TECHNOLOGICAL FEASIBILITY OF PROPOSED WATER QUALITY CRITERIA FOR NEW JERSEY

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Table of Contents

Appendices

List of Exhibits

List of Acronyms

BMP	Best management practice
B.O.	Biological opinion
CARP	Contamination Assessment and Reduction Project
CSO	Combine sewer overflow
DDT	Dichlorodiphenyl-trichloroethane
DOE	Department of Energy
DRBC	Delaware River Basin Commission
EMDL	Estimated method detection limit
EML	Estimated minimum level
EPA	Environmental Protection Agency
ESA	Endangered Species Act
FWS	Fish and Wildlife Service
GAC	Granular activated carbon
GLWQI	Great Lakes Water Quality Initiative
mgd	Million gallons per day
NADP	National Atmospheric Deposition Program
NJDEP	New Jersey Department of Protection
NPDES	National Pollutant Discharge Elimination System
O&M	Operation and maintenance
ORNL	Oak Ridge National Laboratory
P2	Pollution prevention
PCBs	Polychlorinated biphenyls
PCS	Permit Compliance System
POTW	Publicly owned treatment work
RO	Reverse osmosis
SDP	Surface deactivation and partitioning
SIC	Standard industrial classification
TMDL	Total maximum daily load
TSD	Technical Support Document
TSS	Total suspended solids
UV	Ultraviolet
WLA	Wasteload allocation
WQS	Water quality standards

Executive Summary

The New Jersey Department of Environmental Protection (NJDEP) is proposing revised criteria for mercury, polychlorinated biphenyls (PCBs), and dichlorodiphenyl-trichloroethane (DDT) for the protection of wildlife in surface waters in the State. Because the proposed criteria reflect extremely low concentration levels, and available analytical methods for two of the pollutants (DDT and PCBs) cannot detect at such low levels, there are questions regarding the feasibility of achieving the criteria. Thus, this report provides a review of the performance of existing treatment technologies for reducing these pollutants in wastewater.

Background

In December 1993 (25 NJR 5569), the NJDEP adopted revised surface water quality standards (WQS). EPA proposed approving the surface WQS (with the exception of the human health-based criteria for PCBs), and requested informal consultation with the U.S. Fish and Wildlife Service (FWS) regarding this action. On June 26, 1996, the FWS issued its Biological Opinion (B.O.), concluding that EPA's approval of the WQS is "...not likely to jeopardize the continued existence of the bald eagle, the American peregrine falcon, and dwarf wedgemussel. Further, no critical habitat will be affected, since none has been designated for these species." However, the FWS concluded that certain of the WQS provisions will result in an incidental take of these species, although it could not quantify this take. As a result, the FWS required EPA, in order to minimize any take, to implement certain "reasonable and prudent measures," which include the establishment of wildlife criteria for PCBs, DDT, and mercury criteria at levels that will minimize adverse effects on the bald eagle and the peregrine falcon.

Exhibit ES-1 summarizes the current and proposed WQS for mercury, PCBs, and DDT in New Jersey. Current standards vary based on water body classification.

Exhibit ES-1. Current and Proposed New Jersey Water Quality Criteria for Mercury, PCBs, and DDT

Pollutant	FW2 Criteria[1] (ng/L)			SC and SE Criteria[2] (ng/L)			Proposed Wildlife Criteria (ng/L)
	Acute	Chronic	Human Health	Acute	Chronic	Human Health	
Mercury	NA	NA	144	NA	NA	146	0.53
PCBs	NA	14	1.7	NA	30	1.7	0.072
DDT	1,100	1.0	0.588	130	1.0	0.591	0.004

Source: New Jersey Surface Water Quality Standards (NJAC. 7:9B) and U.S. FWS, et al. (2001).

1. FW2 waters are those fresh water that are not designated FW1 (exceptional waters) or Pinelands.
2. SC waters are coastal saline waters, and SE waters are estuarine saline waters.

Although the approved analytical method for mercury (Method 1631) has a minimum level sufficiently low to determine compliance with an end-of-pipe criterion of 0.53 ng/L (i.e., 0.5 ng/L), the current analytical methods for PCBs and DDT have minimum levels well above the proposed criteria levels. Thus, compliance cannot be demonstrated with these methods for PCBs and DDT.

Technologies for Removing DDT, Mercury, and PCBs from Wastewater

Theoretically (i.e., based on the literature), there are treatment technologies for mercury, PCBs, and DDT that could achieve the proposed criteria end-of-pipe. **Exhibit ES-2** summarizes these technologies. However, there are no data to support the theory, likely because the analytical techniques for measuring pollutant to such low levels have only been approved recently or have not yet been developed, and few, if any, dischargers are required to meet such low levels end-of-pipe.

Exhibit ES-2. Summary of Treatment Technologies for Mercury, PCBs, and DDT

Treatment Process	Feasibility Summary
Mercury	
Reverse Osmosis	Research (NDWC, 1997) indicates that when RO units are operated in series, almost 100% removal of mercury is attainable. However, data are not available to confirm that mercury levels <0.5 ng/L are being meet. Also, disposal of residuals may be extremely costly.
Selective Sorbents	Several sorbents (e.g., Keyle:X, SIR-200) have been proven to produce low mercury levels (e.g., less than 19 ng/L), however, additional testing is required to confirm that this technology is capable of achieving the proposed criterion, especially on a large scale.
Mercury Precipitation	Mercurous sulfides are extremely insoluble in water. Therefore, precipitating mercury with sulfides and then removing the mercurous sulfide compounds with activated alumina should, in theory, achieve the proposed criterion. However, actual performance data is lacking. The process may also form toxic odors and create harmful by-products.
PCBs	
Granular Activated Carbon	Data indicate that GAC, when operated properly, should be able to achieve PCB levels near the proposed criterion. However, sensitive analytical methods are not available to confirm that the proposed criterion can be met. Also, because GAC is an adsorptive process, residuals disposal may be costly.
UV Oxidation	PCBs can be almost completely destroyed in wastewater through oxidation (99.99% destruction efficiency). UV oxidation using strong oxidants or a catalyst is an effective process for destroying PCBs. Testing is necessary to determine which oxidation process best destroys PCBs for a particular type of wastewater. However, sensitive analytical methods are not available to confirm that the proposed criterion can be met.
Sonochemical Degradation	Employs same mechanisms as UV oxidation (i.e., radical chemistry) to destroy PCBs, however, ultrasound is much more energy intensive. Also, the technology has not been proven effective for PCBs in wastewater outside of the laboratory.
DDT	

Exhibit ES-2. Summary of Treatment Technologies for Mercury, PCBs, and DDT

Treatment Process	Feasibility Summary
Granular Activated Carbon	Because DDT is similar in behavior and structure to PCBs, GAC, when operated properly, should be able to achieve nondetect levels. However, sensitive analytical methods are not available to confirm that the proposed criterion can be met. Residuals disposal and pH adjustment may also increase treatment costs.
Advanced Oxidation Processes	AOPs destroy DDT so there are no residuals. AOPs can theoretically destroy all DDTs in wastewater, however, sensitive analytical methods are not available to confirm that the proposed criterion can be met.

NA = not available.
mgd = million gallons per day

The performance of these available treatment controls will be extremely facility-specific, depending on factors such as flow, water quality, and the reductions needed for compliance. However, facilities pursuing such controls would have no guarantee that their expenditures would result in compliance.

A more cost-effective strategy may be pollution prevention. Pollution prevention could produce gains toward achieving standards without imposing the costs of unproven end-of-pipe technologies large scale. For municipal wastewater treatment plants in particular, mercury, PCBs, and DDT are not usually generated or introduced during treatment processes (the addition of chlorine contaminated with mercury may be one exception). Generally, mercury enters the plant from indirect dischargers such as dentists, hospitals, laboratories, and schools as well as from human waste and consumer products. For PCBs and DDT, storm water and other sources such as human and food waste are the main sources of contamination. Whereas end-of-pipe treatment removes contaminants from discharges, it does nothing to eliminate the actual source of contamination (e.g., air deposition, commercial dischargers). In comparison, pollution prevention efforts focus on eliminating the remaining sources of the pollutants (e.g., current uses of mercury, sites contaminated from past PCB and DDT manufacturer and use), instead of applying treatment to larger waste streams in which the contaminants are more diluted.

Conclusions

Additional testing of available end-of-pipe treatment technologies is necessary to ensure that installation of a particular technology will achieve the proposed criteria. However, for DDT and PCBs, sensitive analytical methods are not available to confirm that the proposed criteria can be met.

Meanwhile, P2 may result in identifying sources of pollutants to municipal and industrial treatment works, and implementing reduction programs that may go a long way toward achieving criteria end-of-pipe. Also, in some cases, treatment that is not feasible for larger discharges is feasible and effective for small waste flows (i.e., indirect dischargers). Therefore, although compliance with end-of-pipe effluent limits may not be achievable with the identified technologies, dischargers may be able to make substantial progress toward the proposed criteria through P2 and source controls. Also, data demonstrating the treatment technologies can achieve the criteria end-of-pipe may become available in the future.

1. Introduction

The New Jersey Department of Environmental Protection (NJDEP) is proposing to revise criteria for mercury, polychlorinated biphenyls (PCBs), and dichlorodiphenyl-trichloroethane (DDT) for the protection of wildlife in surface waters in the State. Because the proposed criteria reflect extremely low concentration levels, and available analytical methods for two of the pollutants (DDT and PCBs) cannot detect at such low levels, there are questions regarding the feasibility of achieving the criteria. Thus, this report provides a review of the performance of existing treatment technologies for reducing these pollutants in wastewater.

1.1 Reason for the Revised Criteria

In December 1993 (25 NJR 5569), the NJDEP adopted revised surface water quality standards (WQS) and submitted them to U.S. Environmental Protection Agency (EPA) Region 2 for review and approval. EPA proposed approving the surface WQS [with the exception of the human health-based criteria for PCBs] and requested informal consultation with the U.S. Fish and Wildlife Service (FWS) regarding this action, pursuant to the Endangered Species Act of 1973 (ESA), as amended (16 U.S.C. 1531 *et seq*.). EPA determined that the revised WQS were not likely to adversely affect Federally listed threatened and endangered species in New Jersey and requested FWS concurrence with this determination.

On June 26, 1996, the FWS issued its Biological Opinion (B.O.) of EPA's proposed approval of these revisions to the WQS. In the B.O., the FWS concluded that EPA's approval of the WQS is "...not likely to jeopardize the continued existence of the bald eagle, the American peregrine falcon, and dwarf wedgemussel. Further, no critical habitat will be affected, since none has been designated for these species." However, the FWS concluded that certain of the WQS provisions will result in an incidental take of these species, although it could not quantify this take. As a result, the FWS required EPA, in order to minimize any take, to implement certain "reasonable and prudent measures," which include the establishment of wildlife criteria for PCBs, DDT, and mercury criteria at levels that will minimize adverse effects on the bald eagle and the peregrine falcon.

EPA, NJDEP, and FWS all agreed to participate in an effort to develop the New Jersey-specific wildlife criteria. The result of this effort is a final document entitled "A Derivation of New Jersey-Specific Wildlife Values as Surface Water Quality Criteria for: PCBs, DDT, Mercury," issued in July 2001. On November 18, 2002, NJDEP proposed to adopt these wildlife criteria.

1.2 Purpose of Analysis

The State of New Jersey is developing procedures for implementing the revised criteria into national pollutant discharge elimination system (NPDES) permits for point sources. Therefore, the State is evaluating the feasibility of achieving the criteria through traditional end-of-pipe controls, as well as through alternative approaches such as pollution prevention (P2). This report provides an assessment of this feasibility.

1.3 Scope of Analysis

This report identifies treatment technologies for municipal treatment plants and industrial facilities that reduce mercury, PCBs, and DDT in wastewater. The information presented here is based on literature on innovative and established technologies. Detailed discussion is provided only for those technologies shown to reduce effluent concentrations to the required levels either in the field, or in theory. In addition, the report provides a discussion of P2 techniques for controlling the source of pollutants in municipal and industrial effluents.

1.4 Organization of Report

Section 2 provides background information for evaluating the feasibility of achieving the proposed criteria. Sections 3, 4, and 5 discuss the available technologies for reducing mercury, PCBs, and DDT in wastewater, respectively. Section 6 discusses alternative reduction strategies for each of the pollutants. Two appendices provide supplemental information: Appendix A provides available data regarding the relative source contributions for each pollutant, and Appendix B discusses cost considerations for the technologies and strategies discussed in the body of the report.

2. Background

This section provides background information for evaluating the technical and economic feasibility of the proposed criteria, including New Jersey WQS, analytical methods detecting mercury, PCBs, and DDT, and NPDES dischargers in New Jersey.

2.1 Water Quality Standards

The existing criteria for mercury, PCBs, and DDT for New Jersey vary based on water body classification. The State has established procedures for implementing these criteria in permits. Implementation guidance outlines procedures for determining whether a discharge may cause, have reasonable potential to cause, or contribute to an exceedance of the criteria and calculating average monthly and maximum daily effluent limits.

2.1.1 Current Criteria

Exhibit 2-1 summarizes the current WQS for mercury, PCBs, and DDT in New Jersey for FW2 SC, and SE waters. FW2 waters are those fresh waters that are not designated FW1 (exceptional waters) or Pinelands waters. SC waters are coastal saline waters, and SE waters are estuarine saline waters.

Exhibit 2-1. New Jersey Surface Water Quality Criteria for Mercury, PCBs, and DDT

Pollutant	FW2 Criteria (ng/L)			SC and SE Criteria (ng/L)		
	Acute	Chronic	Human Health	Acute	Chronic	Human Health
Mercury	NA	NA	144	NA	NA	146
PCBs	NA	14	1.7	NA	30	1.7
DDT	1,100	1.0	0.588	130	1.0	0.591

Source: New Jersey Surface Water Quality Standards (NJAC. 7:9B).

2.1.2 Proposed Criteria

Exhibit 2-2 summarizes the proposed wildlife criteria.

Exhibit 2-2. Proposed New Jersey Wildlife Criteria for Mercury, PCBs, and DDT

Pollutant	Wildlife Criteria (ng/L)
Mercury	0.53
PCBs	0.072
DDT	0.004

Source: U.S. FWS, et al. (2001).

The species of concern for adopting these criteria are the bald eagle and peregrine falcon. Due to its high position on the predator-prey food chain, the peregrine falcon may be significantly more susceptible than other wildlife to adverse impacts from exposure to bioaccumulative pollutants. Proposed criteria reflect values protective of the peregrine falcon (U.S. FWS, et al., 2001).

The majority of the input and exposure parameters, including bioaccumulation and biomagnification factors, on which these wildlife criteria are based come from the Great Lakes Water Quality Initiative (GLWQI). However, U.S. FWS, EPA, and NJDEP revised some of the values (e.g., bioaccumulation factors) based on subsequent studies and New Jersey-specific values for dissolved and particulate organic carbon (U.S. FWS, et al., 2001).

2.1.3 Implementation Procedures

The State's implementation procedures for toxic pollutants are based primarily on EPA's Technical Support Document for Water Quality-based Toxics Control (TSD, EPA 1991), although certain procedures (e.g., effluent flows used to determine limits) are apparently undefined in the State Water Quality Standards (NJAC 7:9B) and associated regulations (NJAC 7:14A-13, NJAC 7:18, NJAC 7:9B).

Reasonable potential is determined from maximum effluent concentrations or the TSD's projected maximum effluent concentrations and instream water concentrations, presumably using the TSD steady state model procedures. Permit limits are derived using TSD procedures (NJAC 7:14A-13.6), most likely using the results of a steady state model to determine the wasteload allocation. Nonetheless, if a calculated limit would not be detectable with available methods, the applicable detection limit would be specified in the permit as the effluent limit (NJAC 7:9B-1.5(e)5.

Mixing zones are generally allowed, although they are prohibited for new discharges of DDT, mercury, and PCBs, among other pollutants (NJAC 7:9B-1.5(h)5,vii), and where the pollutants could impact threatened or endangered species (NJAC 7:9B-1.5(h).iv). Also, TSD procedures do not provide for dilution if the receiving waters exceed the criteria. Thus, new dischargers, dischargers to endangered species critical habitat, or dischargers to waters impaired for these pollutants, would be required to meet the proposed criteria end-of-pipe.

For discharges to tidal water bodies, mixing zones for chronic and human health criteria are limited to one fourth the distance between the discharge port closest to the shoreline and the shoreline, or 100 meters, whichever is greater. For discharges to nontidal water bodies, mixing zones are based on receiving water and effluent flows. Receiving water flows are specified, for mixing zone calculations, as the flow which is exceeded 75% of the time for the appropriate 'period of record' for both DDT and PCBs (NJAC 7:9B-1.5(c)2), subject to other constraints in NJAC 7:9B-1.5(h). The 7Q10 (MA7CD10) receiving water flow is specified for mercury (NJAC 7:9B-1.5(c)2), also subject to other constraints. These procedures apply to human health and aquatic life criteria, not to wildlife criteria. The State's procedures imply, but do not explicitly state, that similar averaging periods (e.g., minimum flow exceeded 75 % of the time and 7-day average flow) apply to effluent flows in determining available dilution.

2.2 Method Detection Limits and Minimum Levels

This section describes the available analytical methods for detecting mercury, PCBs, and DDT. Note that methods capable of detecting the pollutants to the proposed criteria may not yet be available.

2.2.1 Mercury

Currently, the method detection limit or minimum level for mercury is sufficiently low to determine compliance with an end-of-pipe criterion of 0.53 ng/L. The promulgated minimum level for EPA Method 1631 (EPA-821-R-02-019, August, 2002) is 0.5 ng/L.

2.2.2 PCBs

For PCBs, the currently approved method for water (Method 608) is for mixtures of PCB congeners, and a method detection limit is specified only for PCB 1242 (65 ng/L). This detection limit is considerably above the proposed 0.072 ng/L PCB criterion, and compliance cannot be demonstrated with this method. In addition, Method 608 would most likely not have an appropriate detection limit to determine compliance for other PCB congener mixtures. Thus, using this method to determine compliance is unlikely to result in reductions in PCB discharges over those based on the current criterion of 0.170 ng/L – operationally, the method detection limit could be incorporated into permit limits, even though the method detection limit is considerably above both the current and proposed PCB criteria.

EPA Method 1668A (U.S. EPA, 1999) is not currently approved in 40 CFR 136, however, it may be used for “data gathering and monitoring associated with the Clean Water Act.” As shown in **Exhibit 2-3**, the estimated method detection limit (EMDL) ranges from 0.058 ng/L for monochlorobiphenyls to 0.450 ng/L for nonachlorobiphenyls.

Exhibit 2-3. Estimated Method Detection Limits and Minimum Levels for Method 1668A

PCB Congener Category	EMDL (ng/L)	EML (ng/L)
All Congeners	0.190	0.535
Monochlorobiphenyls	0.058	0.137
Dichlorobiphenyls	0.066	0.195
Trichlorobiphenyls	0.124	0.250
Tetrachlorobiphenyls	0.129	0.410
Pentachlorobiphenyls	0.174	0.572
Hexachlorobiphenyls	0.206	0.648

Exhibit 2-3. Estimated Method Detection Limits and Minimum Levels for Method 1668A

PCB Congener Category	EMDL (ng/L)	EML (ng/L)
Heptachlorobiphenyls	0.299	0.762
Octachlorobiphenyls	0.370	0.900
Nonachlorobiphenyls	0.453	1.00
Decachlorobiphenyls	0.153	0.500

Source: U.S. EPA (1999).
EMDL = estimated method detection level.
EML = estimated minimum level.

The average estimated method detection limit for the 12 World Health Organization-designated toxic PCB congeners using Method 1668A is 0.149 ng/L, and the estimated minimum level (EML) is 0.473 ng/L. Thus, even with this method, compliance could not be demonstrated for either the existing or the proposed PCB criterion. Nevertheless, requiring dischargers to use this method could result in requirements for discharge reductions, since it is about two orders of magnitude more sensitive than the currently approved method (Method 608).

2.2.3 DDT

The currently approved method for DDT in water (Method 608) has a method detection limit of 12 ng/L (Appendix A to 40 CFR 136), about three orders of magnitude higher than the proposed criterion of 0.004 ng/L. This detection limit would not allow dischargers to demonstrate compliance with either the existing or proposed criterion. Operationally, limits based on the new criterion could be set to the estimated minimum level with either the existing or proposed criterion.

2.3 NPDES Dischargers

EPA's Permit Compliance System (PCS) database indicates that there are 3,793 facilities permitted to discharge to New Jersey surface waters. EPA classifies approximately 96% of these facilities as minor dischargers (facilities discharging less than 1 million gallons per day (mgd) and not likely to discharge toxic pollutants in toxic amounts). **Exhibit 2-4** provides a summary of all permits by industry and permit type.

Exhibit 2-4. NPDES Permitted Direct Dischargers in New Jersey[1]

Standard Industrial Classification	Number of Facilities[2]	
	Majors	Minors
Agriculture, Forestry, and Fishing		
02 Agricultural Production – Livestock and Animal Specialties	-	1
09 Fishing, Hunting, and Trapping	-	1
Mining		
10 Metal Mining	-	2

Exhibit 2-4. NPDES Permitted Direct Dischargers in New Jersey[1]

Standard Industrial Classification		Number of Facilities[2]	
		Majors	Minors
13	Oil and Gas Extraction	-	1
14	Nonmetallic Minerals	-	22
Construction			
15	General Contractors and Operative Builders	-	1
17	Construction - Special Trade Contractors	-	4
Manufacturing			
20	Food and Kindred Products	1	53
22	Textile Mill Products	1	6
23	Apparel and Other Finished Products	-	1
24	Lumber and Wood Products	-	24
25	Furniture and Fixtures	-	2
26	Paper and Allied Products	3	26
27	Printing and Publishing	-	10
28	Chemicals and Allied Products	22	196
29	Petroleum and Coal Products	6	35
30	Rubber and Miscellaneous Plastics Products	-	76
32	Stone, Clay, and Glass Products	1	74
33	Primary Metal Industries	3	47
34	Fabricated Metal Products	-	83
35	Industrial Machinery and Equipment	-	48
36	Electronic and Other Electronic Equipment	1	24
37	Transportation Equipment	-	27
38	Measuring, Analyzing, and Controlling Instruments	1	7
39	Miscellaneous Manufacturing Industries	-	10
Transportation and Public Utilities			
40	Railroad Transportation	-	6
41	Local and Interurban Passenger Transit	-	35
42	Trucking and Warehousing	-	149
43	United States Postal Service	-	6
44	Water Transportation	-	84
45	Transportation by Air	1	48
46	Pipelines, except Natural Gas	-	6
47	Transportation Services	-	9
48	Communications	-	1
49	Electric, Gas, and Sanitary Services; except 4952	14	85
4952	Sewerage Services (POTWs)	100	93
Wholesale Trade			
50	Wholesale Trade – Durable Goods	-	174

Exhibit 2-4. NPDES Permitted Direct Dischargers in New Jersey[1]

Standard Industrial Classification		Number of Facilities[2]	
		Majors	Minors
51	Wholesale Trade – Nondurable Goods	2	58
	Retail Trade		
54	Food Stores	-	2
55	Automotive Dealers and Service Stations	-	65
57	Home Furniture, Furnishings, and Equipment	-	1
58	Eating and Drinking Establishments	-	1
59	Miscellaneous Retail	-	6
	Finance, Insurance, and Real Estate		
62	Security and Commodity Services	-	1
65	Real Estate	-	13
	Services		
70	Hotels and Other Lodging Places	-	3
73	Business Services	-	2
75	Auto Repair, Services, and Parking	-	9
76	Miscellaneous Repair Services	-	4
79	Amusement and Recreational Services	-	5
80	Health Services	1	7
82	Educational Services	-	30
83	Social Services	-	1
86	Membership Organizations	-	2
87	Engineering, Accounting, Research, Management, and Related Services	1	14
88	Private Households	-	1
89	Miscellaneous Services	-	1
	Public Administration		
91	Executive, Legislative and General Government, except Finance	-	1
92	Justice, Public Order, and Safety	-	3
95	Administration of Environmental Quality and Housing Programs	1	2
97	National Security and International Affairs	-	12
99	Nonclassifiable Establishments	-	3
	No SIC Code (blank in PCS)	-	1,910
	Total	159	3,634

'-' = none

1. Source: U.S. EPA (2004a). Accessed on July 1, 2004.
2. Facilities are differentiated based on major/minor classification in EPA's PCS database.

Due to the increased stringency of the proposed criteria, dischargers not traditionally known for discharging toxic pollutants, may now have reasonable potential to cause or contribute to an exceedance of water quality standards. For example, by definition, minor dischargers generally discharge less than 1 mgd and do not discharge toxic pollutants in toxic amounts. However, these proposed criteria redefine toxic amounts. A minor municipal facility that treats mostly residential waste is very likely to have mercury levels above the proposed wildlife criteria, because mercury can be found in numerous household products and foods that are either discarded down the drain or excreted in human waste. Therefore, it is possible that a number of dischargers could get new, more stringent effluent limits for mercury, PCBs, and DDT.

3. Mercury Treatment Controls

This section provides a discussion of the technical feasibility associated with treatment technologies applicable to mercury. Limited data exist on treatment technologies that can consistently achieve an effluent mercury concentration of 0.5 ng/L, most likely because EPA only recently approved analytical techniques for measuring mercury to such low levels. Also, few, if any, dischargers are required to meet such low levels end-of-pipe.

For additional considerations, including the relative source contributions of mercury, and the potential costs of available treatment technologies, see the appendices.

3.1 Reverse Osmosis

Theoretically, reverse osmosis (RO) can remove nearly all inorganic contaminants from water, including mercury (NDWC, 1997). The process removes contaminants from water using a semipermeable membrane that permits water, and not dissolved compounds, to pass through its pores. The contaminated water is subjected to high pressure that forces pure water through the membrane, leaving pollutants behind in a brine solution (NDWC, 1997).

Properly operated RO units can achieve 96% removal of mercury, however, RO units operated in series are capable of nondetectable or almost 100% mercury removal (NDWC, 1997). RO has several advantages, including:

- Removal of nearly all contaminant ions and most dissolved non-ions
- Relatively insensitive to flow and total dissolved solids levels
- Also removes bacteria and particulates (NDWC, 1997).

Although RO removal rates suggest that the proposed mercury criterion is achievable, EPA does not have data that indicate that meeting levels below 0.5 ng/L on a consistent basis is possible. This may be due to the fact that testing has not been done with the newer, more sensitive analytical methods, or that no facility has been required to reduce mercury to such low levels. The use of RO for treating wastewater is limited and, therefore, performance data is scarce. Because removal efficiencies are less than optimal when treating influents with low contaminant concentrations, performance testing using influent concentrations in the ppb or ppt range should be conducted on various types of wastewater to ensure that RO is capable of producing effluent with mercury concentrations below 0.5 ng/L.

However, even if such low mercury levels are possible with RO, it is important to consider the potential disadvantages and limitations of this technology, including:

- High capital and operating costs (which increase exponentially with decreasing desired effluent concentrations)
- Management and disposal of residuals (brine and media) is potentially costly and problematic

- Expensive pretreatment may be required to maintain the high treatment efficiency levels needed to achieve extremely low levels of pollutants (e.g., carbon prefilter for the reduction of chlorine, which can damage some types of chlorine sensitive RO membranes, sediment prefilter to ensure that fine suspended materials do not clog the membrane)
- Membranes are prone to fouling (NDWC, 1997)
- May not be feasible for treatment of large flows
- Does not remove pollutants from the environment, rather transfers them to another media (pollutants are concentrated in the reject discharge or brine).

The most limiting of these disadvantages, especially for larger systems, is the management and disposal of residuals. In most RO systems, about 10% to 25% of influent water will be rejected as a concentrated brine solution (U.S. EPA, 1996). For a 10 mgd facility running 2 RO units in series, this translates into disposal of about 2 mgd to 5 mgd of residual waste; this brine contains the contaminants removed from the wastewater, and must be treated prior to disposal. Residual disposal options include:

- Land disposal (e.g., evaporation pond, underground injection)
- Land application
- Treatment and recycle (U.S. EPA, 2002).

However, the first two options require that the waste be classified as nonhazardous. Otherwise, the brine must be disposed of as a hazardous waste, which can be extremely costly, especially for larger facilities. The land disposal option may require a large amount of land, and the land application or reuse option depends on demand for such water as well as the contaminant concentrations. The treatment and recycle option produces two waste streams: a cleaner liquid waste that can be recycled to the headworks, and a more concentrated solid waste stream that can be disposed of similar to sludge. Brine treatment options include settling, gravity thickening, chemical precipitation, or flash evaporation (U.S. EPA, 2002).

In addition, these disposal options do not guarantee that the mercury in the residuals will not be re-released into the environment. The mercury is simply being transferred from the discharged effluent to a residual brine. If the brine is sent to an evaporation pond, the mercury could evaporate to the atmosphere and be redeposited into rivers and lakes.

3.2 Selective Sorbents

There are a number sorbents designed to remove dissolved and ionic mercury from wastewater. Most of these sorbents rely on absorption and chemical bonding of ions through chelation and/or ion exchange. Absorptive technologies work best when waste streams are pretreated to remove suspended and dissolved solids to enhance mercury removal and eliminate competing ions.

Selective resins are typically used in packed columns, and consist of four operations: service, backwash, regeneration, and rinse (U.S. EPA, 1997). Note that some sorbents cannot be regenerated, so once they are spent (i.e., there is no more capacity to absorb mercury), they are disposed of and replaced with fresh media. Advantages include:

- Relatively insensitive to variability in influent quality
- Can theoretically achieve proposed criterion, under certain conditions (e.g., a number of columns in series, extensive pretreatment)
- Contaminants can be recovered from most resins upon regeneration.

Use of selective sorbents is also not without its disadvantages:

- Management and disposal of spent media or regenerated brine can be costly
- Effluent quality may be highly variable (depending on influent concentrations or sorbent loading)
- To achieve low levels of mercury, extensive pretreatment may be necessary
- May need multiple units in series to achieve desired effluent quality, which increases costs.

Historically, the use of sorbent resins for mercury removal has been limited to anion resins applied to industrial wastewater containing inorganic mercury with high concentrations of chloride. The chloride produces negatively charged mercury chloride complexes that are attracted to the positively charged resin. Cationic resins containing thiol groups can also be used to remove mercury because they have a high selectivity for mercury (U.S. EPA, 1997). Another resin that can be used to remove mercury is a chelating resin, which, when attached to a complexing group, can bond negatively charged metal ions to form a ring (or chelate) (U.S. EPA, 1997).

Limited performance data on selective absorbent resins indicate that mercury can be removed to low ng/L levels. Few studies have used analytical methods sensitive enough to measure below 200 ng/L. However, studies conducted at two U.S. Department of Energy (DOE) sites used clean analytical methods (e.g., Method 1631), and focused on the removal of mercury from mg/L levels to low ng/L levels (Klasson, et al., 1999, 2000; TN & Associates 1998).

Klasson et al. (1999) treated wastewater from the Oak Ridge National Laboratory (ORNL) with various selective sorbents to reduce effluent mercury concentrations. For a portion of the wastewater, treatment prior to entering the sorbent columns consisted of dual media filtration, air stripping, and granular activated carbon, and the remaining water was treated with precipitation, flocculation, and settling. The waste streams were combined prior to entering the sorptive media columns. Influent mercury to the columns ranged from 40 to 625 ng/L. Overall, the thiol-based sorbents such as Keyle:X and SIR-200 performed the best. Effluent levels were below 19 ng/L about 50% of the time with the Keyle:X resin when the water was chlorinated prior to entering the column to oxidize the mercury to a more treatable form. The removal efficiency of the selective resins was not specified; however, the report indicates that efficiencies vary considerably over time, most likely the result of changes in the speciation of the mercury in the wastewater (Klasson, et al., 2000).

None of the tested sorbents at the second DOE site, the Y-12 Plant, reduced mercury concentrations to less than 12 ng/L from an average influent concentration of about 1,100 ng/L. Three sorbents, SIR-200, Keyle:X, and GT-73 were found to reduce mercury concentrations to less than 51 ng/L at flow rates of 0.3 bed volumes per minute to 1 bed volume per minute (TN & Associates, 1998). Removal efficiencies were approximately 95%, on average.

While the sorbents tested above were capable of reducing mercury concentrations significantly, they did not demonstrate the ability to produce effluents that would comply with the proposed mercury criterion. In fact, an additional 96% to 99% of the mercury would need to be removed to achieve mercury levels below 0.5 ng/L. However, it is important to note that at both sites, effluent mercury concentrations appeared to be strongly dependent on influent mercury concentrations (lower influent concentrations resulted in lower effluent concentrations), and not on the number of bed volumes treated or sorbent loading rate (Klasson, et al., 2000). Therefore, effluent concentrations could decrease as influent concentrations decrease (e.g., through source control efforts or operating columns in series), although this result would need to be substantiated with further testing of waters with lower concentrations.

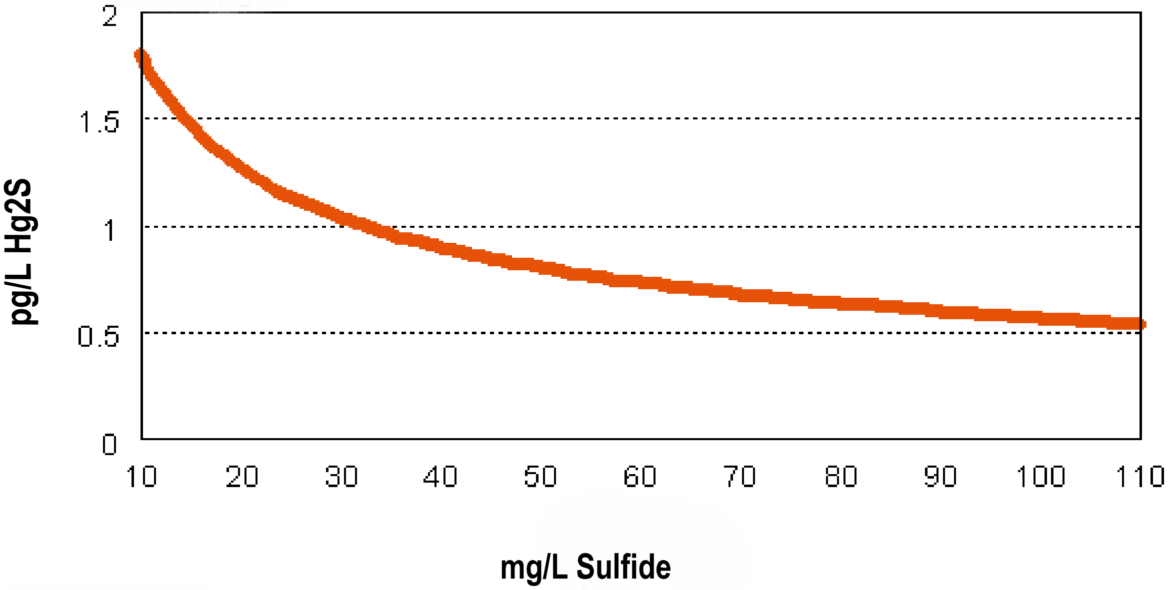
3.3 Mercury Precipitation

Mercury may be present in three forms in wastewater: metallic mercury (Hg^0, elemental), ionic mercury (Hg^{+2}, usually present as complexes and chelates with ligands), and organic (e.g., methylmercury). These species of mercury may bind to particulate matter in the wastewater to form physical agglomerates containing mercury that more readily settle out (MASCO, 1997). Particulate mercury is often the predominant form in wastewater streams (MASCO, 1997). However, in order to achieve mercury levels in the low ng/L range, the dissolved and elemental forms of mercury, as well as particulate mercury need to be removed.

The removal of mercury from wastewater to levels less than 0.5 ng/L may be achieved using chemical precipitation and adsorption. Ionic mercury readily combines with phosphate or sulfide to precipitate into mercury phosphates or sulfides. However, when mercury is precipitated as sulfide, high mercury residuals are often observed. This effect is due to reduction of mercury to elemental mercury by sulfides. Once in the elemental form, mercury does not form the insoluble sulfide. Elemental mercury is soluble in water at about 25 µg/l, well above the proposed criterion. Therefore, this residual mercury must by oxidized to the ionic form and retreated with sulfide to achieve low residual concentrations (West General, 2004). Such a conversion can be done by adding an oxidant such as chlorine. [This process is similar to the addition of chlorine to enhance the performance of the Keyle:X resin tested at the DOE site discussed above. The chlorine oxidizes the mercury to a more readily adsorbed form (Klasson et al., 2000).]

Exhibit 3-1 shows the solubility of mercurous sulfide in water. Facilities may choose to add sulfide in the form of iron sulfide or hydrogen sulfide. Adding iron sulfide is more favorable than adding hydrogen sulfide because the generation, handling, and application of hydrogen sulfide may be challenging due to its highly toxic nature. Ferrous sulfide should be added in doses of about 1 to 2 mg/L, concentrations much greater than the µg/L mercury concentrations found in effluent waters. The solubility curve for mercurous sulfide in the exhibit shows that very low levels of mercury (less than 2 pg/L, or 0.002 ng/L) remain in the water when mercury is precipitated as sulfide.

Exhibit 3-1. Solubility of Mercury (I) Sulfide in Water



Once these sulfide precipitants are formed, additional treatment is needed because a portion of the mercury precipitants will be fine particles that are only minimally different in density from water, and therefore, will not settle out (U.S. EPA, 1980). U.S. EPA (1997) indicates that only levels in the µg/L range can be achieved with precipitation alone. The most efficient way to remove this trace level solid mercurous sulfide is to attach it to a non-leaching media such as activated alumina. Activated alumina adsorption is a physical/chemical process by which ions in solution are removed by the available adsorption sites on an oxide surface. Activated alumina is used primarily in packed beds where feed water is passed continuously through one or more beds (U.S. EPA, 2000a). The precipitant will adsorb onto the activated alumina as the wastewater passes through it, thus, reducing the mercury concentration of the final effluent. Note that, although the chemistry of the process indicates that mercury will be effectively reduced, bench- and pilot-scale testing would be necessary to confirm that the necessary levels can be met for a particular wastewater, and to determine the operating conditions required for optimal performance.

This process is also not without its disadvantages and limitations. The addition of sulfides to wastewater may generate toxic sulfide residual levels. Extensive testing would need to be conducted to ensure minimal residual levels, otherwise the excess sulfides would have to be removed prior to discharge. Another concern associated with the addition of sulfides is the formation of hydrogen sulfide gas. If the pH of the wastewater is too low, hydrogen sulfide gas may be produced causing air toxicity and odor problems (U.S. EPA, 1980). Control of sulfide dose or enclosing and vacuum evacuating vessels may be necessary to prevent this problem.

The oxidation of elemental mercury may also cause problems. The use of chlorine or ozone to oxidize mercury could result in the formation of harmful by-products such as trihalomethanes, haloacetic acids, aldehydes, and carboxylic acids. Ultraviolet (UV) light may be used to oxidize the mercury, however, extensive pretreatment may be necessary to ensure that the UV rays reach the mercury.

Lastly, the most significant limitation to mercury precipitation followed by activated alumina is the limited performance data available. Although in theory this process should remove mercury to the proposed criterion, actual performance data on different wastewater types is necessary to support the science behind the process. The effects that factors such as flow rate, competing contaminants, residuals production, and by-product formation have on the process and design requirements should be investigated prior to implementation for compliance with revised permit limits.

3.4 Summary

There are no field data that prove that any technology is capable of achieving 0.5 ng/L end-of-pipe on a consistent basis. The little field data available shows that levels below 12 ng/L can be achieved with selective sorbents, however, this is still 2 orders of magnitude greater than the proposed criterion. Also, the data are from a pilot, not a full-scale, system. The lack of available performance data for full-scale systems may be due to the fact that few, if any, facilities are required to meet such low levels. The proposed criterion would be the lowest mercury criterion adopted by any State. In addition, analytical methods capable of measuring mercury to less than 0.5 ng/L are relatively new. Any case studies or testing performed prior to development of clean analytical methods would not have been able to measure mercury concentrations below 200 ng/L. Therefore, additional testing is necessary to show that compliance with the proposed criterion is possible with the installation of one of the technologies discussed above.

4. PCB Treatment Controls

PCBs encompass a class of chlorinated compounds that includes up to 209 variations or congeners with different physical and chemical characteristics. PCBs alone are not usually very mobile in subsurface soils or water (FRTR, 2002). In water, a small amount of PCBs may remain dissolved, however, most stick to organic particles and bottom sediments (ATSDR, 2001). PCBs also have low solubility in water which may result in colloidal transport of the pollutant through water. Therefore, influent and effluent testing to determine the nature of PCBs in wastewater may be useful in determining the most appropriate treatment technology for reducing PCBs to the proposed levels.

This section describes the treatment controls that exist for removal or destruction of PCBs from wastewater. Most research on PCB treatment technologies concerns contaminated sediments, soils, groundwater, or industrial waste. However, the following treatment technologies could be used to remove or destroy PCB from wastewater:

- Granular activated carbon (GAC)
- Ultraviolet oxidation processes
- Sonochemical degradation

Note that there is uncertainty as to the exact levels these technologies can treat to, and that treatment effectiveness depends on the amount and influent quality of the wastewater to be treated. For additional considerations, including the relative source contributions of PCBs, and the potential costs of available treatment technologies, see the appendices.

4.1 Granular Activated Carbon

GAC has been used successfully in advanced treatment of municipal and industrial wastewater for years. GAC is primarily used to adsorb relatively small quantities of soluble organics, such as PCBs, and inorganics such as nitrogen, sulfides, and heavy metals (U.S. EPA, 2000).

U.S. EPA (2000b) indicates that PCBs are amenable to absorption using GAC. GAC adsorption occurs when PCB molecules adhere to the internal walls of pores in carbon particles produced by thermal activation (U.S. EPA, 2000b). At municipal wastewater treatment plants, GAC is usually installed prior to disinfection, but after filtration (or some other type of pretreatment) that removes suspended solids in order to prevent fouling and maximize the time before breakthrough occurs (i.e., the point at which the contaminant is no longer absorbed).

Advantages of GAC include the following:

- Proven, reliable technology for removing dissolved organics
- Low space requirements
- Easy to incorporate into existing systems (U.S. EPA, 2000b)
- Lower capital and operating costs than new, less proven technologies

However, GAC is not without its disadvantages, including:

- Under certain conditions, hydrogen sulfide may be generated from bacterial growth, creating odor and corrosion problems
- Wet GAC is highly corrosive and abrasive
- Requires pretreated wastewater with low suspended solids, and little variations in pH, temperature, and flow rate (U.S. EPA, 2000b)
- Management and disposal of residuals (e.g., spend media).

There are limited data on the actual PCB effluent levels that can be obtained using GAC because few dischargers have been required to measure or obtain levels in the ng/L range. Also, analytical methods sensitive enough to measure down to the proposed criterion of 0.072 ng/L have not yet been approved. However, there are data that indicate that GAC, when operated properly, should be able to achieve effluent concentrations near the proposed criterion.

Chemviron Carbon (2004) claims that its activated carbon adsorption systems can achieve removal efficiencies greater than 99% for chlorinated hydrocarbons such as PCBs. This is based on treating contaminated groundwater with only one GAC unit and no pretreatment.

Paquin et al. (1997) compares the PCB removal efficiency of GAC to the PCB removal efficiency of a combination of surface deactivation and partitioning (SDP) media and GAC; the research indicates that the removal efficiency of GAC alone may be as high as 99.99%. This removal percentage is based on an effluent concentration of <100 ng/L and an influent concentration of 740,000 ng/L (Paquin, et al, 1997). Although this influent concentration is considerably higher than typical wastewater influent concentrations, removal efficiencies are unlikely to vary significantly with lower influent concentrations, unless the PCBs attach to extremely fine particles that may pass through the carbon (Sherbondy, 2004). GAC systems should produce nondetectable effluent concentrations using sensitive analytical methods when optimized for PCB removal, regardless of influent concentrations (Sherbondy, 2004).

If it is determined that PCBs are attaching to fine particles, resulting in higher than desired effluent concentrations, a combination of SDP media and GAC can be used. The SDP media system would be used prior to GAC to remove contaminants primarily in the colloidal form (i.e., particles ranging in size from 0.001 μm to 10 μm) in water. PCBs are naturally hydrophobic, but their affinity with water may increase if other competing contaminants lower the surface tension of the water. SDP media reduces the affinity of PCBs for water by increasing the surface tension of the water. This allows the media to capture and retain the PCBs. Using SDP media also increases the useful life of GAC by a factor of two to three, thus lowering operating costs (Paquin et al., 1997).

Therefore, theoretically, GAC should be able to remove virtually all PCBs from wastewater with influent levels typically observed at most facilities. However, these data are based primarily on laboratory bench-scale tests. Additional testing of pilot- and full-scale systems would be necessary to determine the effect influent water quality has on performance, and ensure that GAC can be used for compliance with the proposed PCB wildlife criterion.

4.2 Ultraviolet Oxidation Processes

UV oxidation involves the use of UV light coupled with one, or a combination of, chemical and physical agents that break up existing bonds, take away electrons, or elevate the energy level of electrons in organic compounds to create new bonds. Examples of agents commonly used in this process include ozone, hydrogen peroxide, titanium dioxide, iron, and other oxidizing compounds. The oxidizing agents produce free radicals which also react with organic compounds such as PCBs, destroying them (Rahuman et al., 2000).

The most common UV oxidation processes involve the use of UV light paired with a catalyst (e.g., titanium dioxide) or strong oxidizer (e.g., ozone, hydrogen peroxide). In UV oxidation, PCBs are oxidized by various mechanisms, including direct reaction with oxidizers, UV photolysis, and the synergistic action of UV light (FRTR, 2002). If complete mineralization is achieved, the final products of oxidation are carbon dioxide, water, and salts. Oxidation of organic compounds using only a strong oxidizer does not always completely oxidize the contaminants, resulting in the presence of intermediate oxidation products. These intermediate products may be as toxic as or even more toxic than the original compound. Therefore, these oxidizing agents are paired with UV light to enhance the completion of the oxidation reactions, eliminating harmful byproducts (Munter, 2001).

UV oxidation using a catalyst generates electrons into the conduction band of the catalyst and produces valence band holes at the surface of the catalyst. Electron accepting species such as oxygen are reduced by the electrons in the conduction band, and water molecules or hydroxyl ions are oxidized at the holes. This process produces two hydroxyl radicals that rapidly break down and mineralize organic contaminants (Powell, 2000). A number of properties such as the surface state of the semiconductor, the flat-band potential, and the dissociation of the organic contaminant are strongly dependent on pH. The optimal pH value varies with the contaminant of concern and wastewater type (Munter, 2001). Therefore, pilot testing to determine optimal operating conditions would be necessary to confirm performance.

Advantages to UV oxidation include:

- Destroys, not removes, contaminants (there is no need to dispose of residuals)
- Also provides disinfection
- Can be used over a wide range of influent concentrations and flow rates (e.g., 1 to 1,000 gpm)
- Can be configured as batch or continuous process, depending on flow (FRTR, 2002; Kidman and Tsuji, 1991).

UV oxidation forms hydroxyl radicals that can potentially react with almost any organic chemical present in the wastewater. Therefore, to ensure that PCBs are completely destroyed there must be a sufficient amount of radicals, especially if other competing organic chemicals are present. However, overdosing with oxidizers such as ozone may actually decrease the destruction efficiency due to ozone absorption of the UV light. Therefore, careful planning and testing would be necessary to determine the optimal operating parameters.

Disadvantages associated with UV oxidation may include:

- Pretreatment to lower suspended solids concentrations and turbidity to provide for good transmission of UV light and minimize reactor and lamp maintenance
- Free radical scavengers can inhibit destruction efficiency (e.g., excessive dosages of oxidizers could become scavengers)
- Fouling of UV quartz sleeve as a result of material deposition from temperature and UV light exposure, or as a result of compounds in the source water reacting with the sleeve surface
- Handling and storage of oxidizers may require special safety precautions (FRTR, 2002)
- Use of ozone or hydrogen peroxide may result in the formation of harmful byproducts that would need to be removed prior to discharge.

Bench-scale experiments indicate that UV oxidation can destroy more than 95% of PCBs in aqueous solutions (Rahuman et al., 2000). However, laboratory testing does not always translate well to actual full-scale systems.

DOE (1994) indicates that PCBs from contaminated groundwater at the Department of Energy's Kansas City Plant were reduced from 300 ng/L to nondetect levels using UV oxidation (note, however, that the detection level was not specified). The groundwater was pumped and treated using a UV system coupled with hydrogen peroxide (the original UV with ozone and hydrogen peroxide system was modified after the initial four year testing period). The system was designed to treat about 30 gallons per minute (about 0.04 mgd).

There is some data on the efficiency of UV/titanium dioxide oxidation in reducing PCBs in wastewater and groundwater (Powell, 2000). Batch tests reduced PCB concentrations in wastewater from about 450,000,000 ng/L to 1,600 ng/L, after 2 hours of operation at a flow rate of 5.3 gallons per minute (0.008 mgd). This reduction corresponds to a destruction efficiency of greater than 99.99%. Continuous flow UV/titanium dioxide oxidation reduced PCB concentrations in groundwater from 2,470 ng/L to 70 ng/L at a flow rate of 2.9 gallons per minute (0.004 mgd) (Powell, 2000). The reduction in groundwater concentrations corresponds to a destruction efficiency of about 97% (Powell, 2000). In these two examples, the destruction efficiency decreased as influent concentrations decreased. Therefore, more testing would be needed to confirm that concentrations below the proposed criterion are attainable with UV/titanium dioxide oxidation.

Also, because most of the studies concerning UV oxidation involve contaminated groundwater, landfill leachate, or contaminated soils, and not municipal and industrial wastewater, additional testing may be necessary to ensure that the treatment technology is capable of producing effluent levels in these waters below the proposed criteria.

4.3 Sonochemical Degradation

Sonochemical degradation is another type of advanced oxidation process that can be used in the destruction of PCBs. The two main mechanisms for sonochemical degradation of pollutants are thermal reaction inside cavitating bubbles and radical chemistry (Destaillants et al., 2001).

Sonochemical degradation involves the use of ultrasound to create micron size gas cavities in water. These cavities collapse, generating a localized zone of high temperature (about 5,000EC) and pressure (greater than 100 atm). This process is called acoustic cavitation. Any organic molecules diffused in these gas bubbles before they collapse are rapidly hydrolyzed or oxidized to less toxic molecules such as carbon dioxide, water, and hydrochloric acid (Bruce, 2004). Cavitation is influenced by vapor pressure, surface tension, and hydrophobicity (Price, et al., 2004). As sound intensity increases, higher temperatures and pressures exist within the bubble interior, which increases the decomposition rate of PCBs (Hua and Zhang, 2000). However, the relatively low vapor pressure of PCBs precludes significant pyrolysis within acoustic cavitation bubbles (Destaillants et al., 2001).

Therefore, in addition to the direct high temperature and pressure destruction of organic molecules within these localized zones, the thermolysis of water to release radical species (e.g., CH, COH, and HOOC) and hydrogen peroxide is an important mechanism for the destruction of PCBs. These free radicals attack organic compounds present in water, comprising another mechanism of sonochemistry (Hua and Zhang, 2000). Free radical formation is also enhanced with increased sound intensities, thus, further increasing PCB decomposition rates (Hua and Zhang, 2000).

There are limited performance data for PCB destruction using sonochemical degradation. One laboratory test indicates that 99% destruction of PCBs occurs with sonication reaction times ranging from 29 minutes to 47 minutes, depending on the PCB congener (Hua and Zhang, 2000). Okuno et al. (2000) showed a 90% destruction efficiency for PCBs after only 30 minutes of sonication. However, since this was a lab test, there were no competing contaminants present in the water. Thus, performance on a much larger scale cannot be extrapolated from these results. Hua (2000) indicates that when other contaminants are present, longer reaction times are necessary to achieve the same destruction efficiencies observed in tests without competing contaminants.

Most studies on sonochemical degradation have been performed using frequencies of 20-25 kHz. However, reaction rates may vary significantly with variations in frequencies (Price et al., 2004). For example, frequencies dictate the type of cavitation that takes place. Under low frequencies transient cavitation occurs in which bubbles undergo only a few oscillations before collapse. A larger concentration of radicals will be formed on each individual bubble collapse during this type of cavitation because bubbles collapse from a larger radius and thus, contain more water vapor. At high frequencies, a bubble undergoes hundreds or thousands of oscillations, known as stable cavitation. Under stable cavitation more collapse events occur per unit time, which may increase the overall rate of radical production (Price et al., 2004). It is also important to note that at higher frequencies, cavitation is produced in the liquid phase far from the surface of the transducer, thus protecting it from erosion cause by bubble implosion (Destaillants et al., 2001). However, these situations have only been tested in laboratories and not on larger, full-scale reactors.

Destaillants et al. (2001) investigated how laboratory conditions translate into larger-scale operation. They found that react rates for the nonvolatile contaminants tested increased by factors of 2 to 5. Destaillants et al. (2001) also found that while the reaction rates for one non-volatile contaminant only varied by about 15% for various flow rates, reaction rates for another varied by more than 50% depending on flow rate. In addition, degradation rates may also vary with reactor size and configuration, temperature, and influent contaminant concentration (Destaillants et al., 2001). Therefore, it is important to conduct extensive testing to determine the optimal operating frequency for destruction of PCBs in a particular type of wastewater, as well as to determine if any harmful byproducts are formed during sonochemical oxidation.

4.4 Summary

There are no analytical methods that can measure PCBs at or below the proposed criterion of 0.072 ng/L (see Section 2.4). The currently approved method for water (Method 608) is for mixtures of PCB congeners, with a detection limit of 65 ng/L for PCB 1242. EPA Method 1668A (U.S. EPA, 1999) is not currently approved in 40 CFR 136, however, it may be used for "data gathering and monitoring associated with the Clean Water Act." The detection limit for this method ranges from 0.058 ng/L for monochlorobiphenyls to 0.450 ng/L for nonachlorobiphenyls. However, this method does not measure total PCBs. Therefore, until analytical methods capable of measuring total PCB concentrations below 0.072 ng/L become available, it will not be possible to demonstrate compliance with the proposed criterion.

Due to this deficiency in analytical methods, there are no field data that show that any technology is capable of achieving 0.072 ng/L of PCBs end-of-pipe. The performance data available does show high destruction efficiencies, for UV oxidation and sonochemical degradation, and high removal efficiencies for GAC. However, the majority of these tests were performed with high influent PCB concentrations and on a laboratory or bench-scale system. Therefore, although GAC, UV oxidation, and sonochemical degradation should be able to achieve the proposed criterion based on their removal/destruction efficiencies, additional testing on full-scale systems will not confirm the theory until there are analytical methods that can demonstrate compliance with the proposed criterion.

5. DDT Treatment Controls

DDT is highly insoluble in water, and its presence in wastewater is largely related to the presence of suspended solids to which it readily attaches. Thus, standard wastewater treatment technologies that reduce levels of suspended solids (e.g., settling, filtration) should also reduce levels of DDT. However, it is not likely that these controls would reduce DDT to levels below the proposed criteria.

Several technologies exist that can reduce environmental DDT levels, although the majority have only been tested primarily on contaminated soils and sediments with relatively high levels of DDT (for example, sites near former DDT production facilities). In those cases where DDT-contaminated soils or sediments have been treated, the primary method used has been removal and disposal in a hazardous waste disposal facility. However, because DDT is an organic pollutant and is similar to PCBs, the treatment technologies applicable for the removal of PCBs from wastewater should also, theoretically, be suitable for the reduction of DDT in wastewater.

Again, additional considerations, including the relative source contributions of DDT, and the potential costs of available treatment technologies, are discussed in the appendices.

5.1 Granular Activated Carbon

As described in Section 4.1, GAC adsorbs relatively small quantities of soluble organics onto its surface, thus, removing them from the finished product water. Thus, GAC has the potential to significantly reduce DDT in wastewater. There are no studies demonstrating DDT removal to the proposed criterion (0.004 ng/L), although a suitably designed system should attain effluent concentrations at this level (Sherbondy, 2004). However, because analytical techniques cannot measure down to the required level, it is not possible to demonstrate, with current analytical techniques, that the level could be attained.

There are several factors that effect the performance of GAC:

- Molecular weight and solubility of target contaminant - the molecular weight and solubility of DDT falls within the range of molecular weights and solubilities for PCBs
- The influent contaminant concentration - the higher the influent concentration, the quicker the carbon is consumed
- Presence of other organic compounds - other organics will compete for available adsorption sites (Lenntech, 2004)
- pH of wastewater - pH of the water must be controlled more so for DDT removal than PCB removal (Sherbondy, 2004).

Thus, the potential for GAC to achieve the proposed DDT criterion end-of-pipe would be very site-specific.

5.2 Advanced Oxidation Processes

As discussed in Sections 4.2 and 4.3, advanced oxidation processes can be used to reduce the concentration of organic contaminants such as DDT in wastewater. Advanced oxidation processes such as UV light coupled with ozone and/or hydrogen peroxide, UV light coupled with titanium dioxide, and sonochemical degradation destroy organic contaminants through the production of hydroxyl radicals which react with organics to break them down into non-toxic compounds such as carbon dioxide and water. Adjustment of pH levels after the oxidation process may also be necessary for oxidation processes designed to reduce DDT because the generation of protons and stoichiometric amounts of chloride ions generated during the photoreduction process may decrease effluent pH (Chu, 1999). However, for the reasons mentioned above for GAC, it is most likely not possible to demonstrate, with current analytical techniques, that the proposed DDT level could be attained with these processes.

5.3 Summary

There are no analytical methods that can measure DDT at or below the current or proposed criterion of 0.004 ng/L (see Section 2.4). The currently approved method for water (Method 608) has a detection limit of 12 ng/L for DDT. Therefore, until analytical methods capable of measuring DDT concentrations below 0.004 ng/L become available, it will not be possible to demonstrate compliance with the proposed criterion.

Due to this deficiency in analytical methods, there are no field data that show that any technology is capable of achieving the proposed criterion end-of-pipe. However, based on the literature, GAC and advanced oxidations processes should be capable of the removal/destruction efficiencies necessary for compliance with the proposed criterion. Note that the application of these processes for the removal of DDT from wastewater would be very site-specific, and may require pilot testing to ensure adequate performance and determine optimal operating conditions.

6. Alternative Reduction Strategies

Applying the technologies reviewed in the previous sections to treat municipal and industrial effluents end-of-pipe may involve targeting pollutants at concentrations that are substantially diluted from the total flow through the facility. However, it may be more cost-effective to reduce or eliminate the pollutants through source control or process modifications, than to install unproven end-of-pipe technologies, especially when methods are not available for verifying compliance for DDT and PCBs. This section discusses potential P2 strategies for mercury, DDT, and PCBs.

6.1 Mercury

There are a number of sources that may contribute to the mercury load in a wastewater stream, and reducing or eliminating the mercury from these sources before it reaches the headworks of a POTW or an industrial discharger's effluent could negate the need for additional and costly end-of-pipe treatment. Mercury in municipal wastewater could come from process chemicals (e.g., chlorine) or indirect dischargers such as dentists, hospitals, laboratories, schools, vehicle service centers, and residential areas (through human waste and household products). Process chemicals, raw materials, and source water may be responsible for the mercury found in industrial wastewater.

Treatment technologies used to control mercury discharges remove mercury from wastewater, but do not remove mercury from the environment. Instead, the mercury is simply transferred from the wastewater to another waste stream (e.g., sludge) that is either incinerated, land applied, or landfilled. Hence, mercury can have long-term adverse impacts on human health and the environment when only end-of-pipe treatment technologies are used. P2 focuses on reducing mercury at the source where it is more concentrated and more easily controlled. Source control measures focus on changing the existing processes that produce mercury wastes, capturing the mercury and recovering it for recycled use, and replacing the mercury used in existing processes with less toxic alternatives.

6.1.1 Industrial Dischargers

For industrial dischargers, P2 would consist of two steps: process analysis and process modifications. The process analysis study should identify mercury sources at the facility and within the process. Mercury may be found in a number of places. For example, cooling water treated with pH-altering chemicals, such as acid and caustic soda, may contain high levels of mercury. Also, some boilers use high pressure steam gauges that contain large amounts of mercury. One steam gauge collected during a mercury amnesty project contained 30 pounds of mercury (WLSSD, 1997). Some boilers may also contain mercury switches (U.S. EPA, 1998).

This analysis of current treatment processes may include additional monitoring at selected sites, data collection, data analysis, and report preparation, as well as an evaluation of process modifications. Process analysis can be implemented in two stages (Inland Ispat Indiana Harbor Works, et al., 2001):

- Initial, screening-level inventory - to get a general idea of where mercury may be and how much is present (e.g., what types and brands of equipment/chemicals contain mercury)
- Detailed inventory - to address every piece of equipment, waste stream, and storage area.

Once the sources of mercury have been identified, process modifications or best management practices (BMPs) should be employed to remove or reduce the mercury from each source. These BMPs can be divided into four categories:

- procedural changes
- material changes
- technology changes
- recycle and reuse.

Procedural Changes: Procedural changes involve changes to operating practices in a facility. The process analysis described above is necessary to identify the operational steps of the process that could be contributing to mercury loads. Procedural changes may include (MWRA/MASCO, 1997):

- Inventory control - consolidation and better management of mercury (e.g., making sure mercury is stored in proper containers and labeled)
- Material handling - training for employees on proper handling of hazardous materials
- Preventive maintenance - replacement of containers and equipment before failure
- Spill and leak prevention - employee training on spill response procedures and cleanup.

Material Changes: Material changes may include substitution, purification, and dilution. To accomplish material changes, a facility should first determine the chemical compositions of the materials used or produced within the process. This task may require contacting the material manufacturers since trace contaminants that account for less than 1% of the product are not required to be reported on material safety data sheets. Then, the facility should investigate the feasibility of replacement chemicals (MWRA/MASCO, 1997). Facilities should attempt to eliminate the use of mercury in their processes to the extent possible, and establish purchasing policies that eliminate or greatly reduce the amount of mercury used (Inland Ispat Indiana Harbor Works, et al., 2001).

Technology Changes: Technology changes involve changing equipment used in the production process or using another technology approach to achieve the same product or result. These options may include (MWRA/MASCO, 1997):

- Process changes (e.g., in surface cleaning, switching to mechanical cleaning methods from mercury-containing solvent cleaning methods)

- Equipment, piping, or layout changes (e.g., replacing mercury-containing equipment with mercury-free alternatives or cleaning out built-up mercury from piping)
- Additional automation (e.g., adjusting new chemical feed to achieve desired treatment efficiency)
- Energy conservation measures - reducing the amount of energy used and coal burned reduces mercury air emissions
- Water conservation measures - if mercury cannot be eliminated from the process, reducing the amount of water discharged could reduce the mercury load to the environment.

Recycle and Reuse: Recycling and reuse aims to reduce water usage and quantities of mercury-containing materials discharged. Recycle and reuse may include (MWRA/MASCO, 1997):

- Recycling or reusing portions of a waste stream within the original process
- Using a waste stream as a raw material in a different process or facility
- Treating a waste stream to reclaim portions for recycle or reuse within the original process or a different process or facility
- Separation technologies to reuse water or remove constituents for recycling as material inputs.

Whether or not such techniques can reduce effluent mercury levels to 0.5 ng/L has yet to be proven, and depends on the source of mercury and the feasibility of each P2 option at a particular facility.

6.1.2 Municipal Dischargers

Similar to industrial dischargers, whether or not P2 can achieve effluent mercury levels of 0.5 ng/L at municipal facilities has yet to be proven, and depends on the sources of mercury and the feasibility of the potential P2 measures for controlling the sources. However, there are several steps likely to be necessary for a successful municipal P2 program: source identification, establishing a workgroup, defining goals, developing a P2 approach, assessing progress, and providing follow-up. These steps are applicable to both voluntary and regulatory programs. Voluntary P2 programs involve efforts to reduce mercury that are done by choice, and regulatory P2 programs involve efforts required by law, ordinance, or code.

Source identification is one of the most important steps in developing a P2 program. Potential sources of mercury in POTW influents include:

- Dentists
- Industrial dischargers
- Hospitals and medical centers
- Laboratories
- Schools and universities
- Vehicle service centers

- Potteries.

Once a discharger has identified possible mercury sources and quantified the loads from each source, implementation procedures should be developed. The municipalities should form a workgroup to organize and develop the voluntary program, identify the tools that can be used for each potential source, determine the appropriate target audiences, develop a budget detailing the costs of the program, and assist in assessing progress.

A P2 program workgroup should start by developing program goals. Program goals should reflect the intentions of the program and include a time line for completion. When defining program goals, possible barriers or obstacles that could prevent or delay the program from achieving its objectives should be kept in mind. Factors such as the number of qualified staff, P2 knowledge and expertise, and community and industry support may have a significant impact on the reductions achievable through P2. Identifying potential problems enables the facility management to take actions to avoid them, or to come up with a plan for handling them when they arise. This can save time and keep the program running smoothly.

The workgroup should then work with the facility management to prioritize the sources of mercury identified based on the contribution to total mercury loadings, relative magnitude of loading, and feasibility of control. Priorities for outreach depend on several factors, not just quantity of loading. After the sources have been prioritized, the workgroup should compile a list of applicable P2 strategies that could be used to control each source. It is important to estimate each strategy’s effectiveness and maximum potential reduction capability. Activities that target specific sources are usually most effective, and may be included in the following categories:

- Public outreach/education campaigns (i.e., brochures targeting sources on mercury use and elimination of such uses)
- Legislation and regulatory improvements (i.e., product bans)
- Collection programs (i.e., mercury-containing equipment exchanges).

Measuring both successes and failures is important for evaluating the effectiveness of a P2 program. Assessment should be done constantly during the life of the project so that problems can be avoided and successes can be repeated. Measuring progress along the way also helps to keep those involved motivated. Follow-up measures are important to make sure that the P2 program is continued even after the goals have been achieved. The workgroup should follow up with targeted sources to ensure that P2 will continue as a part of their everyday operations. The workgroup follow-up activities should also identify any gaps in knowledge. Through the monitoring of sources, a municipality can determine what percent of the mercury influent load is accounted for and what percent remains unidentified. From this analysis, a list of needed information or areas that may require more research can be compiled so that more work can be done to identify and eventually control any additional mercury sources.

6.2 PCBs

PCBs enter surface water primarily through surface runoff of PCB-contaminated soil. Available evidence indicates that PCB concentrations in surface waters are higher at higher receiving water flows and at higher suspended solids concentrations (McKee et al. 2004; DRBC 1998). This is consistent with the fact that PCBs are readily adsorbed to particles and are relatively insoluble in water. The largest probable source of PCBs to soils is leakage from products in which PCBs were historically incorporated (PCB usage, except for very minor amounts, was banned in 1977). Electrical transformers comprised the major use of PCBs because of their property as a dielectric (insulating) fluid. Other sources, however, included fluorescent tube ballasts and other electrical equipment. Transformer fires, in which PCB containment was compromised, likely contributed the greatest amount of PCBs to soils, although other types of leakage have also occurred.

Because of its association with contaminated soil, PCBs are more likely to be contained in storm water from contaminated sites than other sources, but they enter wastewater treatment plant collection systems where storm water and domestic and industrial wastewaters are combined. Again, available evidence indicates that PCBs are detected most frequently during storm events, especially at combined sewer overflows. PCBs that enter treatment plants generally readily partition to sludge, and removal efficiencies of typical secondary treatment plants are greater than 95% (Morris and Lester, 1994). Storm events, however, do not appreciably affect treatment plant effluent loadings (DRBC, 1998).

P2 for PCBs has not typically been employed by municipal wastewater treatment plants. Few of these facilities have PCB limits. Thus, there has been little incentive to undertake P2 activities related to PCBs. Such activities, however, like identifying contaminated soils that would run off to a collection system and either isolating those soils or requiring the owner to clean up the soils, could be effective in reducing PCBs in treatment plant influents. The identification of potential inputs from PCB spills at facilities that discharge to municipal wastewater treatment plants would generally be a lower priority, as these potential sources could likely discharge much lower concentrations of PCBs, and the discharge would not likely be associated with wet weather (although there could be exceptions). DRBC (1998) discovered an illegal discharge to a Philadelphia treatment plant collection system, so the tracing of pollutant sources could be beneficial for facilities with high influent loadings during dry weather.

6.3 DDT

Even if the treatment technologies discussed above could be demonstrated to remove DDT to criterion levels, it would be necessary to determine when and where such technologies should be required. Although DDT is no longer used or produced in the United States, other parts of the world continue to use it in agricultural practices and disease-control programs. Therefore, atmospheric deposition is the main source of new DDT contamination (U.S. EPA, 2004b).

Because atmospheric deposition is the primary source of DDT contamination, wet weather events may result in higher DDT concentrations at wastewater treatment plants than dry weather conditions. Therefore, it may be inefficient to treat an entire effluent all the time to control release of DDT during wet weather. This issue becomes problematic when trying to design a treatment system that can bring appropriate technologies on-line only when they are necessary to meet water quality standards, and would be especially problematic for POTWs with combined sewers. Controls on storm water outfalls or overflows may prove more successful at reducing DDT loads to surface waters than end-of-pipe wastewater treatment. However, there are no data to indicate that such controls would achieve the proposed criterion.

6.4 Summary

Due to the lack of performance data demonstrating the technologies' ability to achieve the proposed criteria end-of-pipe, and limitations in analytical techniques, a more viable and cost-effective strategy may be for facilities to implement P2 programs. P2 could produce gains toward achieving standards without imposing the costs of unproven end-of-pipe technologies large scale. For municipal wastewater treatment plants in particular, mercury, PCBs, and DDT are not usually generated or introduced during treatment processes (the addition of chlorine contaminated with mercury may be one exception). Generally, mercury enters the plant from indirect dischargers such as dentists, hospitals, laboratories, and schools as well as from human waste and consumer products. For PCBs and DDT, storm water and other sources such as human and food waste are the main sources of contamination. Whereas end-of-pipe treatment removes contaminants from discharges, it does nothing to eliminate the actual source of contamination (e.g., air deposition, commercial dischargers). In comparison, pollution prevention efforts focus on eliminating the remaining sources of the pollutants (e.g., current uses of mercury, sites contaminated from past PCB and DDT manufacturer and use), instead of applying treatment to larger waste streams in which the contaminants are more diluted.

7. Summary

There are no field data that demonstrate that any of the technologies discussed above are capable of achieving the proposed criteria for mercury, PCBs, and DDT end-of-pipe on a consistent basis. In addition, the limited performance data that are available are for bench- or pilot-scale systems, not full-scale wastewater treatment plants. Most of these applications will vary among facilities, and their application will be very site-specific. For PCBs and DDT, there are no analytical methods with detection limits low enough to demonstrate compliance with the proposed criteria. Therefore, while in theory the treatment technologies should be able to achieve the proposed criteria based on their removal/destruction efficiencies, additional testing on full-scale systems for mercury, PCBs, and DDT and the development of more sensitive analytical methods for PCBs and DDT are needed to justify the installation of end-of-pipe treatment controls.

Exhibit 7-1 summarizes the available treatment technologies for the removal of mercury, PCBs, and DDT from wastewater. It is important to note that there are no data that show that any of these technologies can consistently achieve the proposed criteria. This may be due to the fact that analytical techniques for measuring pollutant to such low levels were only recently approved or have not yet been developed, and few, if any, dischargers are required to meet such low levels end-of-pipe.

Exhibit 7-1. Summary of Available Technologies for Treating Mercury, PCBs, and DDT in Wastewater

Treatment Process	Feasibility Summary
Mercury	
Reverse Osmosis	Research (NDWC, 1997) indicates that when RO units are operated in series, almost 100% removal of mercury is attainable. However, data are not available to confirm that mercury levels <0.5 ng/L are being meet. Also, disposal of residuals may be extremely costly.
Selective Sorbents	Several sorbents (e.g., Keyle:X, SIR-200) have been proven to produce low mercury levels (e.g., less than 19 ng/L), however, additional testing is required to confirm that this technology is capable of achieving the proposed criterion, especially on a large scale.
Mercury Precipitation	Mercurous sulfides are extremely insoluble in water. Therefore, precipitating mercury with sulfides and then removing the mercurous sulfide compounds with activated alumina should, in theory, achieve the proposed criterion. However, actual performance data is lacking. The process may also form toxic odors and create harmful by-products.
PCBs	
Granular Activated Carbon	Data indicate that GAC, when operated properly, should be able to achieve PCB levels near the proposed criterion. However, sensitive analytical methods are not available to confirm that the proposed criterion can be met. Also, because GAC is an adsorptive process, residuals disposal may be costly.

Exhibit 7-1. Summary of Available Technologies for Treating Mercury, PCBs, and DDT in Wastewater

Treatment Process	Feasibility Summary
UV Oxidation	PCBs can be almost completely destroyed in wastewater through oxidation (99.99% destruction efficiency). UV oxidation using strong oxidants or a catalyst is an effective process for destroying PCBs. Testing is necessary to determine which oxidation process best destroys PCBs for a particular type of wastewater. However, sensitive analytical methods are not available to confirm that the proposed criterion can be met.
Sonochemical Degradation	Employs same mechanisms as UV oxidation (i.e., radical chemistry) to destroy PCBs, however, ultrasound is much more energy intensive. Also, the technology has not been proven effective for PCBs in wastewater outside of the laboratory.
DDT	
Granular Activated Carbon	Because DDT is similar in behavior and structure to PCBs, GAC, when operated properly, should be able to achieve nondetect levels. However, sensitive analytical methods are not available to confirm that the proposed criterion can be met. Residuals disposal and pH adjustment may also increase treatment costs.
Advanced Oxidation Processes	AOPs destroy DDT so there are no residuals. AOPs can theoretically destroy all DDTs in wastewater, however, sensitive analytical methods are not available to confirm that the proposed criterion can be met.

NA = not available.
mgd = million gallons per day

For municipal dischargers, PCBs and DDT are not generated during treatment processes, but rather pass through the plant from contaminated storm water and other sources such as human and food waste. Therefore, although end-of-pipe treatment may remove the contaminants from the discharge, it does nothing to eliminate the actual source of contamination (e.g., air deposition). In comparison, P2 efforts focus on eliminating the remaining sources of the pollutants (e.g., current uses of mercury, sites contaminated from past DDT use), instead of applying treatment to larger waste streams in which the contaminants are more diluted. Considering the costs of the unproven end-of-pipe technologies, such efforts may result in substantial progress toward achieving standards at a much lower cost.

8. References

Agency for Toxic Substances and Disease Registry (ATSDR). 1999. Toxicological Profile for Mercury. U.S. Department of Human and Health Services.

Bruce, David A. 2004. Personal communication (email) in July with L. Petruzzi, SAIC.

Chemviron Carbon. Activated Carbon Applications: Groundwater Remediation. Online at http://www.chemvironcarbon.com/applications/water/grdwater.htm.

Chu, Wei. 1999. Photodechlorination Mechanism of DDT in UV/Surfactant System,. Environ. Sci. Technol., Vol. 33, No. 3, pp421-425.

Delaware River Basin Commission (DRBC). 1998. Study of the Loadings of Polychlorinated Biphenyls from Tributaries and Point Sources Discharging to the Tidal Delaware River. June 1998.

Department of Energy (DOE). 1994. Pump and Treat of Contaminated Groundwater at U.S. Department of Energy Kansas City Plant, Kansas City, Missouri. Online at https://denix.osd.mil/denix/Public/Library/Remedy/Kansas/ksplnt01.html.

Destaillants, H., T.M. Lesko, M. Knowlton, H. Wallace, M.R. Hoffmann. 2001. Scale-Up of Sonochemical Reactors for Water Treatment. Ind. Eng. Chem. Res.; 40(18), 3855-3860.

Federal Remediation Technologies Roundtable (FRTR). 2002. Remediation Technologies Screening Matrix and Reference Guide, Version 4.0. Online at http://www.frtr.gov/matrix2/top_page.html.

Hua, Inez. University of Purdue. 2004. Personal communication in July with L. Petruzzi, SAIC.

Hua, Inez. An Investigation of Homogeneous and Heterogeneous Sonochemistry for Destruction of Hazardous Waste. Prepared for U.S. Department of Energy, Environmental Management.

Hua, I. and G. Zhang. 2000. Cavitation Chemistry of Polychlorinated Biphenyls: Decomposition mechanisms and Rates. Environmental Science and Technology. Vol. 34, No. 8 (1529 - 1534).

Inland Ispat Indiana Harbor Works, Bethlehem Steel Burns Harbor Division, United States Steel Gary Works, The Delta Institute, Lake Michigan Forum. 2001. A Guide to Mercury Reduction in Industrial and Commercial Settings.

Kidman, R.B. and K.S. Tsuji. Preliminary Cost Comparison of Advanced Oxidation Processes. Los Alamos National Laboratory. SJB 1-21-91.

Klasson, K.T., D.T. Bostick, L.L. Farr, D.R. McTaggart, P.A. Taylor. 2000. Demonstration of Mercury Sorbents to Meet DOE Customer Needs. ORNL/TM-2000/12.

Klasson, K.T., D.T. Bostick, P.A. Taylor, L.L. Farr. 1999. Removal of Trace-Level Mercury from Surface Water. For Presentation at the Air and Waste Management Association's 92nd Annual Meeting and Exhibition, June 20-24, 1999, St. Louis, MO.

Lenntech. 2004. Adsorption/Active Carbon. Online at http://www.lenntech.com/adsoprtion.htm.

Medical Academic and Scientific Community Organization (MASCO). 19997. Phase II Reports: Pretreatment Guidance Manual. Online at http://www.masco.org/mercury/pretreatment/index.html.

McKee, L, J. Leatherbarrow, R. Eads, and L. Freeman. 2004 Draft. Concentrations and Loads of PCBs, OC Pesticides, and Mercury Associated with Suspended Particles in the Lower Guadalupe River, San Jose, California. SFEI Contribution #86, January.

Morris, S., and J.N. Lester. 1994. Behaviour and fate of polychlorinated biphenyls in a pilot waste water treatment plant. Water Research 28(7): 1553-1561.

Munter, Rein. 2001. Advanced Oxidation Processes - Current Status and Prospects. Proc. Estonian Acad. Sci. Chem. 50(2), 59-80.

MWRA/MASCO Mercury Workgroup. 1997. End-of-Pipe Subcommittee Mercury Workgroup Pretreatment Guidance Manual. Online at www.masco.org/mercury/pretreatment/index.html.

National Drinking Water Clearinghouse (NDWC). 1997. Tech Brief: Ion Exchange and

Okuno, H., B. Yim, Y, Mizukoshi, Y. Nagata, Y. Maeda. 2000. Sonolytic Degradation of Hazardous Organic Compounds in Aqueous Solution. Ultrasonics Sonochemistry Vol 7, pp 261-264.

Paquin, J., S. Panasuk, and , B., Cote. 1997. Removal of Contaminants from Wastewater Using an Absorption System Acting on Surface Tension. 6th Symposium on Groundwater and Soil Remediation. Montreal, Canada.

Powell, Tony. 2000. Treatment of PCBs in Water. Purifics ES Inc.

Price, G.J., M. Ashokkumar, F. Grieser. 2004. Sonoluminescence Quenching of Organic Compounds in Aqueous Solution: Frequency Effects and Implications for Sonochemistry. Journal of American Chemical Society, 126, 2755-2762.

Rahuman, M., L. Pistone, F. Trifirò, S. Miertus. 2000. Destruction Technologies for Polychlorinated Biphenyls (PCBs). UNIDO Publications: Proceedings of Expert Group Meetings on POPs and Pesticides Contamination.

Sherbondy, John. 2004. Tigg Corporation. Personal communication in July with L. Petruzzi, SAIC.

TN & Associates. 1998. Mercury Treatability Study Final Report, Oak Ridge Y-12 Plant, Oak Ridge, Tennessee. BJC/OR-46.

U.S. EPA. 2004a. Permit Compliance System Database. Accessed July 1, 2004. Online at http://www.epa.gov/enviro/html/pcs/adhoc.html.

U.S. EPA. 2004b. Persistent, Bioaccumulative, and Toxic (PBT) Chemical Program: DDT. Online at http://www.epa.gov/pbt/ddt.htm.

U.S. EPA. 2002. Arsenic Rule Training Sessions: Waste Disposal Issues. Online at http://www.epa.gov/safewater/ars/presentations/train6-residuals.pdf.

U.S. EPA. 2000a. Arsenic Removal from Drinking Water by Ion Exchange and Activated Alumina Plants. EPA 600-R-00-088.

U.S. EPA. 2000b. Wastewater Technology Fact Sheet: Granular Activated Carbon Absorption and Regeneration. EPA 832-F-00-017.

U.S. EPA 1999. Method 1668, Revision A: Chlorinated Biphenyl Congeners in Water, Soil, Sediment, and Tissue by HRGC/HRMS. EPA-821-R-00-002.

U.S. EPA. 1998. Appendix A - Project Summaries FY 1992-1995.

U.S. EPA. 1997. Capsule Report: Aqueous Mercury Treatment. EPA 625-R-97-004. Office of Research and Development, Washington D.C.

U.S. EPA. 1996. Capsule Report: Reverse Osmosis. EPA 625-R-96-009. Office of Research and Development, Washington D.C.

U.S. EPA. 1991. Technical Support Document for Water Quality-Based Toxics Control. EPA 505-2-90-001.

U.S. EPA. 1980. Summary Report: Control and Treatment Technology for the Metal Finishing Industry, Sulfide Precipitation. EPA 625/8-80-003.

U.S. FWS, U.S. EPA, NJDEP. 2001. Derivation of New Jersey-Specific Wildlife Values as Surface Water Quality Criteria for: PCBs, DDT, Mercury.

West General. 2004. Out of the Box Chemical Specifications: Mercury. Online at http://www.westgeneral.com/outofthebox/masterframeset.html.

Western Lake Superior Sanitary District (WLSSD). 1997. Blueprint for Mercury Elimination.

Appendix A. Source Contribution Data

There are no comprehensive data on the relative contribution of different sources to the loadings of mercury, PCBs, and DDT to New Jersey surface waters. Therefore, the overall environmental loading and the relative contributions to the total loading from sources such as wastewater treatment plants cannot be estimated. However, the following sections describe the few data that are available, and discuss the potential implications of the information.

A.1 Mercury

The National Atmospheric Deposition Program (NADP), a cooperative research support program of the State Agricultural Experiment Stations, Federal and State Agencies, and non-governmental research organizations, provides a map of total mercury wet deposition in 2003 for the eastern United States (NADP, 2003). The State of New Jersey falls in an area of the country that receives 10-12 $\mu g/m^2$ of mercury per year. Based on an area of water of 3.4×10^9 m^2 (1,303 square miles) in the State (NSTATE LLC, 2004) the loading rate implies wet deposition of between 34-41 kg per year.

U.S. EPA (1996) published waste load allocations (WLAs) for the New York/New Jersey Harbor total maximum daily load (TMDL) for mercury, based on a projected 10% reduction in combined sewer overflow (CSO) and boundary loadings, a 30% reduction in storm water loadings, and a 60% reduction in atmospheric deposition (**Exhibit A-1**). Adjusting the WLAs to obtain loadings before projected reductions (e.g., dividing the CSO WLAs by 0.9) results in the values and relative percent contributions shown in **Exhibit A-2**.

Exhibit A-1. New York/New Jersey Harbor TMDLs

Area	Loading (pounds/day total recoverable mercury)					
	Point Sources	CSOs[1]	Stormwater[2]	Boundary	Atmosphere[3]	TMDL
Hudson River	0.185	0.057	0.481	0.138	0.245	1.106
Inner Harbor	0.183	0.034	0.007	0	0.054	0.278
Outer Harbor	0.000	0.026	0.010	0	1.139	1.175
Kills	0.328	0.066	0.516	0	0.225	1.135
East & Harlem R	1.005	0.216	1.260	0	0.679	3.160
Jamaica Bay	0.274	0.106	0.119	0	0.093	0.592
Raritan Bay	0.442	0.005	0.628	0.003	0.328	1.406
Hack/Pas/ Newark B	0.215	0.060	0.784	0.002	0.036	1.097

Source: U.S. EPA (1996a).
1. Loads include a projected 10% reduction.
2. Loads include a projected 30% reduction.
3. Loads include a projected 60% reduction.

Exhibit A-2. Estimated Loadings and Source Contributions from New York/New Jersey Harbor TMDLs[1]

Area	Loading (pounds/day total recoverable mercury)					
	Point Sources	CSOs	Storm Water	Boundary	Atmosphere	Total Estimated Loadings
Hudson River	0.185	0.063	0.687	0.153	0.613	1.701
Inner Harbor	0.183	0.038	0.010	0.000	0.135	0.366
Outer Harbor	0.000	0.029	0.014	0.000	2.848	2.891
Kills	0.328	0.073	0.737	0.000	0.563	1.701
East & Harlem R	1.005	0.240	1.800	0.000	1.698	4.743
Jamaica Bay	0.274	0.118	0.170	0.000	0.233	0.794
Raritan Bay	0.442	0.006	0.897	0.003	0.820	2.168
Hack/Pas/ Newark B	0.215	0.067	1.120	0.002	0.090	1.494
Sum	2.632	0.633	5.436	0.159	6.998	15.857
Percent of Total Loadings						
Hudson River	10.9%	3.7%	40.4%	9.0%	36.0%	100.0%
Inner Harbor	50.0%	10.3%	2.7%	0.0%	36.9%	100.0%
Outer Harbor	0.0%	1.0%	0.5%	0.0%	98.5%	100.0%
Kills	19.3%	4.3%	43.3%	0.0%	33.1%	100.0%
East & Harlem R	21.2%	5.1%	38.0%	0.0%	35.8%	100.0%
Jamaica Bay	34.5%	14.8%	21.4%	0.0%	29.3%	100.0%
Raritan Bay	20.4%	0.3%	41.4%	0.2%	37.8%	100.0%
Hack/Pas/ Newark B	14.4%	4.5%	75.0%	0.1%	6.0%	100.0%

1. Based on adjusting waste load allocations in U.S. EPA (1996a) for the projected reductions of 10% for CSOs, 30% for storm water, and 60% for atmospheric deposition.

Note, however, that the estimated loadings from atmospheric deposition for the New York-New Jersey Harbor exceed what would be implied for all of New Jersey, based on the NADP-estimated wet deposition rate. New Jersey has an area of 2.3×10^{10} m^2 (8,722 square miles) (NSTATE LLC, 2004). Multiplying this area by 10-12 $\mu g/m^2$ of mercury per year results in an estimate of 230 to 276 kg/year. In comparison, the total estimated atmospheric deposition load shown in Exhibit A-2 is about 6.998 lbs/day total recoverable, or 1,160 kg/year. If the TMDL estimates represent deposition to the total land and water area in the watershed, some portion of the storm water loadings may be double counted. If atmospheric or storm water loadings are overstated by the estimates, the percent of loadings attributable to other sources (e.g., point sources) may be understated.

EPA's PCS database contains mercury concentration data and facility flows for permitted point sources (**Exhibit A-3**).

Exhibit A-3. Mercury Concentration Data for Facilities in New Jersey with Data in PCS[1]

NPDES Number	SIC Code[2]	Facility Type	Number of Samples	Number of Detects	Number of Nondetects	Average of Detects (µg/L)	Nondetect Detection Limits (µg/L)
NJ0025178	4952	Major	2	1	1	22	< 0.1
NJ0024015	4952	Major	66	5	61	1.7	< 0.1 - < 6.7
NJ0034339	4952	Major	21	2	19	1.2	< 0.2
NJ0021636	4952	Major	19	5	14	0.84	< 0.03 - < 0.2
NJ0021016	4952	Major	77	10	67	0.72	< 0.2 - < 0.5
NJ0024953	4952	Major	22	1	21	0.60	< 0.2 - < 0.5
NJ0023787	4952	Major	5	2	3	0.55	< 0.5
NJ0020141	4952	Major	106	6	100	0.46	< 0.0002 - < 1
NJ0029084	4952	Major	9	1	8	0.30	< 0.2
NJ0024937	4952	Major	10	10	0	0.20	NA
NJ0023701	4952	Major	6	1	5	0.20	< 0.1
NJ0024660	4952	Major	2	1	1	0.20	< 0.1
NJ0024511	4952	Major	45	4	41	0.10	< 0.1 - < 1.31
NJ0020028	4952	Major	19	19	0	0.043	NA
NJ0024902	4952	Major	76	13	63	0.042	< 0.02 - < 0.16
NJ0024791	4952	Major	11	2	9	0.00025	< 0.0002
NJ0021113	4952	Major	3	0	3	NA	< 1
NJ0024686	4952	Major	2	0	2	NA	< 1
NJ0025160	4952	Major	10	0	10	NA	< 0.5
NJ0025321	4952	Major	9	0	9	NA	< 0.2 - < 2
NJ0026085	4952	Major	9	0	9	NA	< 0.2 - < 2
NJ0028142	4952	Major	2	0	2	NA	< 0.2 - < 1
NJ0020427	4952	Major	1	0	1	NA	< 0.2
NJ0020591	4952	Major	28	0	28	NA	< 0.2
NJ0024694	4952	Major	1	0	1	NA	< 0.2
NJ0024741	4952	Major	13	0	13	NA	< 0.2
NJ0024864	4952	Major	14	0	14	NA	< 0.2
NJ0025038	4952	Major	6	0	6	NA	< 0.2
NJ0027961	4952	Major	2	0	2	NA	< 0.2
NJ0031119	4952	Major	1	0	1	NA	< 0.2
NJ0020915	4952	Major	1	0	1	NA	< 0.1
NJ0021601	4952	Major	3	0	3	NA	< 0.1
NJ0022519	4952	Major	5	0	5	NA	< 0.1

Exhibit A-3. Mercury Concentration Data for Facilities in New Jersey with Data in PCS[1]

NPDES Number	SIC Code[2]	Facility Type	Number of Samples	Number of Detects	Number of Nondetects	Average of Detects (µg/L)	Nondetect Detection Limits (µg/L)
NJ0024040	4952	Major	1	0	1	NA	< 0.1
NJ0024856	4952	Major	2	0	2	NA	< 0.1
NJ0027545	4952	Major	1	0	1	NA	< 0.1
NJ0024643	4952	Major	8	0	8	NA	< 0.028 - < 0.17
NJ0020184	4952	Major	3	0	3	NA	< 0.0002
NJ0027481	4952	Major	3	0	3	NA	< 0.0001 - < 0.1
NJ0052256	4952	Minor	1	1	0	0.3 0	NA
NJ0050369	4952	Minor	3	1	2	0.20	< 0.2
NJ0020290	4952	Minor	8	2	6	0.20	< 0.0002 - < 2
NJ0098922	4952	Minor	2	1	1	0.20	< 0.0001
NJ0026867	4952	Minor	3	2	1	0.15	< 0.1
NJ0029831	4952	Minor	6	2	4	0.15	< 0.1
NJ0027561	4952	Minor	10	1	9	0.13	< 0.1
NJ0026891	4952	Minor	3	1	2	0.10	< 0.5 - < 1
NJ0080811	4952	Minor	3	2	1	0.10	< 0.1
NJ0020605	4952	Minor	3	0	3	NA	< 1
NJ0026387	4952	Minor	1	0	1	NA	< 1
NJ0026816	4952	Minor	1	0	1	NA	< 1
NJ0027715	4952	Minor	2	0	2	NA	< 1
NJ0027669	4952	Minor	11	0	11	NA	< 0.5 - < 1
NJ0026905	4952	Minor	1	0	1	NA	< 0.5
NJ0031771	4952	Minor	2	0	2	NA	< 0.5
NJ0050130	4952	Minor	1	0	1	NA	< 0.5
NJ0024465	4952	Minor	1	0	1	NA	< 0.2
NJ0021326	4952	Minor	3	0	3	NA	< 0.1
NJ0021890	4952	Minor	1	0	1	NA	< 0.1
NJ0022985	4952	Minor	2	0	2	NA	< 0.1
NJ0024414	4952	Minor	3	0	3	NA	< 0.001 - < 0.5
NJ0027049	4952	Minor	5	0	5	NA	< 0.0005 - < 0.5
NJ0027006	4952	Minor	3	0	3	NA	< 0.0005
NJ0027057	4952	Minor	3	0	3	NA	< 0.0002 - < 0.2
NJ0023493	4952	Minor	3	0	3	NA	< 0.0001
NJ0109061	4952	Minor	2	0	2	NA	< 0.0001

Exhibit A-3. Mercury Concentration Data for Facilities in New Jersey with Data in PCS[1]

NPDES Number	SIC Code[2]	Facility Type	Number of Samples	Number of Detects	Number of Nondetects	Average of Detects (µg/L)	Nondetect Detection Limits (µg/L)
NJ0005444	4911	Major	69	67	2	10	< 0.2 - < 1.87
NJ0005240	4953	Major	28	7	21	0.56	< 0.2
NJ0000019	2841	Major	18	9	9	0.29	< 0.2
NJ0022586	8063	Major	1	1	0	0.20	NA
NJ0002089	2951	Major	6	2	4	0.20	< 0.2 - < 20
NJ0005100	2869	Major	62	16	46	0.035	< 0.02 - < 1.2
NJ0002348	2833	Major	2	0	2	NA	< 0.2
NJ0004286	2821	Major	1	0	1	NA	< 0.2
NJ0027201	1522	Minor	5	1	4	70	< 0.0005 - < 0.5
NJ0027073	8211	Minor	3	2	1	9.6	< 0.5
NJ0032522	2869	Minor	30	11	19	6.1	< 0.001 - < 1
NJ0023311	8211	Minor	3	1	2	3.0	< 1
NJ0032522	2869	Minor	35	14	21	1.9	< 0.001 - < 1
NJ0021105	8211	Minor	3	2	1	1.0	< 0.5
NJ0023841	8211	Minor	3	2	1	0.95	< 0.5
NJ0031046	8211	Minor	5	2	3	0.50	< 0.5
NJ0021091	8211	Minor	3	2	1	0.45	< 0.5
NJ0132004	2851	Minor	4	1	3	0.40	< 0.1
NJ0022284	8211	Minor	3	1	2	0.30	< 0.5 - < 5
NJ0024104	6513	Minor	2	1	1	0.20	< 0.1
NJ0026824	6512	Minor	2	2	0	0.10	NA
NJ0027731	8731	Minor	15	1	14	0.055	< 0.00004 - < 0.16
NJ0027731	8731	Minor	15	2	13	0.054	< 0.00004 - < 0.16
NJ0026727	5812	Minor	1	1	0	0.050	NA
NJ0027731	8731	Minor	15	1	14	0.039	< 0.000026 - < 0.16
NJ0026514	6512	Minor	4	0	4	NA	< 0.5
NJ0032395	6512	Minor	2	0	2	NA	< 0.5
NJ0035718	6512	Minor	3	0	3	NA	< 0.5
NJ0053112	4953	Minor	2	0	2	NA	< 0.5
NJ0108367	4311	Minor	2	0	2	NA	< 0.5
NJ0028894	8211	Minor	3	0	3	NA	< 0.2 - < 0.5
NJ0000752	5171	Minor	1	0	1	NA	< 0.2
NJ0050563	3081	Minor	4	0	4	NA	< 0.2

Exhibit A-3. Mercury Concentration Data for Facilities in New Jersey with Data in PCS[1]

NPDES Number	SIC Code[2]	Facility Type	Number of Samples	Number of Detects	Number of Nondetects	Average of Detects (µg/L)	Nondetect Detection Limits (µg/L)
NJ0134929		Minor	40	0	40	NA	< 0.1 - < 5
NJ0023124	8211	Minor	2	0	2	NA	< 0.1 - < 1
NJ0050580	6513	Minor	2	0	2	NA	< 0.1
NJ0132004	2851	Minor	3	0	3	NA	< 0.1
NJ0028487	9223	Minor	9	0	9	NA	< 0.03471 - < 0.2
NJ0027731	8731	Minor	4	0	4	NA	< 0.026 - < 0.12
NJ0022306	8211	Minor	3	0	3	NA	< 0.01 - < 0.5
NJ0021083	9711	Minor	3	0	3	NA	< 0.001 - < 1
NJ0027529	8059	Minor	3	0	3	NA	< 0.001 - < 1
NJ0021865	7997	Minor	1	0	1	NA	< 0.001
NJ0022144	8211	Minor	2	0	2	NA	< 0.001
NJ0029858	8059	Minor	1	0	1	NA	< 0.0005
NJ0035084	8731	Minor	4	0	4	NA	< 0.0001 - < 0.2
NJ0098663		Minor	1	0	1	NA	< 0.0001

Source: EPA's PCS database.

1. Facilities that have multiple outfalls are listed more than once.

2. SIC code of 4952 are municipal wastewater treatment plants. All other SIC codes are considered industrial facilities (including blanks).

Exhibit A-4 summarizes the PCS data by facility type and category. Much of the data are nondetects, and the range for detection limits is wide (e.g., by orders of magnitude for the same facility). Also, the detection limits for many facilities are very high (e.g., 0.2, 0.5, and 1 µg/L), which does not allow estimation of concentrations in the ng/L range. However, some New Jersey facilities have relatively high detected values (0.3 ng/L to 70,000 ng/L) compared to levels achieved by similar facilities (e.g., a number of municipal and industrial facilities in California that have sampled with sensitive methods have average mercury concentrations ranging from 1.6 ng/L to a maximum of 600 ng/L, including a number with average concentrations below 10 ng/L.)

Exhibit A-4. Summary of PCS Mercury and Flow Data for New Jersey Facilities

Facility Type	Number of Facilities with Mercury Data	Average of Detected Values[1] (µg/L)	Range of Nondetects[2] (µg/L)	Number of Facilities with Flow Data	Average Flow[3] (L/day)
Major Municipals	39	0.15	<0.0001 - <6.7	94	46,000,000
Minor Municipals	9	0.19	<0.0001 - <1	64	860,000
Major Industrials	8	0.13[4]	<0.02 - <20	25	9,300,000

Exhibit A-4. Summary of PCS Mercury and Flow Data for New Jersey Facilities

Facility Type	Number of Facilities with Mercury Data	Average of Detected Values[1] (µg/L)	Range of Nondetects[2] (µg/L)	Number of Facilities with Flow Data	Average Flow[3] (L/day)
Minor Industrials	39	0.17[5]	<0.000026 - <5	291	1,300,000
Total	95	NA	NA	474	NA

NA = not applicable.

1. Based on EPA PCS data from 2001 to 2004, assuming nondetects equal to one half of the detection limit and facilities for which standard industrial classification (SIC) codes are missing in PCS are minor industrial facilities, although some of these facilities may municipal facilities. Values to 2 significant figures.
2. Note that units may have been entered incorrectly in PCS. The minimum detection level for mercury Method 1631 is 0.0002 µg/L, but there are reported values that are nondetect less than that minimum level (e.g., <0.000026 µg/L).
3. Based on EPA PCS data from 2001 to 2004, and assuming facilities for which SIC codes are missing in PCS are minor industrial facilities, although some of these facilities may be municipal facilities. Values to 2 significant figures.
4. Excludes the concentration for one facility (NJ0005444, Outfall 014, maximum value 67.4 µg/L) from the average because it is an outlier. The estimated load from this facility, however, is added to the extrapolated load.
5. Reflects the median concentration, due to a small number of facilities with high concentrations.

Calculating average loadings by facility (e.g., by multiplying the average concentration by average flow and a conversion factor) and summing across facilities may not provide an accurate estimate of the total mercury loadings from point sources because facilities without limits for mercury (and thus no mercury data in PCS) do not necessarily have zero loadings. Indeed, if the average for facilities with concentration data is representative of similar facilities (major/minor, municipal/industrial) without data, loadings from point sources could be approximately 560 kg/year (assuming that nondetects are one-half the detection limit, which may overstate actual loadings when detection limits are high). **Exhibit A-5** summarizes this calculation.

Exhibit A-5. Potential Mercury Loadings to New Jersey Surface Waters

Facility Type	Average Mercury Concentration[1] (µg/L)	Average Facility Flow[2] (L/day)	Total Number of Facilities	Potential Load[3] (kg/year)
Major Municipals	0.15	46,000,000	100	250
Minor Municipals	0.19	860,000	93	5.5
Major Industrials	0.13[4]	9,300,000	59	26
Minor Industrials	0.17[5]	1,300,000	3,541	280
Total	NA	NA	3,793	560

Exhibit A-5. Potential Mercury Loadings to New Jersey Surface Waters

Facility Type	Average Mercury Concentration[1] (µg/L)	Average Facility Flow[2] (L/day)	Total Number of Facilities	Potential Load[3] (kg/year)

NA = not applicable.

1. Based on EPA PCS data from 2001 to 2004, assuming nondetects equal to one half of the detection limit and facilities for which SIC codes are missing in PCS are minor industrial facilities, although some of these facilities may be municipal facilities. Values to 2 significant figures.
2. Values to 2 significant figures.
3. Based on extrapolating average concentrations for facilities with mercury data (assuming nondetects equal to one half of the detection limit) to all facilities, by category. Values to 2 significant figures.
4. Excludes the concentration for one facility (NJ0005444, Outfall 014, maximum value 67.4 µg/L) from the average because it is an outlier. The estimated load from this facility, however, is added to the extrapolated load.
5. Reflects the median concentration, due to a small number of facilities with high concentrations.

Note, however, that for municipal wastewater treatment plants, mercury is not usually generated or introduced during the treatment processes. On rare occasions, mercury may be a contaminant in process chemicals such as chlorine, but, generally, mercury enters the plant from indirect dischargers such as dentists, hospitals, laboratories, and schools as well as from human waste and consumer products.

A.2 PCBs

There are no New Jersey-specific data that allow calculation of atmospheric deposition of PCBs Statewide. Atmospheric deposition varies considerably based on temperature and distance from major municipal areas (Eisenreich, 2000). Deposition increases with temperature and decreases with distance from the source. Data from near Chicago, IL indicates deposition rates are on the order of 0.1 µg/m^2 (Eisenreich, 2000).

The Delaware River Basin Commission (DRBC) has data on point source contributions of PCBs for the Delaware River Watershed, as shown in **Exhibit A-6**.

Exhibit A-6. Delaware River Basin Commission PCB Data

NPDES Number	Facility Type	PCB Loading	
		Milligrams per Day	Kilograms per Year
NJ0026182	Major Municipal	818.459	0.299
NJ0020923	Major Municipal	243.612	0.0889
NJ0026301	Major Municipal	220.791	0.0806
NJ0022519	Major Municipal	124.107	0.0453
NJ0023361	Major Municipal	123.392	0.0450
NJ0024686	Major Municipal	113.497	0.0414
NJ0025178	Major Municipal	67.433	0.0246
NJ0021598	Major Municipal	63.353	0.0231

Exhibit A-6. Delaware River Basin Commission PCB Data

NPDES Number	Facility Type	PCB Loading	
		Milligrams per Day	Kilograms per Year
NJ0024015	Major Municipal	54.904	0.0200
NJ0024660	Major Municipal	46.336	0.0169
NJ0023507	Major Municipal	37.419	0.0137
NJ0021709	Major Municipal	34.901	0.0127
NJ0024007	Major Municipal	27.98	0.0102
NJ0024678	Major Municipal	26.292	0.0096
NJ0024023	Major Municipal	23.206	0.00847
NJ0027481	Major Municipal	18.89	0.00689
NJ0023701	Major Municipal	15.682	0.00572
NJ0027545	Major Municipal	12.114	0.00442
NJ0021601	Major Municipal	10.265	0.00375
NJ0024856	Major Municipal	10.062	0.00367
NJ0024449	Minor Municipal	19.235	0.00702
NJ0030333	Minor Municipal	12.11	0.00442
NJ0020532	Minor Municipal	6.093	0.00222
NJ0021610	Minor Municipal	3.853	0.00141
NJ0022021	Minor Municipal	3.296	0.00120
NJ0005100	Major Industrial	241.334	0.0881
NJ0005029	Major Industrial	99.473	0.0363
NJ0004219	Major Industrial	82.206	0.0300
NJ0005401	Major Industrial	55.374	0.0202
NJ0005045	Major Industrial	12.228	0.00446
NJ0005240	Major Industrial	10.952	0.00400
NJ0005584	Major Industrial	7.358	0.00269
NJ0004995	Major Industrial	5.01	0.00183
NJ0005134	Major Industrial	4.12	0.00150
NJ0004286	Major Industrial	1.701	0.000621
NJ0004669	Major Industrial	1.528	0.000558
NJ0005622	Major Industrial	0.984	0.000359
NJ0005002	Major Industrial	0.929	0.000339
NJ0005185	Major Industrial	0.917	0.000335
NJ0025411	Major Industrial	0.915	0.000334
NJ0004391	Major Industrial	0.748	0.000273

Exhibit A-6. Delaware River Basin Commission PCB Data

NPDES Number	Facility Type	PCB Loading	
		Milligrams per Day	Kilograms per Year
NJ0004375	Major Industrial	0.33	0.000120
NJ0005363	Major Industrial	0.325	0.000119
NJ0004332	Major Industrial	0.07	0.0000256
NJ0004235	Minor Industrial	15.051	0.00549
NJ0033022	Minor Industrial	10.35	0.00378
NJ0004278	Minor Industrial	10.041	0.00366
NJ0005266	Minor Industrial	0.807	0.000295
NJ0064696	Minor Industrial	0.157	0.0000573
NJ0033952	Minor Industrial	0.087	0.0000318
NJ0131342	Minor Industrial	0.012	0.00000438
NJ0025413	Minor Industrial	0.011	0.00000402

Source: DRBC (1998).

These data indicate that CSOs contribute one to two orders of magnitude higher loads during wet weather than point sources and tributary inputs to the Delaware River (DRBC, 1998). Loadings from CSOs are also quantified for the NY/NJ Harbor by the New York/New Jersey Harbor Contamination Assessment and Reduction Project (CARP) (2003), and indicate that 11 CSOs in New York City contribute 0.01 to 38 kg/year, with a median load of 2.45 kg/year (**Exhibit A-7**).

Exhibit A-7. New York City CSO PCB Loading Data

CSO Name	PCB Loading	
	Grams per Hour	Kilograms per Year
CSO-26th Ward	4.3	37.7
CSO-Red Hook Influent	0.76	6.66
CSO-Newtown Creek Influent	0.45	3.94
CSO-Bowery Bay	0.42	3.68
CSO-Jamaica Influent	0.32	2.80
CSO-North River Influent	0.28	2.45
CSO-Wards Island	0.22	1.93
CSO-Hunts Point Influent	0.14	1.23
CSO-Owls Head Influent	0.097	0.850
CSO-Port Richmond Influent	0.092	0.806
CSO-Coney Island Influent	0.069	0.604

Exhibit A-7. New York City CSO PCB Loading Data

CSO Name	PCB Loading	
	Grams per Hour	Kilograms per Year

Source: New York/New Jersey Harbor CARP (2003).

If the loads from these facilities are similar to those discharged from CSOs in New Jersey, inputs to surface waters may be on the order of 690 kg/year (2.45 kg/yr (280 CSOs in New Jersey) (NJDEP, 2003).

Municipal wastewater treatment plants and industrial facilities discharging to the Delaware River contribute, on average, between 0.004 kg/day and 0.10 kg/day of PCBs (**Exhibit A-8**). If these data are representative of point sources elsewhere in the State, inputs to surface waters may be on the order of 10.6 kg/yr (Exhibit A-8). The appendix provides the facility level loadings.

Exhibit A-8. Potential PCB Loadings to New Jersey Surface Waters[1]

Facility Type	Number with PCB Data	Average PCB Load[2] (g/day)	Number Statewide	Load[2] (kg/year)
Major Municipals	20	0.10	100	3.6
Minor Municipals	5	0.0089	93	0.33
Major Industrials	19	0.028	59	0.60
Minor Industrials	8	0.0046	3,541	5.9
Total	52	NA	3,793	10.6

1. Based on DRBC (1998), calculating average load by facility type, and multiplying by the number of facilities in the State to estimate total load. Also assuming that all facilities without SIC codes assumed to be industrial minor facilities, although a number of these facilities are known to be municipal facilities.
2. Values to 2 significant figures.

The PCB data from 12 major publicly owned treatment works (POTWs) near New York City (**Exhibit A-9**), however, indicate significantly higher loadings, and imply an average loading of approximately 4.3 kg/year per facility. This is on the order of 40 times the average load from the major POTWs in the Delaware River watershed. This difference could be real in that there may be higher PCB deposition in the New York City area, and since most of the POTWs in that area have combined sewers, atmospheric deposition might account for the difference.

Exhibit A-9. PCB Loading Data for Major Municipal Facilities

NPDES Number	PCB Loading	
	Pounds per Day	Kilograms per Year
NJ0021016	0.23016	38
NJ0020141	0.03689	6.1

Exhibit A-9. PCB Loading Data for Major Municipal Facilities

NPDES Number	PCB Loading	
	Pounds per Day	Kilograms per Year
NJ0020028	0.02168	3.6
NJ0024741	0.01277	2.1
NJ0024643	0.00486	0.80
NJ0024953	0.00308	0.51
NJ0151122	0.00233	0.39
NJ0024813	0.00197	0.33
NJ0025321	0.00125	0.21
NJ0020591	0.00055	0.091
NJ0029084	0.00052	0.086
NJ0025038	0.00042	0.070

Source: New Jersey Toxics Reduction Program (2003).

EPA's PCS indicates that 3 facilities in New Jersey have PCB data. However, all but two observations are nondetects (with detection limits ranging from 0.03 to 6.2 μg/L). Similar to mercury, the absence of facilities with limits and data for PCBs in PCS does not necessarily mean that there are not loadings from these facilities in the ng/L range.

Note that for municipal and industrial dischargers, PCBs should not be generated or introduced during the treatment processes because use and manufacturing of PCBs has been banned in the United States since 1979. The Palo Alto Regional Water Quality Control Plant (RWQCP) conducted a study to identify the sources of PCBs in their influent. Although there is great uncertainty surrounding the estimated quantity of PCB loadings from various sources, the facility determined that the greatest sources of PCBs are the water supply (i.e., drinking water) and storm water, with sources such as landfill leachate, hazardous waste management, human waste, and food waste also accounting for a portion of the influent load (EIP Associates, 1997a). Similar sources could also contribute to PCB loadings at industrial facilities (especially storm water runoff). Therefore, at least part of the estimated loading from point sources may be attributable to atmospheric deposition (that ends up in the water supply and storm water).

A.3 DDT

Data on sources of DDT to surface water are extremely limited, especially in New Jersey. There are no New Jersey facilities in PCS that have limits or data for DDT. The only available data are from New Jersey's Toxic Reduction Program (2003) for 12 POTWs in the New York City area. These data (**Exhibit A-10**) indicate an average loading of approximately 0.000354 kg/day, or 0.13 kg/year. All analytical results for DDT for nine POTWs in the Great Lakes (U.S. EPA, 1994) are nondetects.

Exhibit A-10. DDT Loading Data for Major Municipal Facilities

NPDES Number	Facility Type	DDT Loading	
		Pounds per Day	Kilograms per Year
NJ0021016	Major Municipal	0.00301	0.498
NJ0024741	Major Municipal	0.00142	0.235
NJ0020028	Major Municipal	0.00135	0.224
NJ0020141	Major Municipal	0.00101	0.167
NJ0024643	Major Municipal	0.00066	0.109
NJ0024953	Major Municipal	0.00061	0.101
NJ0025321	Major Municipal	0.00041	0.0679
NJ0151122	Minor Municipal	0.00033	0.0546
NJ0024813	Major Municipal	0.00021	0.0348
NJ0025038	Major Municipal	0.00017	0.0281
NJ0029084	Major Municipal	0.00015	0.0248
NJ0020591	Major Municipal	0.00003	0.00497

Source: New Jersey Toxics Reduction Program (2003).

For municipal and industrial dischargers, DDT should not be generated or introduced during the treatment processes because EPA banned all uses, including exportation by 1988. However, DDT is still used in other countries. The Palo Alto RWQCP conducted a study to identify the sources of organochlorine pesticides including DDT in their influent. Again, due to a lack of data and limitations in analytical methods there is great uncertainty surrounding the estimated quantity of DDT loadings from various sources. However, the facility determined that the greatest sources of PCBs are storm water, human waste, and food waste. Other less significant sources include landfills and hazardous waste management (EIP Associates, 1997b). Therefore, at least part of the estimated loading from point sources may be attributable to atmospheric deposition (that ends up in storm water).

A.4 References

Delaware River Basin Commission (DRBC). 1998. Study of the Loadings of Polychlorinated Biphenyls from Tributaries and Point Sources Discharging to the Tidal Delaware River. June 1998.

EIP Associates. 1997a. Polychlorinated Biphenyls Source Identification. Prepared for Palo Alto Regional Water Quality Control Board.

EIP Associates. 1997b. Organochlorine Pesticides Source Identification. Prepared for Palo Alto Regional Water Quality Control Board.

Eisenreich, Steven. 2000. Polychlorinated Biphenyl Emissions to Urban Atmospheres: Enhanced Concentrations, Atmospheric Dynamics, and Controlling Processes. Online at http://www.delta-institute.org/publications/EisenreichWP.pdf.

National Atmospheric Deposition Program (NADP). 2003. Total Mercury Wet Deposition. Online at http://nadp.sws.uiuc.edu/mdn/maps/2003/03MDNdepo.pdf.

New Jersey Department of Environmental Protection (NJDEP). 2003. New Jersey Pollutant Discharge Elimination System (NJPDES) General Permit for Combined Sewer Systems (CSS) NJPDES No. NJ0105023. Online at http://www.state.nj.us/dep/dwq/pdf/cso_rfa.pdf.

New Jersey Toxics Reduction Program. 2003. Collection and Analysis of New Jersey POTW Effluents for Trace Contaminants Entering the NY-NJ Harbor. Online at http://www.state.nj.us/transportation/works/maritime/pdf/treatmentplant.pdf.

New York/New Jersey Harbor Contamination Assessment and Reduction Project (CARP). 2003. Online at http://www.dec.state.ny.us/website/dow/bwam/CARP.

NSTATE LLC. 2004. The Geography of New Jersey. Online at http://www.netstate.com/states/geography/nj_geography.htm.

U.S. EPA. 2004. Permit Compliance System Database. Accessed July 1, 2004. Online at http://www.epa.gov/enviro/html/pcs/adhoc.html.

U.S. EPA. 1996. CWA 303(d): Establishment of Phased Total Maximum Daily Loads for Copper, Mercury, Nickel and Lead in New York-New Jersey Harbor. Federal Register: January 24, 1996, Vol. 61, No. 16, pp. 1930-1932.

U.S. EPA. 1994. An Analytical Survey of Nine POTWs from the Great Lakes Basin. Office of Science and Technology. December 15, 1994.

Appendix B. Economic Considerations

Because there are no proven technologies for achieving the proposed water quality standards, or analytical methods for establishing compliance with the DDT and PCB criteria, evaluation of economic feasibility is not warranted. That is, with currently available data, it is not possible to reliably estimate which facilities would need to pursue pollution prevention or add wastewater treatment, and what the result of such actions would be (e.g., compliance with criteria end-of-pipe), to calculate the costs of achieving standards.

Despite the lack of field trials and analytical methods, in theory, the end-of-pipe technologies discussed in the body of this report could produce effluent levels necessary for compliance with the proposed criteria end-of-pipe. However, currently, these methods are costly, and implementation could impose substantial costs on a facility with no guarantee that it would be able to comply with its permit limits. This Appendix discusses the types of cost considerations applicable to each technology.

Pollution prevention techniques are also not proven to result in effluent levels that would achieve the proposed criteria end-of-pipe. However, these techniques may be the most cost-effective methods for making progress towards achieving the standards, because they focus on eliminating the pollutants from use, and removing remaining concentrations from the effluent at the source where they are more concentrated and easier/more cost-effective to identify and treat. This appendix also discusses the types of costs that could be applicable to a pollution prevention strategy.

B.1 Method for Estimating Costs

Because the treatment technologies discussed in the body of this report have not been proven to effectively reduce effluent pollutant concentrations to the levels necessary at full-scale treatment plants, cost curves and detailed case studies have not been developed that necessarily reflect the level of treatment needed. However, cost curves for the technologies based on removing other contaminants may provide a rough, order-of-magnitude estimate of costs.

The costs for all the technologies discussed will vary by type of facility (e.g., municipal and industrial) and size (e.g., flow). As an example of how costs may vary, information is shown for five combinations of facility type and size (**Exhibit B-1**).

Exhibit B-1. Example Categories of Potentially Affected Facilities[1]

Facility Type	Design Flow (mgd)	Average Flow[1] (mgd)
Large Flow Facility	100	85
Major Municipal	25	18.8
Minor Municipal	0.5	0.38
Major Industrial	10	7.5

Exhibit B-1. Example Categories of Potentially Affected Facilities[1]

Facility Type	Design Flow (mgd)	Average Flow[1] (mgd)
Minor Industrial	0.1	0.08

1. Assumed to be 75% of design flow.

Because the exact configuration and operating parameters for many of the treatment technologies considered in this report have not been tested, assumptions have to be made to develop any cost estimates (e.g., for mercury removal, a selective resin would be used in a column similar to one used for activated alumina). Assumptions regarding pH adjustment, pretreatment and post treatment requirements will greatly influence costs.

The estimates below also represent applications of the technologies for drinking water treatment. Therefore, actual costs applicable to wastewater treatment plants may be greater due to increased suspended solids, and competing contaminants that could act to lower removal or destruction efficiencies. In addition, some of the assumptions used to estimate costs for drinking water applications may vary from those assumptions applicable to the treatment of wastewater (e.g., drinking water residuals are discharged to a POTW, whereas, POTW residuals must be treated prior to disposal). Therefore, costs for components such as residual treatment and disposal are incorporated separately, where needed.

B.2 Reverse Osmosis

To remove mercury to concentrations less than 0.5 ng/L, multiple RO units will most likely have to be run in series because RO does not selectively remove dissolved contaminants. The number of RO units needed will depend on flow, influent mercury concentration, the existing treatment train (e.g., pretreatment), and presence of contaminants that may cause fouling or decrease the mercury removal efficiency. There are no performance data from a full-scale treatment plant indicating the number of RO units necessary to achieve 0.5 ng/L, however, the addition of a second unit would be less costly than the first because it would be smaller (less flow would need to be treated because about 70% - 90% of the influent water is recovered and sent to the next unit, while the remaining 10% - 30% becomes residual brine).

Although there is no recent cost information (e.g., case studies, cost curves) specific to wastewater treatment facilities available, there is cost information for drinking water facilities that EPA used to estimate costs for drinking water regulations such as the national primary drinking water regulations for radionuclides in potable water. While costs for drinking water applications may underestimate costs for wastewater applications, they can still be used to derive rough, order of magnitude estimates of treatment costs. Costs for pretreatment and operating costs may be greater for wastewater treatment due to the quality of influent water (e.g., greater suspended solids may increasing fouling or decrease run time of the media which increases costs). Post-treatment costs would be permit-specific and may not be necessary for RO unless pH adjustment or additional treatment to meet other permit limits is necessary.

Exhibit B-2 shows how capital and operation and maintenance (O&M) costs may vary for the categories in Exhibit B-1 based on operating two RO units in series with microfiltration pretreatment. Annual costs for major facilities ranges from about $1,300,000 to $1,600,000 per mgd; costs range from about $2,000,000 to $3,300,000 per mgd for minor facilities.

Exhibit B-2. Potential Reverse Osmosis Costs for Example Facility Types (2004$)[1]

Facility Type	Design Flow (mgd)	Capital Costs ($1,000)	O&M Costs ($1,000)	Total Annual Costs[2] ($1,000)	Annual Unit Costs ($/mgd)[3]
Large Flow Facility	100	$447,000	$85,400	$127,600	$1,280,000
Major Municipal	25	$129,400	$22,300	$34,600	$1,380,000
Minor Municipal	0.5	$2,800	$730	$990	$1,980,000
Major Industrial	10	$61,900	$9,800	$15,600	$1,560,000
Minor Industrial	0.1	$1,100	$230	$330	$3,310,000

mgd = millions of gallons per day

Source: U.S. EPA (2000a; 2003). Escalated to 2004 dollars using the ENR Construction Cost Index (CCI) for capital costs and the Consumer Price Index (CPI) for O&M costs.

1. Includes costs for microfiltration pretreatment and two RO units in series. Residual management costs are not included.
2. Capital costs annualized at 7% over 20 years plus annual O&M costs.
3. Calculated by dividing total annual costs by the design flow in mgd.

The costs assume that the recovery rate for systems less than 10 mgd is 80% and 85% for systems greater than 10 mgd. Therefore, the costs for the second RO unit are based on those recovery rates. Also included is the cost of microfiltration pretreatment. The costs have also been adjusted to reflect the change in the technology since the original development of costs. Capital and O&M costs for RO spiral-wound membranes have decreased by about 50% due to advances in the industry and its use in more applications (U.S. EPA, 2000a). Therefore, U.S. EPA (2000a) deflated both capital and O&M costs by 50% to reflect this change.

U.S. EPA (2000a) indicates that chemical precipitation can be used to treat the residual brine from RO. The supernatant from the chemical precipitation can be recycled to the head of the plant, however, the solid sludge would need to be dewatered similar to conventional wastewater sludge, sent to an evaporation pond, or be mechanically evaporated. Costs for the chemical precipitation alone (not including the additional sludge treatment) can range from about 1% to 6% of capital costs, and about 4% to 12% of annual O&M costs (U.S. EPA, 2000a).

The most common residual treatment for RO brine is an evaporation lagoon. However, due to the wet climate and lack of available of land such an option may not be feasible in New Jersey, especially for larger facilities in which residuals may be more than 30% of influent flows. Multi-stage flash evaporators or shipment to off-site treatment plants may be necessary to handle residuals, both of which could greatly increase costs. For example, a multi-stage flash evaporator that can treat 0.5 mgd costs about $3 million (Randall, 2004). However, once the liquid residuals are evaporated, the remaining solids would need to be sent to a landfill for disposal. Annual transportation costs could be about the same as the total capital costs for the flash evaporator unit (costing about $2,000 per trip for a truck that can haul 20 tons).

B.3 Selective Sorbents

Costs for selective sorbent columns depend on the volume of water to be treated, the influent mercury concentrations, the media used, and source water quality (i.e., pretreatment necessary). No selective sorbents have been tested and proven to remove mercury to 0.5 ng/L at a full-scale treatment plant. The sorbents would likely be used in a column, with water passing through the column, similar to activated alumina. Many of the sorbents operate best within specific pH and temperature ranges. Therefore, the following assumptions are used to illustrate costs for a selective sorbent system:

- pH adjustment would be necessary - for example, the Keyle:X resin operates best at a pH between 5.5 and 6 (although it can function at pH values that range from 4 to 12)
- microfiltration pretreatment is needed to remove the colloidal fraction of mercury from solution
- nanofiltration post-treatment is needed to remove adsorbed mercury particles in the nanometer size range (necessary to achieve low nanogram per liter levels)

Exhibit B-3 summarizes potential costs for the categories of facilities described in Exhibit B-1.

Exhibit B-3. Potential Selective Sorbent Costs for Example Facility Types (2004$)[1]

Facility Type	Design Flow (mgd)	Capital Costs[1] ($1,000)	O&M Costs ($1,000)	Total Annual Costs[2] ($1,000)	Annual Unit Costs ($/mgd)[3]
Large Flow Facility	100	$160,900	$41,700	$56,900	$570,000
Major Municipal	25	$42,600	$10,300	$14,400	$570,000
Minor Municipal	0.5	$2,400	$280	$510	$1,000,000
Major Industrial	10	$18,200	$4,200	$5,900	$590,000
Minor Industrial	0.1	$1,600	$110	$260	$2,600,000

mgd = millions of gallons per day

Source: U.S. EPA (2000b; 2003). Escalated to 2004 dollars using the ENR CCI for capital costs and the CPI for O&M costs.

1. Includes costs for microfiltration pretreatment, activated alumina column system, and nanofiltration post-treatment. Residual management costs are not included for pre- and post-treatment. Assumes that selective resin is not regenerated and disposed of in landfill once spent.
2. Capital costs annualized at 7% over 20 years plus annual O&M costs.
3. Calculated by dividing total annual costs by the design flow in mgd.

These costs do not include the cost of any specific sorbent resin. In Exhibit B-3, resin costs are about $1 per pound (U.S. EPA, 2000b). However, costs may greatly increase given the resin selected. For example, the Keyle:X resin costs about $47 per pound (Solmetex, 2004), or 47 times the cost of the resin used to calculate O&M costs in Exhibit B-3 (Note that not all of the

O&M costs are for resin). At the same time, when designing a treatment system, various parameters may be modified to balance capital and O&M costs. For example, increasing the depth of the column will increase initial capital costs because more resin will be required for start-up, however, it will significantly reduce O&M costs because the larger depth will increase the time between resin replacements, thus, reducing the annual material costs.

The costs in Exhibit B-3 include the cost of disposing the selective resin in a landfill (assuming that resin is not regenerated), however, they do not include the cost of residuals management associated with micro- or nanofiltration. Residual disposal costs will depend on the pre- and post-treatment controls needed, as well as the adsorptive capacity of the resin and whether backwashing and regeneration of the resin is necessary. For smaller facilities, evaporation lagoons may be a feasible option. Another feasible and relatively inexpensive residual disposal option would be to ship residuals to a centralized waste treatment facility such as the Dupont Chambersworks facility in Deepwater, New Jersey. However, large facilities may need to treat their residuals prior disposal. In microfiltration, about 7% of the influent will become residuals, and about 15% for nanofiltration. The microfiltration residuals can be treated similar to typical wastewater sludge, whereas the nanofiltration residuals are similar to RO brine. If evaporation ponds or off-site residual treatment are not feasible, more costly options such as multi-stage flash evaporation or brine concentration followed by crystallization may be necessary. The magnitude of these costs are discussed above in Section B.2.

B.4 Mercury Precipitation

Costs for mercury precipitation followed by activated alumina will be similar to the costs for selective sorbents. An oxidant would be added to the influent in the same manner that chemicals would be added for pH adjustment for selective sorbents. This pre-chlorination step may not be necessary for all dischargers though. For municipal dischargers, activated alumina would come after disinfection (with either chlorine, ozone, or UV), where elemental mercury would be oxidized by the disinfectant. Costs for additional oxidation chemicals would most likely be minimal following disinfection. For industrial dischargers, activated alumina would be the last polishing process prior to discharge. Because most industrial dischargers usually use chlorine or ozone to prevent scaling and algae growth, as a disinfectant, or to oxidize other chemicals prior to discharge, costs for additional oxidation chemicals needed to oxidize mercury at such low levels would also most likely be minimal.

It is likely that microfiltration would be necessary to ensure optimal mercury removal and prevent fouling of the activated alumina. Depending on the mercury removal efficiencies obtained, post-treatment with nanofiltration may or may not be necessary. Residual disposal costs would also be similar and depend on the same factors. Therefore, based on the facility type characteristics described in Exhibit A-1, potential annual costs for major facilities could be about $600,000 per mgd; costs may range from about $1,000,000 to $3,000,000 per mgd for minor facilities.

B.5 Granular Activated Carbon

Estimates from one vendor indicate that annual costs for a GAC system designed to remove PCBs may range from about $17,000 to $26,000 per mgd, depending on flow (Sherbondy, 2004). To remove DDT, the pH of the system may need to be adjusted, increasing annual costs to about $19,000 to $31,000 per mgd (Sherbondy, 2004). These costs do not include components such as pretreatment, residual disposal, process piping, site work, pilot studies, administrative fees, and a contingency factor. In addition, more than one GAC unit in series may be necessary to reduce PCBs to the required levels. The costs may also vary considerably based on source water quality and influent contaminant concentration.

U.S. EPA (2000b; 2003) provides estimates for two GAC units in series, and including pretreatment, residual disposal, process piping, site work, pilot studies, administrative fees, and contingencies. Based on this data, costs for the example facility types and sizes described in Exhibit A-1 may range from about $140,000 to $190,000 per mgd for major facilities, and from about $880,000 to $3,900,000 per mgd for minor facilities. **Exhibit B-4** summarizes these costs.

Exhibit B-4. Potential GAC Costs for Example Facility Types (2004$)[1]

Facility Type	Design Flow (mgd)	Capital Costs ($1,000)	O&M Costs ($1,000)	Total Annual Costs[2] ($1,000)	Annual Unit Costs ($/mgd)[3]
Large Flow Facility	100	$70,000	$7,700	$14,400	$140,000
Major Municipal	25	$20,600	$2,000	$4,000	$160,000
Minor Municipal	0.5	$2,400	$220	$440	$880,000
Major Industrial	10	$10,100	$950	$1,900	$190,000
Minor Industrial	0.1	$2,100	$190	$390	$3,900,000

mgd = millions of gallons per day

Source: U.S. EPA (2000b; 2003). Escalated to 2004 dollars using the ENR CCI for capital costs and the CPI for O&M costs.

1. Costs for pretreatment with coagulation filtration, and two GAC units in series with an empty bed contact time of 20 minutes and a reactivation frequency of 90 days. Costs for the second GAC unit are assumed to be 75% of cost for the original unit because permitting, piloting, redundancy, regeneration units, and operator training costs would only be incurred once if both units are installed at the same time. For smaller systems the GAC is sent offsite for regeneration, and for larger systems it is regenerated onsite. Residuals management costs are not included.
2. Capital costs annualized at 7% over 20 years plus annual O&M costs.
3. Calculated by dividing total annual costs by the design flow in mgd.

These costs do not include the cost of residuals management associated with pretreatment or GAC. Residual disposal costs will depend on type of filtration used as pretreatment, and whether or not the GAC is regenerated onsite, off-site, or used on a throwaway basis. For smaller facilities, or where land is readily available, evaporation lagoons may be a feasible option. However, large facilities, or areas in which land is scarce, may need to treat their residuals prior disposal. The residuals produced from both filtration and GAC will be similar to conventional wastewater treatment plant residuals. One option would be to construct a holding tank or equalization basin so that the current sludge dewatering equipment can be used to treat the additional residuals.

B.6 Ultraviolet Oxidation

Potential costs for UV oxidation differ depending on whether the process is coupled with ozone, hydrogen peroxide, or titanium dioxide catalyst. The major difference between the two processes is that UV oxidation with ozone or hydrogen peroxide requires the generation and handling of ozone and hydrogen peroxide onsite, whereas UV/titanium dioxide oxidation does not require the handling of any hazardous chemicals. Also, with UV oxidation with titanium dioxide, if the more efficient powder form of titanium dioxide is used, a mechanism for recovering the catalyst is necessary. Both processes will have relatively high energy costs due to the use of UV light. **Exhibit B-5** provides a qualitative comparison of operating costs for the various UV oxidation systems.

Exhibit B-5. Comparison of Operating Costs for Various UV Oxidation Processes

Process	Cost of Oxidant	Cost of UV
UV/Ozone	High	Medium
UV/Hydrogen Peroxide	Medium	High
UV/Titanium Dioxide	Very Low	Medium to High

Source: Munter (2001).

Note, however, that the efficiency of each process is strongly dependent on the specific compounds targeted (Munter, 2001). Preliminary bench testing should be conducted to determine the appropriate oxidation process for removal of PCBs from a particular type of wastewater.

FRTR (2002) indicates that costs for UV oxidation system using ozone and/or hydrogen peroxide for the treatment of contaminated groundwater are generally between $36,000 to $3,650,000 per mgd. Costs may be slightly greater for wastewater treatment due to increases in the variability of water quality as well as additional pretreatment requirements (e.g., filtration). The major operating cost for the ozone/hydrogen peroxide system is the cost of electricity for generating the chemicals (Munter, 2001).

Hand et al. (1995) estimated that annual costs for a 1 mgd UV oxidation system using titanium dioxide range from approximately $139,000 to $433,000 per mgd. Note that these costs were developed when this technology was still fairly new, therefore, actual costs may be lower due to advances in the technology or an increase in the number of manufacturers.

Factors that may influence the cost of UV oxidation include:

- Types and concentration of contaminants - affect oxidizer selection, oxidizer dose, UV light intensity, and contact time
- Influent concentration (i.e., destruction efficiency required)
- Flow

- Pretreatment and post treatment requirements (FRTR, 2002).

Neither of the cost estimates above include pre- or post-treatment costs. To achieve such low levels of PCBs and DDT, pretreatment is necessary to ensure that the UV rays effectively reach and destroy the PCB and DDT compounds. Suspended solids and other contaminants may reduce the destruction efficiency of UV oxidation. Sand or dual media filtration should be sufficient for this purpose, however, the exact type of treatment necessary will be site-specific and depend on influent water quality.

B.7 Sonochemical Degradation

There are many engineering and scientific issues such as scaling up bench test to full scale processes, variation of the sound field in reactors, and effects of source water quality associated with sonochemical degradation that still need to be resolved before the technology can be used on a commercial scale. Cost information for this technology is not available. However, since the main mechanism involved in sonochemical degradation is the same mechanism that destroys PCBs and DDT in UV oxidation (i.e., radical chemistry), it is not likely that sonochemical degradation would be used in place of UV oxidation, especially given the limited performance data for sonochemical degradation.

B.8 Pollution Prevention

P2 programs comprise identifying the sources of mercury in a facility's influent, and then developing and implementing activities that reduce the loadings from these sources that enter a facility's headworks. The cost of P2 programs to municipalities will vary depending on a number of factors (such as the number of industrial and commercial dischargers, population served), and actual costs may only be known after source identification has occurred. There is little information available on these costs because few programs have been implemented, and the costs may not be accounted for as a separate budget item that can be verified apart from other base program costs.

Direct Program Costs

One way to estimate the potential costs that could be incurred is to identify likely P2 program components based on the experiences of facilities that have already developed P2 programs. Then, unit costs could be developed for each component. For example, **Exhibit B-6** summarizes the various tasks that may be associated with developing a municipal P2 program, and the potential components of costs for each task.

Exhibit B-6. Summary of Tasks for Typical P2 Program

Task	Cost Components
Program Planning/Legislative Support	Participation (labor): workgroup head, outreach specialist, technical and legal experts

Exhibit B-6. Summary of Tasks for Typical P2 Program

Task	Cost Components
Source Identification	Labor and laboratory costs for sampling and analysis, and developing mass balance
Develop Materials	Labor and materials to: develop mailing lists, produce brochures and online materials, develop website, produce public service announcements; postage
Implement Pilot Programs	Labor to identify and work with pilot facilities
Perform Site Visits	Labor: technical expert
Conduct Workshops	Labor: technical expert, outreach specialist; materials

However, as the frequency of such activities increases, information available from facilities that have already implemented P2 may enable other municipalities to implement similar programs at a lower cost. Therefore, costs that have been incurred by facilities that have already implemented P2 programs may not be indicative of the cost of implementing P2 on a larger scale. For example, POTWs can lower program development and implementation costs by collaborating with other facilities and trade associations to develop outreach materials and pilot programs. The time requirements to develop P2 materials can be reduced if materials developed by other facilities can be used with minimal alteration, or if trade associations prepare materials for members to use.

Ongoing national efforts (e.g., to implement P2 programs in hospitals) and potential future legislative developments (e.g., requiring amalgam separators in dental offices) may also reduce the cost and need for municipal P2. At the same time, the nature of the requirement for implementing P2 will also affect costs: for example, costs could vary widely depending on whether only relatively cost-effective measures are implemented, or whether actions are required until compliance with applicable water quality based effluent limits is demonstrated.

Indirect Control Costs

P2 costs also include the indirect costs of controls on sources of the pollutants, such as the use of amalgam separators for dental offices and product substitution at hospitals. The cost of technologies such as amalgam separators can be fairly easily estimated (see **Exhibit B-7**). However, costs associated with product substitution may be more difficult to obtain: some facilities may actually experience cost savings from eliminating the use of some pollutants due to reduced disposal or emergency training requirements. In addition, current costs may not reflect the prices that could be obtained by industries through large scale, group purchases.

Exhibit B-7. Potential Unit Cost for Amalgam Separators

Amalgam Separator	Capital Costs	O&M Costs	Total Annual Cost
DRNA Particle Trap and Amalgam Separator	-	$1,575[1]	$1,575
DRNA Fine Particle Filters & Absorbent Columns	-	$2,175[1]	$2,175

Exhibit B-7. Potential Unit Cost for Amalgam Separators

Amalgam Separator	Capital Costs	O&M Costs	Total Annual Cost
SolmeteX	$1,250	$900[2]	$1,078
Average	$417	$1,550	$1,609

Source: EIP Associates (2000).
1. Represents annual cost of equipment lease and recycling and particle trap replacement costs.
2. Represents replacement cups replacement costs and recycling costs.

Hospitals and Medical Centers. P2 measures at hospitals and medical centers include numerous BMPs such as eliminating the use and handling of mercury-containing products and equipment through the modification of purchasing practices. Mercury-free substitutes are available for most mercury-containing chemicals and equipment. On the basis of initial purchase price, the mercury-free alternatives are generally more expensive than the mercury-containing item. However, the purchase price does not reflect the costs of clean up, storage and disposal, hazardous waste training, potential health risks to staff, patients, and visitors, potential noncompliance costs, and, in the case of electronic thermometers, time savings since an electronic thermometer gives a quicker temperature reading than a mercury thermometer (Pollution Probe, 1996). For example, comparison of a mercury-containing sphygmomanometer to a mercury-free aneroid sphygmomanometer shows that once staff training, spill cleanup, and administrative costs are taken into account, the mercury-free alternative is actually more cost-effective (Pollution Probe, 1996).

The University of Minnesota-Duluth reports that phasing out mercury has significantly reduced costs due to hazardous spill cleanups (Second Nature, 2003). The average wage rate of spill team members ranges from $15-$20 per hour and may take anywhere from 15 minutes to 12 hours to clean up. Spill kits range in costs from $15 to $200. Using mercury also requires administrative costs to keep procedures up to date and staff trained. The Butterworth hospital in Grand Rapids, Michigan spent over $3,000 per mercury spill for clean up for several spills between 1993 and 1995 (Pollution Probe, 1996).

Another example of cost savings as a result of mercury BMPs involves Kaiser-Permanente's integrated health delivery system. Kaiser-Permanente owns and operates 30 hospitals and 360 clinics. In 1998, they began a mercury minimization policy aimed at switching to mercury-free thermometers and sphygmomanometers, and proper disposal of flourescent lamps, through a contract with a recycler. Kaiser-Permanente predicted that elimination of mercury thermometers and sphygmomanometers would result in cost reductions of about $550,000 over the next 10 years across the system. These cost savings would also offset the cost of the recycling program (Tellus Institute, 2000).

Laboratories. The P2 measures for laboratories that may reduce the amount of mercury released to the environment as a result of daily operations are similar to those implemented by hospitals and medical centers. In addition to replacing mercury containing equipment (e.g., manometers) and chemicals containing mercury with mercury-free alternatives, laboratories can also work to minimize the amount of waste generated during experiments and testing procedures. The

University of Minnesota-Duluth has instituted micro-scale projects in undergraduate labs that dramatically reduced the quantities of possible mercury-containing chemicals used, purchased, and discarded. One professor reported reducing solvent waste from 2,500 gallons to 23 gallons, saving $37,000 a year (Second Nature, 2003). Laboratories in hospitals that have replaced their mercury-containing chemicals with mercury-free alternatives found that the cost of the mercury-free chemicals are about the same as the cost of the chemicals containing mercury (Pollution Probe, 1996).

There may be instances, however, in which mercury-free alternatives do not exist. A 250-employee laboratory in Massachusetts uses thimerosal as a bacteriostat/fungistat in many clinical tests. The facility is permitted to discharge up to 1 µg/L under the Massachusetts Water Resources Authority (MWRA) pretreatment program. The laboratory determined that the thimerosal could not be readily replaced, so the laboratory employed source reduction techniques (e.g., collecting waste streams for hazardous waste disposal, reducing the amount of reagent used) to reduce their mercury concentrations by 90%. The 90% reduction was not great enough for compliance with the pretreatment standard, and the facility had to install silver-impregnated granular activated carbon with flow equalization to reduce mercury to the required levels (Massachusetts OTA, 1995). However, since this case study, manufacturers have been working to create more alternatives for mercury-containing chemicals.

Secondary Schools. There are a number of BMPs schools can implement to reduce mercury in their wastewater:

- Educate students, other teachers and administrators about the health hazards and environmental fate of mercury (e.g., see the Mercury in Schools Pollution Prevention project, located at http://www.mercuryinschools.uwex.edu)
- Promote proper management and recycling of mercury and mercury-containing products (e.g., educate teachers and maintenance personnel on items that may contain mercury such as thermometers and laboratory chemicals, and proper disposal techniques)
- Eliminate the use of mercury wherever possible and promote the use of alternative products that do not contain mercury (e.g., schools may contain mercury-containing thermostats, barometers, thermometers, and wall switches that can easily be replaced with mercury-free alternatives)
- Clean out plumbing (e.g., mercury builds up in plumbing over the years resulting in a constant mercury discharge even after the use of mercury is eliminated).

Schools would most likely take an inventory of mercury and mercury containing equipment in each building, replace each item with a mercury free alternative, and set up an educational program for teachers and students on the health effects of mercury and its environmental fate. Activities such as cleaning out of plumbing and traps would likely not be cost effective because the mercury load from schools is usually minor, and the costs associated with clean out are high (e.g., $5,000 per school). However, the potential costs savings are large: In 2003, at a District of Columbia public highschool, a mercury spill, which closed the school for over 2 weeks and contaminated more than 11 student homes, cost over $1 million to clean up (Blume and Dvorak, 2003).

Vehicle Service Centers. Vehicle service centers should implement BMPs to reduce the amount of mercury discharged to the POTW. BMPs may include employee training on spill prevention, proper disposal methods for mercury-containing tilt switches, and waste minimization practices that reduce the amount of wastewater generated. Association of Metropolitan Sewerage Agencies (AMSA) (2002) suggests that the average cost of implementing BMPs at vehicle service centers is about $1,025 per facility.[1]

Industrial Facilities. The BMPs that an industrial discharger would pursue to reduce mercury in its effluent depends entirely on the type of industrial discharger, the amount of wastewater generated (e.g., flow that needs to be treated), the processes employed, and the chemicals and raw materials used. As such, these costs are highly site-specific. However, a P2 program could require all industrial dischargers to sample their effluent (e.g., using mercury Method 1631), which can me more readily estimated (see below).

Monitoring Costs

Sampling costs for municipalities developing P2 programs, as well as indirect dischargers, would depend on the frequency (e.g., twice per month) and length (e.g., one year, unless concentrations of concern are found) of any monitoring requirements. Additional labor requirements could be minimal if facilities are already monitoring with other methods, or for other pollutants.

As an example of potential costs, in 1999, Ohio EPA surveyed laboratories that provide contract analytical services using EPA Method 1631. The average sample cost across laboratories was $80. Some facilities may need to monitor fish tissue concentrations of the pollutants. For example, for mercury, SRCSD (2003) indicates that monitoring costs include boat rental, labor, sample pick up and delivery, sample dissection and digestion, and laboratory analysis of mercury. **Exhibit B-8** summarizes the unit costs for each monitoring component.

Exhibit B-8. Summary of Fish Tissue Monitoring Costs for Mercury

Cost Component	Unit Cost	Frequency
Boat Rental	$650/day	1 day per event
Sampling Labor	$47/hour	8 hours per event
Sample pick up and delivery	$0.34/mile	100 miles round-trip per event
Dissection and digestion	$91/fish	12 fish per event
Mercury analysis	$99/composite	6 composites per event
Total	$2,746/event	3 events per year

Source: SRCSD (2003).

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[1] This cost represents the cost of implementing BMPs, and excludes the cost of site visits attributable to a POTW.

U.S. EPA (2000c) indicates that facilities should choose two target species to sample. Freshwater dischargers should select one bottom feeder and one predator fish species, and marine or estuarine dischargers should select either one bivalve and one fin fish species, or two fin fish species. At least one composite sample for each of three size classes should be analyzed for each target species, with at least two fish per composite. This would result in collection of a minimum of 12 fish per sampling event (2 fish per composite x 1 composite per target species x 3 sizes classes per target species x 2 target species = 12 fish). EPA's guidance also indicates that samples should be collected during the period when the target species is most frequently harvested. In fresh waters, generally, the most desirable sampling period is from late summer to early fall (i.e., August to October) (U.S. EPA, 2000c). In estuarine and marine waters, the most appropriate sampling time is during the period when most fish are caught and consumed (usually summer for recreational and subsistence fishers) (U.S. EPA, 2000c). Thus, sampling could entail 3 trips during these time periods, with 12 fish collected per trip.

B.9 Summary

There are a number of considerations in estimating the potential costs of available technologies for removing mercury, PCBs, and DDT from wastewater. In addition, under the proposed standards, a facility could receive revised effluent limits for all three contaminants, affecting the treatment controls it implements as well as the cost of compliance. Additional testing would be needed to determine the sequence of controls for optimizing pollutant removal efficiencies (e.g., reverse osmosis prior to or after UV oxidation), as well as to determine the post treatment requirements (e.g., pH adjustment, aeration to raise dissolved oxygen levels). Therefore, total costs as well as selection of necessary treatment controls (e.g., mercury precipitation versus aggressive P2 strategies) would be highly facility-specific.

B.10 References

Association of Metropolitan Sewerage Agencies (AMSA). 2002. Mercury Source Control & Pollution Prevention Program Evaluation. Prepared by Larry Walker Associates.

Blum, Justing and Petula Dvorak. 2003. Washington Post. “Makeup Classes Offered at Ballou: St

EIP Associates. 2000. Technical Memorandum - Mercury Amalgam Treatment Technologies for Dental Offices. To: Palo Alto Regional Water Quality Control Plant.

Federal Remediation Technologies Roundtable (FRTR). 2002. Remediation Technologies Screening Matrix and Reference Guide, Version 4.0. Online at http://www.frtr.gov/matrix2/top_page.html.

Hand, D.W., D.L. Perram, and J.C. Crittenden. 1995. Destruction of DBP Precursors with Catalytic Oxidation, American Water Works Association Journal, Vol. 87, No. 6, pp. 84-96.

Massachusetts Office of Technical Assistance (OTA). 1995. Toxics Use Reduction Case Study: Reducing Mercury Discharge at a Testing Laboratory. Case Study No. 35.

Munter, Rein. 2001. Advanced Oxidation Processes - Current Status and Prospects. Proc. Estonian Acad. Sci. Chem. 50(2), 59-80.

Ohio EPA. 1999. Laboratories Providing Contract Analytical Services Using EPA Method 1631 for Mercury. Online at http://www.epa.state.oh.us/dsw/guidance/permit10att3.pdf.

Pollution Probe. 1996. Mercury in the Health Care Sector: The Cost of Alternative Products.

Randall, Patrick. 2004. Aquatech Inc. Personal communication in September with F. Bekdash, SAIC.

Sacramento Regional County Sanitation District (SRCSD). 2003. Exhibit B - Budget: Third Amendment to Agreement with Pacific Ecorisk to Conduct Year 6 Laboratory and Sampling Services for the Sacramento River Watershed Program, Contract No. 80714. Online at: http://www.srcsd.com//pdf/staffreports/081303-EcoRisk80714ExhibitBBudget.pdf.

Second Nature. 2003. Greening Medical Facilities: Facts and Resources. Online at http://www.secondnature.org/pdf/snwritings/factsheets/green_hosp.pdf

Sherbondy, John. 2004. Tigg Corporation. Personal communication in July with L. Petruzzi, SAIC.

Solmetex. 2004. Personal communication in September with F. Bekdash, SAIC.

Tellus Institute. 2000. Healthy Hospitals: Environmental Improvements Through EA.

U.S. EPA. 2000a. Technologies and Costs for Removal of Radionuclide from Potable Water Supplies. Targeting and Analysis Branch, Standards and Rish Management Division, Office of Ground Water and Drinking Water.

U.S. EPA. 2000b. Technologies and Costs for Removal of Arsenic from Drinking Water. Targeting and Analysis Branch, Standards and Rish Management Division, Office of Ground Water and Drinking Water.

U.S. EPA. 2000c. Guidance for Assessing Chemical Contaminant Data for Use in Fish Advisories: Volume 1, Fish Sampling and Analysis. EPA 823-B-00-007.

U.S. EPA. 2003. Technologies and Costs for Control of Microbial Contaminants and Disinfection Byproducts. Office of Ground Water and Drinking Water.