Since there likely are more than one angler in some households, this assumption is conservative in that it will result in a lower estimate of nonangling households and values.)

As an upper bound estimate of passive use values for nonangling 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 nonangling households are nonuser households, and that they hold lower passive use values compared to angling households. EPA did not find any literature that provides an indication of how much lower these values might be. However, some studies provide information on the relationship between total willingness to pay (WTP) for water quality improvements for users and nonusers.

Willingness to Pay (WTP) Values for Users and Nonusers

EPA found several contingent valuation studies that estimated WTP for users and nonusers of water resources (**Exhibit 8-19**); 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.

EXHIBIT 8-19. 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%
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.

Perhaps the most applicable study, Bockstael et al. (1989), reports WTP to raise Chesapeake Bay water quality from unacceptable to 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 percent of the value for users ($121). Since WTP for users includes a use value, 31 percent likely understates the relationship between passive use values between users and nonusers (if use value is subtracted out for users, passive use values are probably more comparable between the two groups).

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 nonangling households may be as low as one-third of those for angling households. This estimate is supported by Bockstael et al. (1989), although no literature specifically addressing the relationship of passive use values for users and nonusers was found.

To estimate the number of nonangling households, EPA assumed the number of recreational angling households is equivalent to the number of licensed recreational anglers, or 1.5 million (CDFG, 1994). Subtracting this from the total number of households in California (approximately 10.9 million in 1994; U.S. Bureau of the Census, 1995) yields approximately 9.4 million nonangling households. Assuming a passive value of one-third the range of values for angling households yields a range of $0.6 million to $10.4 million per year for all nonangling households.[37]

[37] EPA calculated a per household value for angling households of $0.20–$3.33 per year by dividing the total value estimated in the previous section ($0.3–$5.0 million per year) by the estimated number of angling households (1.5 million). One third of this value is $0.07–$1.11.

Upper Bound Estimate

As an upper bound, EPA assumed that all nonangling households have the same passive use value as angling households. Multiplying the range of values for angling households to account for all households in California results in a range of $1.9 million to $31.3 million.

Using the lower and upper bound estimates of passive use for nonangling households results in a range of $0.6 to $31.3 million.

8.4 SUMMARY OF MONETIZED BENEFITS

A summary of the estimated monetized benefits from implementation of the CTR is provided in **Exhibit 8-20**. Human health benefits are estimated for San Francisco Bay and statewide freshwater resources; all other benefits are estimated statewide.

The *key* omissions, biases, and uncertainties associated with the benefits analysis are shown in **Exhibit 8-21**. 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 under- or overestimated cannot be predicted with accuracy. However, among the key factors described in Exhibit 8-21, 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

EXHIBIT 8-20. SUMMARY OF ANNUAL BENEFITS FROM IMPLEMENTATION OF THE CTR (MILLIONS OF 1996 DOLLARS)[1]

Benefit Category	Annual Value
Human Health (cancer risk)	
San Francisco Bay	$0.0
Other Saltwater Resources	+
Freshwater Resources	$0.0–$5.3
Recreational Angling	
Increased Value of Existing Trips	$0.6–$8.6
Increased Participation[2]	$0.0–$1.5
Passive Use	
Households with Recreational Anglers	$0.3–$5.0
Other Households	$0.6–$31.3
Omitted Benefits[3]	+
Total	$1.5–$51.7 +

[1] The benefits are based on use of the Model 2 baseline analysis. Model 1 baseline analysis produces zero cost and therefore zero benefits.

[2] A lower bound of zero is used because of difficulties in accounting for substitute sites at the statewide level.

[3] 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.

likely to be the most significant contributors to the underestimation of the monetized values presented in Exhibit 8-20:

- Improvements in water-related (in-stream and near stream) recreation apart from fishing. The omission of potential motorized and nonmotorized 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 state-wide. Some of these activities can be closely associated with water quality attributes (notably, swimming). Other of these 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 observation. 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, it is conceivable that these benefits could be appreciable.

EXHIBIT 8-21. 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 673,000 (based on Huppert, 1989, and U.S. FWS, 1993).
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 under- or over-estimated is unknown.

\+ Potential overestimate.
\- Potential underestimate.
? Uncertain impact.

9. COMPARISON OF POTENTIAL BENEFITS TO COSTS

This chapter compares the costs and monetized benefits attributable to implementation of the California Toxics Rule (CTR).

9.1 ESTIMATED COSTS

EPA estimated the annualized cost of implementation of the NPDES permits program to achieve the water quality standards established by the CTR to range from $0 under Model 1 to $14.9 to $86.6 million under Model 2. Annualized costs include capital costs (construction of treatment, waste minimization, treatment process optimization, and regulatory relief), annual operating and maintenance expenses, and monitoring costs. A summary of the estimated low and high cost scenarios is shown in **Exhibit 9-1**.

EXHIBIT 9-1. SUMMARY OF ESTIMATED COSTS OF IMPLEMENTING THE CTR (1996 [FIRST QUARTER] DOLLARS)

		Model 2	
	Model 1	**Low Scenario**	**High Scenario**
Costs to Direct Dischargers:			
Total Capital and Other Annualized Costs	$0	$36,638,098	$240,281,423
Operation and Maintenance Costs (O&M)	$0	$0	$49,140,094
Monitoring Costs	$0	$1,664	$1,664
Costs to Indirect Dischargers (annualized):			
Annual Costs	$0	$9,700,000	$3,200,000
Total Annualized Costs[1]	$0	$14,900,000	$86,600,000

[1] Capital costs are annualized over 10 years at 7 percent. To compute total annualized costs, annualized capital costs are added to O&M costs, monitoring costs, and annualized costs to indirect dischargers.

9.2 COMPARING POTENTIAL BENEFITS TO COSTS

EPA compared the estimated costs of implementing the CTR to its anticipated benefits using two approaches: (1) a direct comparison of annualized costs to benefits, and (2) a comparison of discounted benefits and costs.

9.2.1 Direct Comparison of Annualized Benefits and Costs

A direct comparison of the estimated annualized cost of the CTR to the potential annual benefits shows that the annual benefits range overlaps the range of annualized costs. As shown in **Exhibit 9-2**, annualized costs are $0 under Model 1 and range from $14.9 million to $86.6 million (1996, first quarter) under Model 2, and the portion of annual benefits that can be monetized amounts to $0 under Model 1 and ranges from $1.5 million to $51.7 million under Model 2.

EXHIBIT 9-2. COMPARISON OF THE POTENTIAL MONETIZED BENEFITS AND COSTS OF IMPLEMENTING THE CTR (MILLIONS OF 1996 [FIRST QUARTER] DOLLARS) (Model 2)

Comparison Method	Monetized Benefits Range		Cost Range	
Direct Annual Comparison[1].	$1.5–$51.7		$14.9–$86.6	
Discounted Benefits and Costs[2]				
Discount Rate	3%	7%	3%	7%
10-Year Phase-In of Benefits	$23–$807	$14–$473	$260–$1,430	$182–$996
20-Year Phase-In of Benefits	$18–$611	$10–$333	$260–$1,430	$182–$996

[1] These monetized costs and benefits are not directly comparable. Since EPA used a number of assumptions that may have overstated costs (especially at the high end of the range) and omitted several benefits categories, benefits and costs may be more commensurate than shown in this table.

[2] Present values over 30 years. Reflects capital costs in years 1 and 16, a 7 percent opportunity cost of capital, and O&M and monitoring costs in years 2 through 30. Benefits are phased in proportionately over 10 and 20 years, and have their full value in the remaining years.

9.2.2 Discounted Benefits and Costs

Exhibit 9-2 also shows that the discounted benefits range also overlaps the range of discounted costs. This method applies a present value social accounting in which the stream of future benefits are discounted to their present values to reflect society's rate of time preference, and compares the benefits to the present value of costs that have been adjusted to reflect the opportunity cost of capital. EPA calculated the streams of discounted benefits and costs assuming discount rates of 3 and 7 percent.

EPA also considered two different phase-in scenarios for benefits to account for the potential delay in realizing benefits because many of the pollutants addressed by the CTR are persistent in the environment. Assuming a 10-year phase-in of benefits, the range of costs and the range of monetized benefits overlap under both discount rate scenarios. However, the overlap of costs and monetized benefits decreases when a 20-year phase-in of benefits is assumed.

9.3 INTERPRETATION AND CONCLUSIONS

Comparison of both annualized benefits and costs and discounted benefits and costs indicates that the monetized benefits of the CTR are of the same general magnitude as the costs. Although monetized benefits fall at the low-end of the cost range, EPA used a number of assumptions that may have overstated costs. Among the assumptions that would tend to overstate costs are:

- Existing facilities that contain effluent limits for toxic pollutants were selected as representative facilities, and were used to extrapolate costs to the universe of facilities. This selection may bias the sample in terms of possibly overstating the number and types of pollutants that require control, thus overestimating the need for WQBELs and costs when extrapolated to the universe of facilities.

- The use of human health criteria based on the consumption of water and organisms (for fresh water discharges only) applies the most stringent criteria for human health protection. This results in more stringent effluent limits which may overestimate potential costs.

- Several assumptions regarding effluent flow, pollutant effluent concentrations, dilution, background pollutant concentrations, and translators (as detailed in Exhibit 4-1) may tend to overestimate costs.

- Capital costs were amortized over 10 years while the useful life of most equipment currently exceeds 10 years. Thus, this assumption also overestimates annual costs.

In addition, although it is difficult to assess the overall impact of omissions, biases, and uncertainties in the estimate of monetized benefits, the overriding factor may be the omission of several benefits categories, resulting in an underestimate of benefits. Thus, benefits and costs may be more commensurate than shown by Exhibit 9-2.

10. ALTERNATIVES ANALYSIS

In conducting an analysis of the potential costs to point source dischargers as a result of implementing the California Toxics Rule (CTR), EPA made a variety of assumptions. To test some of these assumptions, EPA conducted two alternative analyses. First, the impact on costs from changing human health risks from carcinogenic pollutants was considered. Second, the impact on costs from changing the application of criteria for heavy metals was considered.

Sections 10.1 and 10.2 present the methodology and attendant costs from varying the human carcinogenic risks and applying toxic metals criteria in total recoverable form, respectively.

10.1 IMPACT OF HUMAN HEALTH RISK LEVEL

According to the U.S. Environmental Protection Agency's (EPA's) *Water Quality Standards Handbook: Second Edition* (U.S. EPA, 1994), EPA generally regulates carcinogenic toxic pollutants are 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 proposed CTR follows this history, and establishes human health criteria for carcinogens based on a 10^{-6} risk level. The potential costs discussed in Section 4 of this report are based on these proposed 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.

10.1.1 Methodology

Essentially, 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 of the methodology was that the proposed CTR criteria for carcinogens were adjusted to reflect a lower risk level of 10^{-5}. All other assumptions and costing procedures used in the main cost analysis were kept the same.

10.1.2 Results

Exhibit 10-1 summarizes the results of the analysis of lowering the risk level for carcinogens in the proposed CTR. As Exhibit 10-1 shows, the changes in estimated costs and pollutant load reductions based on the lower risk level of 10^{-5} are minimal. Under the low-end scenario, costs decrease by $1.4 million, approximately 25 percent less than the costs based on the higher risk level. Under the high-end scenario, annual costs decrease by less than $1 million, a less than 1 percent 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 3 and 7 percent under the low- and high-end scenarios, respectively.

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 affect primarily organic pollutants.

EXHIBIT 10-1. COMPARISON OF ESTIMATED COSTS IF CTR-BASED WQBELS ARE CALCULATED USING A CANCER RISK LEVEL OF 10^{-5}

Approach	Low-end Scenario		High-end Scenario	
	Estimated Annual Costs ($Millions)	Load Reductions (10^6 lbs-eq/yr)	Estimated Annual Costs ($Millions)	Load Reductions (10^6 lbs-eq/yr)
CTR Cost Analysis	$5.2	0.63	$83.4	7.02
Alternative Analysis	$3.8	0.61	$82.8	6.50

Note: All costs are in first quarter 1996 dollars.

10.2 IMPACT OF METAL TRANSLATORS

The criteria for metals in the proposed rule are in the dissolved form. The use of dissolved 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 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 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 which 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 choose this conservative approach to derive permit limits. Thus, a sensitivity analysis was performed. This analysis is described below.

10.2.1 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. CTR-based WQBELs were calculated using the same methods described in Section 4, except that total recoverable criteria were used in place of dissolved criteria for metals.

10.2.2 Results

The results of this analysis show that costs may be sensitive to the translator chosen by the State. **Exhibit 10-2** shows the expected costs and load reductions using conversion factors as the translators.

EXHIBIT 10-2. COMPARISON OF POTENTIAL COSTS IF CTR-BASED WQBELS ARE CALCULATED USING CRITERIA EXPRESSED AS TOTAL RECOVERABLE

	Low-end Scenario			High-end Scenario		
Approach	Estimated Annual Costs ($Millions)	Load Reductions (10^6 lbs-eq/yr)	Cost Effective-ness ($/lb-eq)	Estimated Annual Costs ($Millions)	Load Reductions (10^6 lbs-eq/yr)	Cost Effective-ness ($/lb-eq)
CTR Cost Analysis	$5.2	0.63	8	$83.4	7.02	12
Alternative Analysis	$34.7	1.03	34	$153.7	9.26	17

Note: All costs are in first quarter 1996 dollars.

As Exhibit 10-2 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-end scenario are almost $35 million per year, a six-fold increase over the estimates in the low-end base analysis. Under the high-end scenario, total costs are estimated to be more than $153 million per year, almost double the cost estimates in the base analysis. Potential load reductions are estimated to increase by approximately 60 percent over the low-end base case and by over 30 percent under the high-end 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-end scenario is $34 per toxic pound-equivalent removed compared to $8 per toxic pound-equivalent removed in the base analysis. The cost-effectiveness of the new high-end scenario is $17 per toxic pound-equivalent removed compared to $12 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 a metal effluent limit, 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.

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