Adopted by TCEQ August 10, 2005

One Total Maximum Daily Load for Polychlorinated Biphenyls (PCBs) in Fish Tissue in Lake Worth

For Segment 0807

Prepared by the:
Field Operations Division, Region 4 and
Chief Engineer's Office, Water Programs, TMDL Section

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TEXAS COMMISSION ON ENVIRONMENTAL QUALITY

Distributed by the
Total Maximum Daily Load Team
Texas Commission on Environmental Quality
MC-203
P.O. Box 13087
Austin, Texas 78711-3087

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Executive Summary

In 2002, Lake Worth was identified as not supporting the fish consumption use following the issuance a fish consumption advisory by the Texas Department of Health in April 2000. The advisory was issued due to elevated concentrations of polychlorinated biphenyls (PCBs) in fish tissue.

The assessment endpoint and ultimate goal of this TMDL is the reduction of fish tissue PCB concentrations to a level that constitutes an acceptable risk to fish consumers, allowing the Texas Department of State Health Services (TDSHS) to remove the consumption advisory. The measurement endpoint of the TMDL is a numeric target (<0.04 mg/kg) that defines the acceptable fish tissue PCB concentration; however, TDSHS is responsible for the final evaluation of fish tissue data with regard to consumption risk. The numeric target is also a surrogate measure of the contaminant load.

Source assessment determined that PCB sources are located within the drainage area to the Woods Inlet portion of Lake Worth, which includes much of Air Force Plant 4 (AFP4), a military aircraft manufacturing facility. The U.S. Air Force has conducted facility investigations and remedial activities at AFP4 under a CERCLA Record of Decision (ROD). PCBs were identified as one of the contaminants at the site.

Studies conducted by the U.S. Geological Survey and private consultants have further isolated AFP4 as the major source of PCBs in the lake. Higher sediment PCB levels were found in areas adjacent to AFP4. PCBs were detected in sediments within the AFP4 storm sewer system and in Meandering Road Creek sediments in an area impacted by two AFP4 storm sewer outfalls. Congener analysis found that PCBs in creek and lake sediments appear to be related to those found in the AFP4 storm sewer system.

Sediment core samples indicate that PCB loading to Lake Worth has declined exponentially since the mid-1960s peak. Available data indicate that fish tissue PCB concentrations are also generally declining.

Because PCBs are already restricted, and sediment and fish tissue concentrations in the lake are in decline, no significant additional loading is expected. Any existing loading from AFP4 will be mitigated through regulatory mechanisms such as the ROD that are currently in place at the site. Continuing natural attenuation of PCBs in lake sediment and fish tissue is also expected.

Introduction

Section 303(d) of the Clean Water Act requires all states to identify waters that do not meet, or are not expected to meet, applicable water quality standards. For each listed water body that does not meet a standard, states must develop a Total Maximum Daily Load (TMDL) for each pollutant that has been identified as contributing to the impairment of water quality in that water body. The Texas Commission on Environmental Quality (TCEQ) is responsible for ensuring that TMDLs are developed for impaired surface waters in Texas.

In simple terms, a TMDL is a quantitative plan that determines the amount of a particular pollutant that a water body can receive and still meet its applicable water quality standards. In other words, TMDLs are the best possible estimates of the assimilative capacity of a water body for the pollutant under consideration. A TMDL is commonly expressed as a load, with units of mass per time period, but it may also be expressed in other ways. TMDLs must also estimate how much the pollutant load needs to be reduced from current levels in order to achieve water quality standards.

The Total Maximum Daily Load Program, a major component of Texas' statewide watershed management approach, addresses impaired or threatened streams, reservoirs, lakes, bays, and estuaries (water bodies) in or bordering the state of Texas. The primary objective of the TMDL Program is to restore and maintain the beneficial uses (such as drinking water, recreation, support of aquatic life, or fishing) of impaired or threatened water bodies.

The ultimate goal of this TMDL is the reduction of polychlorinated biphenyls (PCB) in fish tissue in Lake Worth to a level that constitutes an acceptable risk to fish consumers, allowing the Texas Department of State Health Services (TDSHS) to remove the existing fish consumption advisory on the lake. Reduction of fish tissue PCB concentrations and removal of the advisory will restore the beneficial fish consumption use to Lake Worth.

Section 303(d) of the Clean Water Act and the U.S. Environmental Protection Agency's (USEPA) implementing regulations (40 Code of Federal Regulations, Section 130) describe the statutory and regulatory requirements for acceptable TMDLs. The TCEQ guidance document, *Developing Total Maximum Daily Load Projects in Texas* (TNRCC 1999), further refines the process for Texas. This TMDL document has been prepared in accordance with these guidelines, and is composed of the following main elements:

- Problem Definition, including Designated Uses and Water Quality Standards
- Endpoint Identification
- Source Assessment
- Linkage Between Sources and Receiving WatersMargin of Safety
- Pollutant Load Allocation
- Public Participation
- Implementation and Reasonable Assurance

Document Preparation

This TMDL document was prepared by:

- Region 4 Water Section, Field Operations Division, Office of Compliance and Enforcement, TCEQ, and the
- TMDL Section, Water Programs, Chief Engineer's Office, TCEQ.

Technical assistance was provided by staff from the Texas Department of State Health Services, the State Toxic Substances Coordinating Committee Fish Sampling Advisory Subcommittee, and the TCEQ Region 4 Waste Section.

The TMDL was adopted by the commission on August 10, 2005. Upon EPA approval, the TMDL will become an update to the state Water Quality Management Plan.

Background Information

This TMDL addresses contamination of fish tissue by polychlorinated biphenyls (PCBs) in Lake Worth, a reservoir located in northwest Tarrant County in north-central Texas (Figure 1). As a result of this contamination, Lake Worth is under a fish consumption advisory issued by the Texas Department of Health (TDH) (Note: TDH became part of the Texas Department of State Health Services (TDSHS) on 1 September 2004.)

PCBs are sometimes referred to as legacy pollutants, which is a collective term for substances whose use has been banned or severely restricted by the U.S. Environmental Protection Agency (USEPA). No additional loading of legacy pollutants is allowed or expected due to the USEPA restrictions; however, because of their slow rate of decomposition, these substances frequently remain at elevated levels in the environment for many years after their widespread production and use has ended. Gradual declines in environmental legacy pollutant concentrations occur as a result of natural attenuation processes.

In preparing TMDLs for legacy pollutants, the TCEQ has modified the typical loading allocation approach, which limits the amount of a pollutant that can be added to an impaired water body. Because PCBs are already restricted and no significant additional loading is expected, this TMDL does not specifically attempt to quantify an allowable load of PCBs that may be released to the lake. Within the context of this TMDL, legacy PCBs are considered background sources that reflect the site-specific release history and loss rates of the subject area. Continuing loading may occur from nonpoint source runoff, leaching, or erosion of any sinks that exist within the watershed. In the case of PCBs in Lake Worth, a principal contributing source has been identified and is undergoing mitigation under the federal CERCLA program.

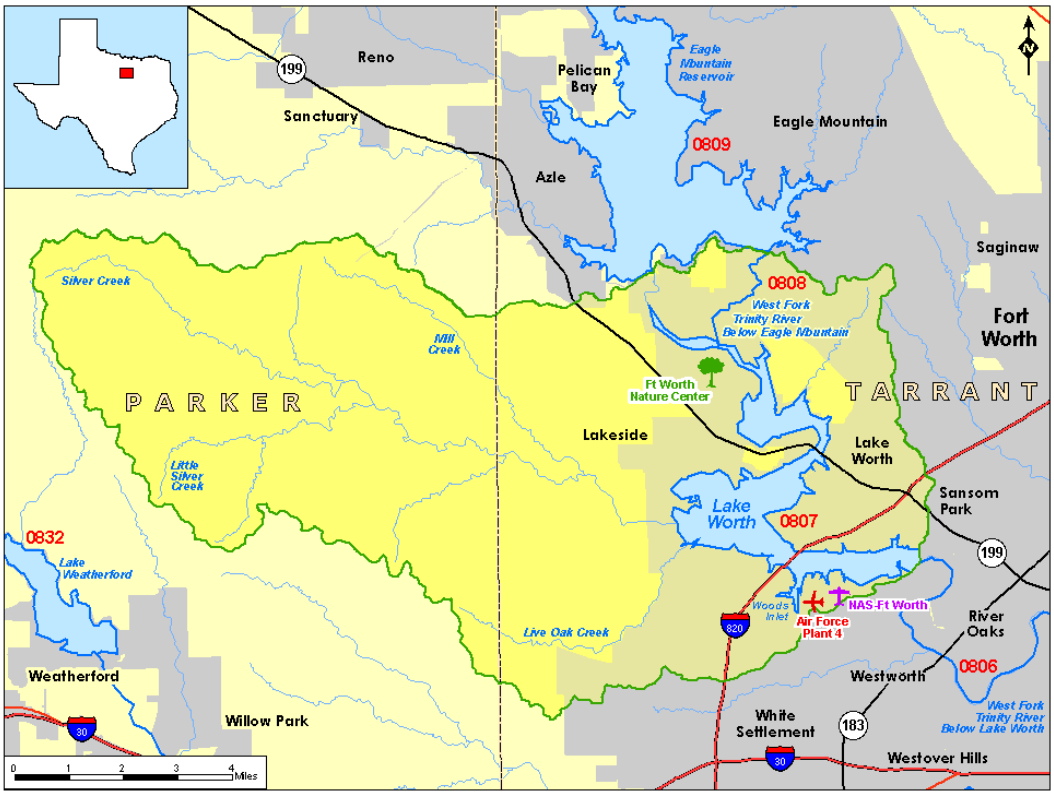

Figure 1. Lake Worth location and watershed.

Calculation of a traditional loading value is difficult and subject to considerable uncertainty due to the tendency of hydrophobic contaminants such as PCBs to partition into sediment and tissue. PCBs may be present in sediment and tissue at concentrations that are orders of magnitude higher than in the water column, where they are typically very low or undetectable (see Smith *et al.* 1988; USEPA 1999). The numeric target, or measurement endpoint (USEPA 1994), for this TMDL is the fish tissue PCB concentration that is considered an acceptable risk to human health. The numeric target is a surrogate measure of the contaminant load that can be translated into an acceptable tissue concentration.

This approach is similar to that used by USEPA (2001) to develop the water quality criterion for methylmercury for the protection of human health, which was established as a concentration in fish tissue. This approach was taken because it better integrates spatial and temporal complexity, it is more closely tied to the goal of protecting human health, it is easier to quantify mercury in fish tissue than in water, and it is consistent with how fish consumption advisories are issued (USEPA 2001). The same rationale applies to fish tissue contamination by legacy pollutants such as PCBs. Use of tissue contaminant concentration as a surrogate measure of contaminant load has been successful in the development of other

legacy pollutant TMDLs (see TNRCC 2000ab, 2001ab, 2003b; CRWQCB 2004). The ultimate goal, or the assessment endpoint (USEPA 1994), of this TMDL is the reduction of fish tissue PCB concentrations in Lake Worth to levels that constitute an acceptable risk to consumers, allowing TDSHS to remove the fish consumption advisory.

Problem Definition

Lake Worth was added to the State of Texas 2002 §303(d) list as a result of the issuance a fish consumption advisory by the Texas Department of Health. Fish and Shellfish Consumption Advisory No. ADV-18 was issued on 19 April 2000, advising persons to not consume fish from Lake Worth until further notice (TDH 2000a). The advisory was issued following a determination of unacceptable human health risk due to elevated concentrations of PCBs in the tissue of fish collected from the lake (TDH 2000b).

Designated Uses and Water Quality Standards

Lake Worth is a classified Water Quality Segment of the State of Texas with designated uses of contact recreation, high aquatic life, and public drinking water supply. Texas Water Quality Standards (Title 30 Texas Administrative Code Chapter 307) presume all designated segments to have sustainable fisheries, defined as ". . . sufficient fish production or fishing activity to create significant long-term human consumption of fish" (30 TAC 307.3(a)(58)). The fish consumption use of a water body is not supported when TDSHS has issued a consumption advisory, and this designated use is thus impaired in Lake Worth. Other designated uses are not affected.

State antidegradation policy states that "existing uses and water quality sufficient to protect those existing uses will be maintained" (30 TAC 307.5(b)(1)). Additional guidance on the TCEQ antidegradation policy can be found in the document entitled *Procedures to Implement the Texas Surface Water Quality Standards* (TNRCC 2003a), which describes TCEQ procedures for ensuring the protection and maintenance of existing designated uses in water bodies in the state.

Lake Worth Watershed

Lake Worth (Segment 0807) is a 3558-acre impoundment of the West Fork Trinity River located in northwest Tarrant County (NCTCOG 1999; see Figure 1). The lake was impounded in 1914 as a drinking water supply for the City of Fort Worth, the owner and operator of the lake. The lake has a conservation capacity of approximately 37,000 acre-feet, and drains a 2064 square mile watershed (Dowell and Breeding 1967; Ulery *et al.* 1993). The main lake tributary is Silver Creek, which drains northwest Tarrant County and east-central Parker County to the west of the lake (Figure 1). Live Oak Creek drains a smaller watershed southwest of the lake. Meandering Road Creek and two smaller tributaries drain a small watershed south of the lake, with those flows entering a large cove known as Woods Inlet (Figures 1 and 2).

The watershed area routinely contributing flow to Lake Worth is significantly reduced by the presence of Eagle Mountain Lake, which was constructed several miles upstream on the West Fork Trinity River in 1932. The Lake Worth watershed was almost entirely rural in 1932, and the lake received predominately rural and agricultural runoff between its 1914 impoundment and 1932, with an accompanying rapid buildup of sediment in the lake (NCTCOG 1999).

Figure 2. Aerial photo detail of Lake Worth

Approximately 94 square miles of the Lake Worth watershed are located downstream from the dam on Eagle Mountain Lake. This area has contributed most of the direct runoff to the lake since 1932, with additional flow contributed by releases from Eagle Mountain Lake. This portion of the Lake Worth watershed is approximately 35 percent urban and 60 percent rural (evaluation based on maps and data in NCTCOG 1999).

The upstream end of Lake Worth is bordered by the Fort Worth Nature Center and Refuge (Figure 1). South of the Nature Center, the lake is bordered on the west by residential housing and commercial activities adjacent to State Highway 199 (Jacksboro Highway), the

Town of Lakeside, and parkland. On the east and north, the lake is bordered by a portion of the City of Lake Worth, parkland, and lakeside residential housing within the City of Fort Worth (see Figure 2). Lakeside residential housing within Fort Worth also borders much of the south side of the lake. A residential section of the City of White Settlement is located south of the lake and to the east of IH-820.

Two large industrial facilities are located adjacent to the south side of the lake (see Figures 1 and 2):

- The U.S. Naval Air Station Joint Reserve Base – Fort Worth (NASFW) is located on 2264 acres near the southeast corner of the lake. The site was originally a dirt runway built to service an adjacent aircraft manufacturing plant. It was established as the Tarrant Field Airdrome in 1942. The Strategic Air Command assumed control in 1946, and the facility was renamed Carswell Air Force Base in 1948. The base was selected for closure and property disposal under the Base Realignment and Closure Act of 1990. Drawdown activities began in1992, and all aircraft were relocated by January 1993. The base officially closed on 30 September 1993. The U.S. Navy assumed control on 1 October 1994, and the base was given its current designation as NASFW (Jacobs Engineering Group 1998). Runoff or waste discharges from the northern end of the base enter Lake Worth. The eastern portion of the base discharges to the West Fork Trinity River below Lake Worth (Segment 0806). The south end of the base discharges to Farmers Branch, a tributary to Segment 0806, or to Kings Branch (tributary to Farmers Branch). Storm water discharges from the base are regulated under General Permit No. TXR050000 (TCEQ Multi-Sector Industrial General Permit for Storm Water).

- Air Force Plant No. 4 (AFP4) is a government owned, contractor-operated facility for the manufacture of military aircraft. The facility is located on 760 acres on the west side of NASFW (USEPA 2003a), and on the east side of Meandering Road Creek and the Woods Inlet cove. Approximately 70 percent of the AFP4 site is covered by buildings, concrete, or asphalt (TDH 1998). Consolidated Vultee Aircraft Corporation built the original plant in 1942. General Dynamics Corporation began operating AFP4 in 1954. The plant has been operated under contract by the Lockheed Martin Corporation since 1993. Runoff or waste discharges from much of the site enter Meandering Road Creek and Woods Inlet along the west side of the facility. TCEQ Permit 01764, issued to the U.S. Air Force and the contractor, authorized the discharge of storm water and noncontact cooling water from three outfalls on the west side and one outfall on the north side of the site. A fifth outfall discharges to Farmers Branch from the southeast corner of the site. The permit was canceled in April 2002 following elimination of the cooling water discharges. Storm water discharges from the site are currently regulated under General Permit No. TXR050000 (TCEQ Multi-Sector Industrial General Permit for Storm Water).

The U.S. Air Force has conducted RCRA facility investigations at NASFW and AFP4 to determine if hazardous constituents have been released from any identified waste

management or disposal sites at either facility. The primary contaminants of concern at NASFW are volatile organic compounds, primarily trichloroethylene and its degradation products (HydroGeoLogic, Inc. 2001). PCB contamination does not appear to be a major issue at NASFW.

Manufacturing operations and associated processes at AFP4 have generated waste oils, waste fuels, paint residues, used solvents, and process chemicals (USEPA 2003a). PCBs were specifically identified as a contaminant at AFP4 (see *Installation Narrative Summaries, Air Force Plant No. 4, Defense Environmental Restoration Program Annual Reports to Congress*, <www.dtic.mil/envirodod/DERP/DERP.htm>). A preliminary 1994 study detected the presence of PCBs in mosquitofish tissue in the vicinity of AFP4 (see TDH 1998). A more comprehensive fish tissue study (Moring 2002) was subsequently conducted by the U.S. Geological Survey (USGS). Evaluation of fish tissue data collected during that study led to issuance of the fish consumption advisory. Sediment sampling conducted as part of the facility investigations detected higher PCB concentrations in the Woods Inlet portion of the lake and in Meandering Road Creek, in areas adjacent to AFP4, relative to other areas of Lake Worth (Harwell *et al.* 2003; Besse *et al.* 2005). Additional investigation detected PCBs in sediments within the AFP4 storm sewer system and in Meandering Road Creek sediments in a reach impacted by two storm sewer outfalls (Schultz *et al*. 2005).

The Texas National Guard operates a maintenance facility at the south end of Woods Inlet. The site drains to a small, unnamed tributary to the inlet, located north and west of the Meandering Road Creek drainage area. A residential area in White Settlement is also within the watershed of this tributary, upstream from the National Guard facility. Evaluation of PCB concentrations in the storm flow and bed sediment of this tributary, conducted as part of the NASFW and AFP4 site investigations, found no evidence of a major PCB source within this watershed (Besse *et al.* 2005).

The North Central Texas Council of Governments Closed and Abandoned Municipal Solid Waste Landfill Inventory (<www.nctcog.org/envir/sw/ACT/CLI/index.html>) lists eight unauthorized and three permitted landfill sites in the Lake Worth watershed, all of which have been closed (see Table 1). All of these sites are located west of the lake except Site U1354, which is located on the east side along the northern arm.

Most of the unauthorized landfills accepted items such as household wastes, appliances, and construction debris. The exception was Site U1342 (White Settlement Landfill) in Tarrant County, near the intersection of Cattlebaron Drive and FM Road 1886. This site accepted a variety of wastes from eight cities and an unknown number of industries. The 55-acre site began accepting waste in 1950. A permit was apparently issued in about 1975 by the Texas Department of Health, the permitting authority at that time. The site was closed in June 1978. The Landfill Inventory refers to Type I waste, mixed municipal waste, tires, drums, toxins, and industrial waste having been dumped at the site. PCBs are not specifically mentioned, but their disposal at the site cannot be definitively excluded. Runoff from this site enters Silver Creek, approximately 4.5 stream miles from the lake.

Table 1. Closed and abandoned municipal solid waste sites (Top – unauthorized; Bottom – permitted) within the Lake Worth watershed. na = not available. Source: NCTCOG Closed and Abandoned Municipal Solid Waste Landfill Inventory, <www.nctcog.org/envir/sw/ACT/CLI/index.html>.

Site ID#	Site Name (Location)	Latitude	Longitude	Date Closed
U829	Bob Lawrence (Tarrant County)	na	na	Unknown
U846	Porter Site (1204 Silver Creek Road, Tarrant County)	32° 46.49'	97° 29.25'	1977
U1342	White Settlement Landfill (Tarrant County)	32° 49.5'	97° 32.5'	1978*
U1354	Bo Lawrence (4931 Hodgkins Road, Tarrant County)	32° 50'	97° 26.3'	Unknown
U2352	Raymond Salter White (Parker County)	32° 49.08'	97° 35.98'	1987
U2357	William Demetri (Hilltop MHP, Parker County)	na	na	1994
U2383	Thomas Thompson (Parker County)	32° 47.05'	97° 33.44'	1987
U2414	Charles Massey (8102 Jacksboro Highway, Tarrant County)	32° 56.09'	97° 31.92'	1991
Permit#	**Site Name (Location)**	**Latitude**	**Longitude**	**Date Closed**
700	City of White Settlement (Tarrant County)	32° 49.5'	97° 32.5'	1978*
1156	Azle Regional Landfill (Tarrant County)	32° 49.45'	97° 32.85'	1991
1748	V.M. Crow & Sons, Inc. (Tarrant County)	29° 32.9'	95° 6.63'	unknown

*Site began accepting waste in 1950 and was permitted by the Texas Department of Health in approximately 1975. See text for additional information.

None of these landfills are known to have been sources of PCB releases, and any such releases should have ceased with closure of the sites. Lake sediment analyses (Harwell *et al.* 2003) conducted as part of the NASFW and AFP4 site investigations did not find elevated PCB concentrations in areas where drainage from these landfills enters the lake.

Four TCEQ-authorized facilities are located west of the lake:

- The Town of Lakeside operates a 30,000-gallons-per-day municipal wastewater treatment plant (Permit No. 11573-001) located off the south end of Aquilla Drive, approximately 0.8 mile from the lake. Disposal of treated effluent is by irrigation on a 16-acre site immediately northwest of the plant. Any discharge from the plant or irrigation field will be to a small unnamed tributary to Lake Worth.

- The R.J. Smelley Company operates a 1500-head dairy (Permit No. 02422) at 4750 Cattlebaron Drive, just inside the Parker County line, approximately 3.5 miles from the lake. Any runoff or waste discharge from the facility will enter an unnamed tributary to Silver Creek, approximately 5.5 stream miles from the lake. There are two small reservoirs on the tributary that act as sediment traps.

- Abraxas, Inc. operates a 15,000-gallons-per-day municipal wastewater treatment plant (Permit No. 11086-001) located on Cattlebaron Drive in Parker County, just south of the R.J. Smelley Dairy and 0.9 mile north of the intersection of Cattlebaron Drive and White Settlement Road. The plant serves the Hilltop Park Addition subdivision. Disposal of treated effluent is by irrigation on a 10-acre site adjacent to the plant. Abraxas has submitted an application to amend the permit to authorize a flow of 30,000 gallons per day, with disposal by irrigation on an adjacent 20-acre site. Any discharge from the plant or irrigation field will enter the same unnamed tributary to Silver Creek as any discharge from the dairy.

- Silver Creek Materials Recovery Center operates a composting facility (MSW No. 42008) located north of Silver Creek and west of Silver Creek Road in Tarrant County, approximately 0.6 mile from the lake. Any discharge from the site will be directly to Silver Creek.

These four facilities are not expected to be sources of PCBs. No waste discharge is authorized from any of these sites. In addition, lake sediment analyses (Harwell *et al.* 2003) conducted as part of the NASFW and AFP4 site investigations did not find detectable PCB concentrations in the middle portion of Lake Worth where Silver Creek and the small tributary from Lakeside enter the lake.

Tissue Contamination and Risk Assessment

PCBs were manufactured and widely used in the United States prior to USEPA restriction (Erickson 2001). These restrictions did not require PCB-containing materials to be removed from service, and many are still in use (USEPA 1999). PCBs are common environmental contaminants (Moore and Ramamoorthy 1984; Schmitt *et al.* 1985, 1990; Smith *et al.* 1988; Tanabe 1988; Kuehl *et al.* 1994; Mora 1995; Knight and Powell 2001; USEPA 2003b), and are frequently found at elevated levels in the tissue of aquatic organisms (Eisler 1986; Evans *et al.* 1991). USEPA (1992) found PCB residues in fish tissue at 91 percent of 388 nationwide locations in 1986-87. PCBs have been detected in fish tissue and sediment in a number of water bodies in Texas (Dick 1982; Van Metre and Callender 1996; Webster *et al.* 1998), including the Dallas/Fort Worth area (Irwin 1988; Moring 1997; Van Metre and Callender 1997). Because of their low solubility, PCBs are not acutely toxic to aquatic life, but instead cause sublethal and chronic effects (Eisler 1986; McFarland and Clarke 1989; Eisler and Belisle 1996; Colborn and Thayer 2000; Gundersen *et al.* 2000; Khan 2003).

PCBs are a frequent cause of fish consumption advisories in the U.S. (USEPA 1999). Elevated concentrations are frequently found in game fish tissue (Kuehl *et al.* 1994; Johnston *et al*. 2002). Fish consumption can be a primary route of human exposure to PCBs, with

significantly elevated blood serum and milk contaminant levels found in those consuming contaminated fish (Schwartz *et al.* 1983; Humphrey 1987; Fiore *et al.* 1989; Asplund *et al.* 1994; USEPA 1999; Hanrahan *et al.* 1999; Stewart *et al.* 1999; He *et al.* 2001).

PCBs can cause a variety of adverse health effects and are classified as probable human carcinogens (Swain 1988; Safe 1994; Longnecker *et al.* 1997; USEPA 1996b; Johnson *et al.* 1999; ATSDR 2000; Colborn and Thayer 2000; Maczka *et al.* 2000; Longnecker 2001; Schantz *et al.* 2001). Cogliano (1998) reviewed available data and found a strong case that all PCB mixtures can cause cancer. Different mixtures have different potencies as a result of environmental processes that alter these mixtures through partitioning, chemical transformation, and preferential bioaccumulation.

The consumption advisory for Lake Worth was issued on the basis of an unacceptable carcinogenic risk of liver cancer and a noncarcinogenic risk of possible adverse liver effects due to fish tissue PCB concentrations. The health assessment (TDH 2000b) evaluated risk to a 70-kg adult consuming an average of 30 grams of contaminated fish per day, and to a 15- to 35-kg child consuming an average of 15 grams of contaminated fish per day, both for an exposure period of 30 years.

Endpoint Identification

The assessment endpoint (USEPA 1994) and ultimate goalof this TMDL is the reduction of fish tissue PCB concentrations to levels that constitute an acceptable risk to fish consumers, allowing TDSHS to remove the fish consumption advisory. The measurement endpoint (USEPA 1994) is a numeric target that defines the fish tissue PCB concentration that is considered an acceptable risk to human health. This numeric target is a surrogate measure of the contaminant load that will translate into an acceptable tissue concentration. USEPA (2001) recently developed a water quality criterion for methylmercury for the protection of human health as a concentration in fish tissue. This approach was taken because it better integrates spatial and temporal complexity, it is more closely tied to the goal of protecting human health, it is easier to quantify mercury in fish tissue than in water, and it is consistent with how fish consumption advisories are issued. The same rationale applies to fish tissue PCB concentrations.

USEPA (2000b) provides guidance for assessing tissue contaminant data for risk assessment. This guidance and TDSHS assumptions (TDH 2000b) were used to develop numeric targets for tissue PCB concentrations that result in an acceptable risk level. USEPA (2000b) presents equations for calculating the maximum allowable fish consumption rate given consumer body weight, contaminant concentration, an acceptable cancer risk level, and the contaminant risk values for carcinogenic and noncarcinogenic risk. A cancer potency value (q_1*) is the risk value for carcinogens. The oral reference dose (RfD) is the risk value used to protect against chronic exposure by noncarcinogens.

The consumption rate and consumer body weight were set at the TDSHS constants of 30 grams of fish per day (0.03 kg/d) for a 70-kg adult, and at 15 grams of fish per day (0.015

kg/d) for a 15- to 35-kg child, both over a 30-year time period. TDSHS uses an acceptable cancer risk level of 1×10^{-4}, adjusted to 2.33×10^{-4} to account for the use of the 30-year time period. Cancer potency and chronic RfD values for PCBs were obtained from the USEPA Integrated Risk Information System (IRIS; see <www.epa.gov/iris/>). The USEPA carcinogenic (q_1*) risk value is the upper-bound slope factor for PCB mixtures. The noncarcinogenic (RfD) risk value used by USEPA is based on the Aroclor 1254 mixture.

Equations in USEPA (2000b) were solved to calculate the maximum allowable concentration of a contaminant in tissue (1) at a given cancer risk level, and (2) based on the noncarcinogenic health effects of a contaminant:

(1) $$C_m = (ARL)(BW) / (C_{lim})(q_1^*)$$

(2) $$C_m = (RfD)(BW) / (C_{lim})$$

where:

C_m = maximum allowable concentration of a contaminant in tissue (mg/kg)
ARL = acceptable cancer risk level = 2.33×10^{-4}
BW = consumer body weight (kg)
C_{lim} = allowable fishes a consumption rate (kg/d)
q_1* = cancers slope factor for given contaminant = 2.0 $(mg/kg/d)^{-1}$ for PCBs
RfD = oral reference dose for given contaminant = 2×10^{-5} mg/kg/d for PCBs.

Substituting the consumption rates and body weights used by TDSHS, the maximum tissue PCB concentrations that can be consumed with an acceptable level of risk were calculated. The target concentrations that will achieve an acceptable noncarcinogenic risk are less than those needed for an acceptable carcinogenic risk (see Table 2). The target concentrations for a 15-kg child are less than those of a 35-kg child and a 70-kg adult. The calculated noncarcinogenic value for a 15-kg child (0.02 mg/kg) is the most protective numeric endpoint; however, this value is less than the reporting limit accepted by TDSHS for the assessment of PCBs in fish tissue (0.04 mg/kg). TDSHS has determined that an ongoing ability to recover PCBs in fish tissue at a reporting limit of 0.04 mg/kg can be demonstrated through the agency's laboratory QA/QC procedures. TDSHS sampling and analysis protocols are based on USEPA (2000a) guidance. Other reporting limits in the literature reflect differences in the ability to measure individual PCB congeners as opposed to Aroclor mixtures, as well as different laboratory capabilities. Reporting limits for single congeners are generally much lower than those of Aroclor mixtures; however, cancer potency factors do not yet exist for individual congeners and health risk assessments are based on Aroclor mixture concentrations. The numeric measurement endpoint for this case is therefore set as less than 0.04 mg/kg.

These measurement endpoints (Table 2) are valid only under the assumed conditions. TDSHS has the authority and jurisdiction for the final evaluation of all fish tissue data with regard to consumption risk, and for any decision to issue, alter, or remove fish consumption advisories. Subsequent risk assessments by TDSHS may result in no change to the advisory,

removal of the advisory, or alteration of the advisory to specify certain fish species and/or certain groups of consumers who are at greater risk. The ultimate goal, or the assessment endpoint for determining the success of this TMDL is the complete removal of the consumption advisory. The measurement endpoint can be used to monitor progress toward meeting the assessment endpoint goal.

Table 2. Maximum fish tissue PCB concentration (mg/kg) that can be ingested by consumers of given body weight, within the acceptable cancer risk level (ARL) used by TDSHS, and without causing adverse noncarcinogenic health effects. Carcinogenic (q_1*) and noncarcinogenic (RfD) risk values were obtained from the USEPA IRIS database (<www.epa.gov/iris/>; see text for details).

	Maximum Fish Tissue PCB Concentration (mg/kg)	
Consumer Body Weight	**Carcinogenic Risk q_1* = 2.0 (mg/kg/d)$^{-1}$ ARL = 2.33 x 10^{-4}**	**Noncarcinogenic Risk RfD = 2 x 10^{-5} mg/kg/d**
15-kg (Child)	0.12	<0.04**
35-kg (Child)	0.27	0.05
70-kg (Adult)	0.27	0.05

**The noncarcinogenic fish tissue concentration calculated for the 15-kg child receptor category (0.02 mg/kg) is less than the current TDSHS reporting limit for PCBs in fish tissue (0.04 mg/kg). The concentration target is therefore set at the reporting limit.

Source Assessment

PCBs are a group of synthetic organic chemicals containing 209 possible individual congeners, which vary in chemical and physical properties, toxicity, environmental persistence, and degree of bioaccumulation (USEPA 1999; Erickson 2001). PCBs were manufactured beginning in1929 as mixtures of different congeners, and generally sold under the trade name Aroclor. PCBs were used in a wide variety of applications, including coolants and lubricants in transformers, capacitors, and other electrical equipment, heat transfer fluids, hydraulic fluids, lubricating and cutting oils, and as additives in pesticides, paints, copying paper, carbonless copy paper, adhesives, sealants, and plastics (Erickson 2001). The dominant use was in capacitors and transformers.

In 1976 the Toxic Substances Control Act (TSCA) banned, with limited exceptions, the manufacture, processing, distribution in commerce, and use of PCBs (Erickson 2001). TSCA also required the USEPA to promulgate regulations for proper use, cleanup, and disposal of PCBs. TSCA and subsequent USEPA rules (see Erickson 2001 for review) did not require PCB-containing materials to be removed from service, and many are still in use (USEPA 1999). A substantial portion of the PCBs manufactured prior to 1977 remain in use, although these are being phased out as equipment is replaced or decontaminated.

Because of their past heavy and widespread use, strong affinities for sorption to sediment organic matter and tissue, and slow rates of decomposition, PCBs frequently remain at elevated levels in the environment for many years after widespread use has ended (Moore and Ramamoorthy 1984; Smith *et al.* 1988; Jones and de Voogt 1999). PCBs can enter the environment via direct and indirect sources such as industrial and municipal waste discharges, spills and leaks, transformer fires, improper disposal methods, and leaching from landfills (Tanabe 1988; Eisler and Belisle 1996). The current major source of PCBs is probably environmental reservoirs caused by past releases (USEPA 1999). Past releases from production and storage facilities have resulted in many of the more heavily contaminated sites (see Smith *et al.* 1988; Renner 1998; Bergen *et al.* 1998; Bremle and Larsson 1998; O'Meara *et al.* 2000; Steuer 2000; USEPA 2002). Lee and Anderson (1998) found that the spatial distribution of PCB concentrations in carp and walleye fillets in the Upper Mississippi River Basin corresponded with historical and current point and nonpoint source PCB inputs. The largest PCB concentrations in Lake Hartwell, South Carolina sediments were those found closer to a point source (Dunnivant *et al*. 1989). Contaminant levels in white sturgeon tissue in the Fraser River in British Columbia reflected proximity to known contaminant sources (MacDonald *et al.* 1997). Sethajinatin *et al.* (2004) found fish tissue PCBs to increase with decreasing distance from a Superfund site.

Transport of PCBs within the environment occurs via volatilization, atmospheric transport and precipitation, runoff, and sediment disturbance (Eisler and Belisle 1996; Wania and Mackay 1996; Erickson 2001). Atmospheric deposition can be a major source of PCB input to water bodies (Stapleton *et al.* 2001b; Dinkins *et al.* 2002; Wethington and Hornbuckle 2005). In other cases, volatilization to the atmosphere can be a major mechanism for the removal of PCBs from water bodies (Swackhamer and Armstrong 1986; Jeremiason *et al.* 1994; Gevao *et al*. 2000; Bamford *et al.* 2002).

A primary method of transport of PCBs and other legacy pollutants into aquatic systems is by erosion of soil and attached contaminants (Munn and Gruber 1997; Van Metre *et al.* 2003b). Sediment deposition is a dominant process in reservoirs (Thornton 1990), and thus can be a major cause of legacy pollutant loss from the water column to the sediments. Aquatic sediments act as a reservoir for PCBs and other hydrophobic contaminants (Moore and Ramamoorthy 1984; Chen *et al.* 2000). PCBs may be present in sediment at concentrations that are orders of magnitude higher than in the water column, where they are typically very low or undetectable (see Smith *et al.* 1988).

Sediments may act as long-term sources of contamination through desorption of contaminants, and as a result of the resuspension of sediment particles by disturbances (Oliver *et al*. 1989; Baker *et al*. 1991; Gevao *et al.* 1997; Maher *et al*. 1999; Davis 2004; Apitz *et al.* 2005). Contaminated sediment can be a major PCB source, with biota and water column concentrations controlled by the underlying sediment levels (Eisler and Belisle 1996; Morrison *et al.* 2002). Lee and Anderson (1998) found that PCB concentrations in carp and walleye fillets in the Upper Mississippi River Basin were greater in areas that historically had elevated bed sediment PCB concentrations.

Larsson *et al.* (1990) determined that sediment from a contaminated lake acted as a source of PCBs entering the associated river system. Steuer *et al.* (1999a,b) found that episodic increases in suspended particle-associated PCB concentrations were associated with bed sediment resuspension during storms. Widespread low-level contamination found during Ohio River TMDL studies suggested that resuspension of sediment during periods of high flow may contribute significant PCB loads to the water column (Dinkins *et al.* 2002). Water pumped through a transfer canal in south Texas is thought to have kept PCB-contaminated canal sediment in suspension, making the exposure pathway to fish tissue more direct (Mahler and Van Metre 2003). Significant resuspension of sediment and associated contaminants may occur at fall turnover in stratified lakes (Evans 1994; Larsson *et al*. 1998). Secondary sedimentation caused by wind-induced resuspension is an important process in large, shallow lakes (Douglas and Rippey 2000). Luettich *et al*. (1990) suggest that lakes may have a thin layer of loosely bound sediment that is continuously involved in resuspension. Sediment disturbance by organisms that live or feed in contact with the sediment can also provide a route for the reintroduction of contaminants into the food web (Larsson 1986; Reynoldson 1987; Klump *et al.* 1991; Lester and McIntosh 1994; Zhou and Wong 2000; Khan 2003; Thibodeaux and Bierman 2003).

Land Use and PCB Sources

Smith *et al.* (1988) concluded that the largest environmental PCB concentrations are typically found near industrial areas. Stamer *et al.* (1985) observed increasing PCB levels to be related to an increasing percentage of urban land. Stackelberg (1997) found the greatest frequencies of sediment PCB detection to be in the most highly-developed areas. Black *et al.* (2000) found the largest number of total PCBs detected in both fish tissue and sediment to be in watersheds dominated by urban land use. The greatest concentrations of PCBs in fish tissue were also found at urban sites, as were most of the highest sediment concentrations. Munn and Gruber (1997) detected total PCBs in sediment and fish tissue only at urban land use sites. Brasher and Wolff (2004) detected PCBs only at urban and mixed land use sites. Higher PCB concentrations at mixed land use sites can make determination of a specific source more difficult (Tate and Heiny 1996). Blanchard *et al.* (1999) found higher fish tissue PCB concentrations downstream from an urban area. Rinella *et al.* (1993) found total PCBs in fish tissue only in samples collected downstream from an urban area. Schmitt (2002) found fish tissue PCB residues in mostly industrial areas of the Mississippi River basin. King *et al.* (2004) found that the percentage of developed land in a watershed explained 93 to 99 percent of the variance in fish tissue PCB concentrations.

The USEPA (1992) National Study of Chemical Residues in Fish examined 388 locations nationwide in 1986-87. The largest tissue PCB concentrations were associated with industrial sites, industrial/urban areas, and Superfund sites, while the median concentration at 20 background sites was less than the detection limit. In an evaluation of 500 sites studied under the USGS National Water-Quality Assessment Program, Wong *et al.* (2000) found that PCBs were detected in every urban fish sample, and had a greater detection frequency in sediment at urban sites than at other land use sites. PCBs were also elevated in fish tissue at integrator sites located downstream from large urban areas, including the Trinity River below the Dallas/Fort Worth area.

Extensive sampling of storm water outfalls in the seven Phase I storm water cities in the Dallas/Fort Worth area in 1992-1993 (Baldys *et al.* 1998) detected small amounts (0.1-0.8 µg/L) of PCBs in storm water outfalls draining industrial land uses. No PCBs were detected in drainage from residential and commercial land uses. Irwin (1988) found PCB concentrations in whole-body mosquitofish samples from the Trinity River to be strongly associated with industrial runoff from the Dallas/Fort Worth area. Moring (1997) detected PCBs in sediment and tissue most frequently at urban sites within the Trinity River Basin.

PCBs are a factor in fish consumption bans and advisories on three small urban lakes within the City of Fort Worth, and on Mountain Creek Lake in Dallas County (see *Texas Department of State Health Services, Seafood Safety Division, Listing of Water Bodies with Advisories or Bans*, at <www.tdh.state.tx.us/bfds/ssd/survey.html>). In a study of pollutant loading to the urban lakes and selected streams in Fort Worth, Van Metre *et al.* (2003b) found that percent urban land use and percent industrial land use correlated strongly with PCB concentrations in suspended sediment and in recently deposited lake bottom sediment. A naval weapons manufacturing plant and an adjacent U.S. Naval Air Station were found to be significant contributors of PCBs to Mountain Creek Lake (Van Metre *et al.* 2003a).

Lake Worth

The U.S. Air Force has conducted investigations at NASFW and AFP4 to determine if hazardous constituents have been released from any identified waste management or disposal sites at either location. PCB contamination does not appear to be a major issue at NASFW; however, PCBs were specifically identified as a contaminant at AFP4 (see *Installation Narrative Summaries* for *Air Force Plant No. 4* and *Carswell Air Force Base, Defense Environmental Restoration Program Annual Reports to Congress* at <www.dtic.mil/envirodod/DERP/DERP.htm>). AFP4 was added to the USEPA Superfund (CERCLA) National Priorities List in August 1990. A Record of Decision (ROD) was issued in August 1996 (USEPA 1996a). PCBs released through the AFP4 storm sewer system, the contamination of fish tissue, and the risk to fish consumers are not explicitly addressed in the ROD; however, PCBs (as Aroclor 1254) in Meandering Road Creek sediments were determined to have the potential to cause excess ecological risk. The ROD established a remediation goal of 0.1 mg/kg for Aroclor 1254 in sediment based on ecological risk to fish in Lake Worth, and stated that monitoring will continue for as long as contamination remains in the sediment in Meandering Road Creek or until the Air Force, USEPA, and TCEQ agree that monitoring is no longer required (see Section 7.0 in USEPA 1996a). Remediation of PCBs and other contaminants at NASFW and AFP4 is also being addressed through the U.S. Air Force Installation Restoration Program and the TCEQ Defense and State Memorandum of Agreement Program.

The USGS conducted a study (Harwell *et al.* 2003) for the U.S. Air Force to describe the spatial distribution and historical trends of contaminants (including PCBs) in Lake Worth sediment, and to make some determination of major sources of contaminants to the lake. PCBs were not detected in any of the upper-lake surface or core sediment samples, suggesting that there has been little or no PCB input from tributaries in the upper end of the lake, including Silver Creek and Live Oak Creek (see Figure 1 in Harwell *et al.* 2003 for

sediment sample locations). The sample results suggest the authorized facilities and the closed landfills in those watersheds have not contributed PCBs to Lake Worth.

The greatest surface sediment PCB concentrations in the middle and lower portions of the lake were in Woods Inlet, which borders the west side of AFP4 (see Figure 2). PCBs in the remainder of the middle and lower-lake surface sediment samples were near or less than the detection limit. Core samples collected in Woods Inlet and near the Lake Worth dam showed an exponential decrease in PCB concentrations in the more recently-deposited sediment compared with the deeper deposits from the 1960s when PCB production and use were at a peak; however, the peak concentrations in the Woods Inlet core were approximately three times higher than those in the core near the dam.

Harwell *et al.* (2003) concluded that the spatial and temporal distributions of PCBs indicated the most likely source of PCBs to be within the drainage area to Woods Inlet. This drainage area includes much of AFP4, as well as the Texas National Guard maintenance facility and residential areas within the cities of Fort Worth and White Settlement. Trends in the Woods Inlet core suggested that PCBs might still be entering Woods Inlet attached to suspended sediment in storm runoff.

A follow-up USGS study (Besse *et al.* 2005) was conducted to map the distribution of PCBs in bottom sediment in Woods Inlet, and to identify possible source areas of PCBs in the Woods Inlet watershed. Total PCB concentrations in surface sediments were highest near the mouth of Meandering Road Creek, the largest tributary to Woods Inlet, and decreased across Woods Inlet toward the main body of the lake. Samples near the mouths of the two smaller tributaries were lower than at Meandering Road Creek and similar to concentrations in the middle of the inlet (Besse *et al.* 2005). Statistical cluster analysis of the PCB congener profiles in surface and bed sediment samples found several distinct clusters of congener signatures, including one formed by samples from the Meandering Road Creek arm and the middle of Woods Inlet.

Besse *et al.* (2005) concluded that the spatial distribution of surface sediment PCB concentrations indicates a source area within the Meandering Road Creek watershed, that PCBs in Woods Inlet enter primarily from Meandering Road Creek, and that runoff from AFP4 is a major source of the PCBs in the creek. Suspended sediment and streambed sediment PCB concentrations and distributions reflected AFP4 runoff contributions. PCB levels were highest at sites receiving runoff from only AFP4, intermediate at sites receiving runoff from both AFP4 and upstream locations, and lowest at sites where there was no AFP4 runoff contribution. Besse *et al.* (2005) also detected evidence of the current transport of PCBs into Meandering Road Creek from an outfall at AFP4; however, there were large decreases in concentration with decreasing age in the sediment core nearest the outfall, indicating that loading has decreased greatly since the 1960s.

An additional investigation (Schultz *et al*. 2005) was conducted to determine if a source could be identified for the PCBs discharged through the AFP4 outfall identified by Besse *et al*. (2005) (Outfall 4), if other parts of the storm sewer system might be pathways for PCB-

contaminated sediments, whether PCBs were infiltrating the storm sewer system from areas containing dense non-aqueous phase liquids (DNAPL) associated with old AFP4 landfills, and to identify possible PCB sources not associated with AFP4. PCBs were detected in all but one sample associated with the Outfall 4 system, and in all three samples associated with a second outfall (SSO) located upstream along Meandering Road Creek. PCBs were detected in all three Meandering Road Creek sediment samples collected downstream from the SSO outfall, but in none of the creek samples upstream from this location, indicating there are no major sources upstream from AFP4. PCBs were also associated with the DNAPL samples; however, analysis of congener patterns indicates that the DNAPL is not a significant component of the PCBs in Meandering Road Creek or Lake Worth. PCBs in Meandering Road Creek and Lake Worth appear to be related to those detected within the Outfall 4 storm sewer system and at a drain in the SSO system at AFP4 (Schultz *et al*. 2005).

Linkage Between Sources and Receiving Waters

The time required for the reduction of tissue PCB concentrations to the measurement endpoint target is a function of PCB persistence and fate in the environment. PCBs are extremely hydrophobic, and their affinity for sorption to soil and sediment, along with their tendency to partition into the lipids of aquatic organisms, determine their transport, fate, and distribution (Smith *et al.* 1988). PCBs degrade slowly, and may be present in sediment and tissue for long periods of time (Oliver *et al.* 1989; USEPA 1999).

PCBs in Sediment

PCBs in sediment may be altered by environmental weathering, anaerobic reductive dechlorination, and aerobic microbial degradation. When weathering is the dominant process, sediments become enriched with higher-chlorinated congeners as the lower-chlorinated forms are preferentially solubilized and vaporized (Erickson 2001; Cacela *et al.* 2002). During anaerobic reductive dechlorination, microorganisms transform PCBs into lower-chlorinated forms by partially dechlorinating the more highly-chlorinated congeners (Erickson 2001; Magar *et al*. 2005). The result is an increase in lower-chlorinated congeners (Abramowicz *et al.* 1993; Bedard and May 1996), which are generally less toxic and more readily attacked by aerobic bacteria (Brown *et al.* 1987b; Quensen *et al.* 1988; Abramowicz 1995; Bedard and Quensen 1995). Table 3 summarizes some of the scientific literature concerning the degradation of PCBs in sediment.

PCB half-lives in sediment vary from a few years for lower-chlorinated congeners to decades for higher-chlorinated forms. Van Metre *et al*. (1998) analyzed sediment core samples from 11 reservoirs, including White Rock Lake in Dallas, and determined the mean PCB sediment half-life to be 9.5 $\pm$ 2.2 years. Davis (2004) estimated the half-life of total PCBs in San Francisco Bay sediment to be about 20 years if all PCB inputs to the bay were eliminated. Half-lives of different PCB congeners varied from four years for lightly-chlorinated congeners to 30 years or more for highly-chlorinated forms. Potential exposure to the more bioaccumulative PCB congeners may continue for decades or longer after direct releases have been halted (Jönsson *et al.* 2003). Burial under uncontaminated sediments may be necessary to isolate the remaining PCBs from the food chain (Pakdeesusuk *et al.* 2003).

Other factors, such as mixing and dispersion of contaminated sediments and the potential for irreversible sorption of PCBs to sediment particles (see Apitz *et al.* 2005), may also reduce the bioavailability of remaining congeners.

Table 3. Summary of additional scientific literature reviewed for the Lake Worth TMDL concerning the degradation of PCBs in sediment.

PCBs in Sediment	References
Anaerobic reductive dechlorination of PCBs in sediment has been documented in laboratory and field studies.	Brown *et al.* (1987a,b); Brown and Wagner (1990); Hooper *et al.* (1990); Abramowicz (1995); Bedard and Quensen (1995); Bedard and May (1996); Bedard (2001); Imamoglu *et al.* (2002, 2004); Pakdeesusuk *et al.* (2003); Magar *et al*. (2005)
Most reductive dechlorination seems to occur in a short initial time period.	Abramowicz *et al.* (1993); Rhee *et al.* (1993b); Sokol *et al.* (1994, 1995, 1998a,b); Renner (1998)
Extent of reductive dechlorination is affected by differences in microorganism populations.	Brown *et al.* (1987b); Quensen *et al.* (1990); Sokol *et al.* (1994, 1998b); Cho *et al.* (2000)
Extent of reductive dechlorination is affected by PCB concentration in the sediment.	Quensen *et al.* (1988); Rhee *et al.* (1993b); Sokol *et al.* (1995, 1998a); Kim and Rhee (1997); Cho *et al.* (2002); Pakdeesusuk *et al.* (2005)
Extent of reductive dechlorination is affected by the degree of chlorination and the position of chlorines on the biphenyl.	Rhee *et al.* (1993b); Erickson (2001)
Some daughter congeners may be less susceptible to further dechlorination and pose a health risk.	Rhee *et al.* (1993a); Liu *et al.* (1996); Ganey and Boyd (2005)

Sediment samples were collected in Lake Worth in November 2000 and January 2001 for a USGS study (Harwell *et al.* 2003) conducted for the U.S. Air Force in conjunction with the facility investigations at NASFW and AFP4. Surface sediment samples were collected from 21 sites distributed throughout the lake. Core samples were collected at three of those locations, in the upper, middle, and lower part of the lake (see Figure 1 in Harwell *et al.* 2003). Total PCB concentrations in surface sediments were elevated in the Woods Inlet area, located immediately west of AFP4. Surface sediment concentrations in other parts of the lake ranged from less than the detection limit to 0.012 mg/kg. Except for one location immediately west of IH-820, total PCB concentrations in surface sediments were less than the detection limit in samples collected adjacent to and upstream from IH-820.

Core samples indicated that PCB concentrations in sediment were not detectable at any time in the upper-lake and prior to the late 1940s in the middle and lower-lake. Peak concentrations in the middle and lower-lake cores occurred in the 1960s, followed by an exponential decrease to the top of the cores (Harwell *et al.* 2003). This trend is typical of sediment cores collected in other urban water bodies, reflecting times of peak PCB use and

subsequent decline following USEPA restrictions (Van Metre and Callender 1997; Van Metre *et al.* 1997a,b, 1998, 2003a,b; Ging *et al.* 1999; Van Metre and Mahler 1999; Imamoglu *et al.* 2002). Although more recent PCB concentrations have declined in both cores, the peak concentrations in the Woods Inlet (middle-lake) core were approximately three times higher than those in the lower-lake core near the dam (Harwell *et al.* 2003).

Harwell *et al.* (2003) concluded that the spatial and temporal distributions of PCBs indicates the most likely source to be within the drainage area to Woods Inlet. This drainage area includes much of AFP4 and largely residential areas within the cities of Fort Worth and White Settlement. Trends in the Woods Inlet core suggested that PCBs might still be entering the lake, probably attached to suspended sediment in storm runoff (Harwell *et al.* 2003). Continuing external loading can delay declines in sediment and tissue PCB levels (Davis 2004).

A follow-up USGS study (Besse *et al.* 2005) was subsequently conducted to map the distribution of PCBs in bottom sediment in Woods Inlet, and to identify possible source areas of PCBs in the Woods Inlet watershed. The study collected surface sediment samples at 20 sites in Woods Inlet and its three tributary arms, sediment core samples in Woods Inlet outside the mouth of each arm, and storm-generated suspended sediment and bed sediment samples in the three tributaries.

Total PCB concentrations in surface sediments were highest near the mouth of Meandering Road Creek, and decreased across Woods Inlet toward the main body of the lake. Samples near the mouths of the other two tributaries were lower than at Meandering Road Creek and similar to concentrations in the middle of the inlet (Besse *et al.* 2005). The sediment core collected near the mouth of Meandering Road Creek contained peak PCB concentrations that were approximately 2.7 and 7.3 times the peak values in cores collected near the mouths of the other tributaries. Statistical cluster analysis of the PCB congener profiles in surface and bed sediment samples found several distinct clusters of congener signatures, including those formed by samples from (1) the Meandering Road Creek arm and the middle of Woods Inlet, (2) upstream urban reference sites and sites in Woods Inlet not influenced by Meandering Road Creek, and (3) the Gruggs Park Creek arm and upstream urban reference sites.

Besse *et al.* (2005) concluded that the spatial distribution of surface sediment PCB concentrations indicates a source area within the Meandering Road Creek watershed, that PCBs in Woods Inlet enter primarily from Meandering Road Creek, and that runoff from AFP4 is a major source of the PCBs in the creek. Suspended and streambed sediment PCB concentrations and distributions reflected AFP4 runoff contributions. PCB levels were highest at sites receiving runoff from only AFP4, intermediate at sites receiving runoff from both AFP4 and upstream locations, and lowest at sites where there was no AFP4 runoff contribution.

Besse *et al.* (2005) also detected evidence of the current transport of PCBs into Meandering Road Creek from an outfall at AFP4; however, there were large decreases in concentration with decreasing age in the sediment core nearest the outfall, indicating that loading has

decreased greatly since 1960s. The authors noted that there may be PCB sources at AFP4 other than the outfall investigated.

An additional investigation was conducted for the U.S. Air Force by Earth Tech, Inc. (Schultz *et al*. 2005) to determine if a source could be identified for the PCBs discharged through the AFP4 outfall identified by Besse *et al*. (2005) (Outfall 4), if other parts of the storm sewer system might be pathways for PCB-contaminated sediments, whether PCBs were infiltrating the storm sewer system from areas containing dense non-aqueous phase liquids (DNAPL) associated with old AFP4 landfills, and to identify other possible PCB sources not associated with AFP4. A variety of samples were collected, including sediment, concrete chips, and base groundwater flow (where present) in the storm sewer system; bed sediment samples in Meandering Road Creek; DNAPL and groundwater from selected wells, and soil samples from trench fill along a storm water drain near the DNAPL area (see Schultz *et al*. 2005 for complete description). High PCB concentrations were found in the DNAPL samples; however, analysis of congener patterns indicated that the DNAPL is not a significant component of the PCBs in Lake Worth and Meandering Road Creek.

PCBs were detected in all but one sample associated with the Outfall 4 system, and in all three samples associated with a second outfall (SSO) located upstream along Meandering Road Creek. The PCB concentrations in the Outfall 4 system were similar, and did not point toward any obvious major source within AFP4. PCBs were detected in all three Meandering Road Creek sediment samples collected downstream from the SSO outfall, but in none of the creek samples collected upstream from this location. This supports the findings of Besse *et al*. (2005) and indicates that there are no major sources upstream from AFP4. Analysis of congener patterns indicates that the PCBs in Meandering Road Creek and Lake Worth sediments appear to be related to those detected within the Outfall 4 storm sewer system and at a drain in the SSO system at AFP4 (Schultz *et al*. 2005).

Most of the sediment samples collected as part of the AFP4 investigations exhibit PCB concentrations similar to or less than the typical background concentrations found in other urban water bodies, and considerably less than values measured in more highly contaminated systems (Table 4). This suggests that recent loading has been similar to that typically encountered in urban settings. Van Metre et al. (2003a) concluded that a PCB concentration of 0.02 mg/kg appears to be typical of recently-deposited lake sediments in the Dallas urban area. All of the surface sediment samples collected in the main body of Lake Worth are less then this value. The only samples with values greater than 0.02 mg/kg were collected in Woods Inlet adjacent to AFP4, or in Meandering Road Creek in the areas influenced by the two AFP4 outfalls (see Table 4 and associated references). Only one sample (0.140 mg/kg collected in Woods Inlet near AFP4 by Harwell *et al.* 2003) exceeded the ROD remediation goal (0.1 mg/kg) for Meandering Road Creek sediments (USEPA 1996a).

Table 4. Surface sediment PCB concentrations measured in Lake Worth, Meandering Road Creek, and selected other water bodies located in urban and industrial areas. PCB concentrations include results from the analysis of both surface sediment samples and the surface layer of core samples. nd = not detected.

Water Body	PCB Concentration (mg/kg)	Reference and Comments
Lake Worth (Tarrant Co., TX) – Main lake – Woods Inlet	 nd – 0.012 0.015 – 0.140	Harwell *et al.* (2003)
Lake Worth (Tarrant Co., TX) – Woods Inlet (all samples) – Woods Inlet (adjacent to AFP4) Meandering Road Creek (Tarrant Co., TX) – upstream from AFP4 outfalls – downstream from AFP4 outfalls Other Woods Inlet tributaries	 0.0048 – 0.074 0.015 – 0.074 nd – 0.001 0.010 – 0.100 0.0011 – 0.0049	 Besse *et al.* (2005) Besse *et al.* (2005) Besse *et al.* (2005) and Schultz *et al.* (2005) (combined data) Besse *et al.* (2005)
Trinity River (Dallas/Fort Worth area, TX)	<0.05 – 0.087	Moring (1997) – composite of data from five sample sites.
White Rock Lake (Dallas, TX)	0.003	Van Metre and Callender (1997)
Town Lake (Austin, TX)	0.031	Van Metre and Mahler (1999)
Lake Harding (Harris Co., GA) Lake Walter F. George (GA) West Point Lake (GA)	0.046 <0.014 0.032	Van Metre *et al.* (1997a) – lakes are located downstream from the Atlanta metropolitan area.
Newbridge Pond (Long Island, NY) Packanack Lake (Passaic Co., NJ) Orange Reservoir (Essex Co., NJ)	approx. 0.600 approx. 0.075 approx. 0.100	Long *et al.* (2003) – lakes are located in heavily urbanized watersheds.
Lake Como (Fort Worth, TX) Echo Lake (Fort Worth, TX) Fosdic Lake (Fort Worth, TX)	0.037 – 0.060 0.114 – 0.156 0.089 – 0.114	Van Metre *et al.* (2003b) – fish consumption bans on all three lakes due to legacy pollutants.
Mountain Creek Lake (Dallas Co., TX) – Main lake – Cottonwood Bay – near Naval Air Station (NAS)	 0.002 –0.024 0.028 – 510 <0.005 – 0.70	Van Metre *et al.* (2003a) – Cottonwood Bay and area near NAS were impacted by releases from two U.S. Navy facilities.
Devil's Swamp Lake (Baton Rouge, LA)	0.11 – 0.30	Van Metre and Wilson 2004) – impacts from wastewater discharge from hazardous chemical disposal facility.
Newburgh Lake (Wayne Co., MI)	nd – 48 mean = 2.4	O'Meara *et al.* (2000) – located in major industrial area.
Fox River (near Green Bay, WI)	mean = 53 maximum = 710	Steuer (2000) (citing others) – area heavily contaminated by industrial discharges.

PCBs in Fish Tissue

PCBs are highly lipophilic (Matthews and Dedrick 1984), and rapidly accumulate in the tissues of aquatic organisms at levels considerably greater than that of both the water column and the sediments (Smith *et al.* 1988). PCB concentrations in aquatic organisms may be 2000 to more than a million times greater than that of the water column (USEPA 1999). Fish tissue PCB concentrations are influenced by a variety of factors (Swackhamer and Hites 1988), and can vary within the same water body (Stow *et al*. 1995; Lamon and Stow 1999) as well as among different fish species and size classes (Swackhamer and Hites 1988; Connor *et al.* 2005). Table 5 summarizes some of the scientific literature concerning factors that influence PCB uptake and elimination in fish.

PCB congener profiles can also vary greatly among fish species (Gerstenberger *et al.* 1997; Connor *et al.* 2005). Contaminated fish tissue is generally dominated by congeners with five to seven chlorines per molecule (McFarland and Clarke 1989; Kuehl *et al.* 1994; Willman *et al.* 1997; USEPA 1999; Ellis 2001; Sethajintanin *et al.* 2004). Kuehl *et al.* (1994) detected fish tissue PCBs in general at 91 percent of sites nationwide; however, hexa- and pentachlorobiphenyls were detected at 89 and 87 percent of the sites, respectively, octa- and dichlorobiphenyls at 35 and 31 percent, and nona- and decachlorobiphenyls at only 10 and 3 percent of sites. Congeners with more than seven chlorines are generally less bioavailable because they are more tightly bound to soil and sediment, and because they are usually present in smaller quantities. Congeners with fewer than five chlorines are more readily metabolized and eliminated, and thus do not bioaccumulate as readily (McFarland and Clarke 1989).

The time necessary for stable hydrophobic compounds such as PCBs to reach equilibrium in fish tissue is variable, difficult to determine, and generally very long (Farrington 1991; Catalan *et al.* 2004). Once equilibrium is reached, the time necessary for a contaminant to be eliminated from tissue is also long. The half-life of PCBs in lake trout has been estimated at 9 to 10 years (see Borgmann and Whittle 1992; Van Metre *et al.* 1998). Niimi and Oliver (1983) found that elimination half-lives of PCB congeners increased from five days to no apparent elimination as the number of chlorines increased, resulting in the selective elimination of lower-chlorinated congeners. de Boer *et al.* (1994) found virtually no elimination of hexa- and octachlorobiphenyls from eels, and suggested that the elimination of some highly-chlorinated PCBs may be so slow that the process cannot be observed. Catalan *et al.* (2004) determined the residence times of PCB congeners in brown trout to range from 2-3 years for lower-chlorinated forms to 20-30 years or more for higher-chlorinated forms.

Fish tissue samples were collected in Lake Worth in March-April 1999 as part of a USGS study (Moring 2002) conducted for the U.S. Air Force in conjunction with the facility investigations at NASFW and AFP4. The study was conducted as a followup to a public health assessment of the AFP4 site (TDH 1998). The public health assessment had concluded that, because several contaminants (including PCBs) were found to be elevated in mosquitofish tissue, exposure to contaminants through the aquatic food chain was a

potential human health hazard; however, information was inadequate to make a conclusive determination.

Table 5. Summary of additional scientific literature reviewed for the Lake Worth TMDL concerning PCB uptake and elimination processes in fish tissue.

PCBs in Fish Tissue	References
Fish tissue contaminant concentrations can vary within the same water body.	Stow *et al.* (1995); Lamon and Stow (1999)
Fish tissue contaminant concentrations can vary among different fish species, size classes within a fish species, and different tissues within a fish.	Swackhamer and Hites (1988); Connor *et al.* (2005)
Contaminant uptake and elimination processes can be influenced by fish characteristics such as lipid content, age, length, weight, diet and feeding habits, habitat use and movement, gender, reproductive status, contaminant transfer from females to young, growth dilution, and metabolism.	Smith *et al.* (1988); Farrington (1991); Borgmann and Whittle (1992); Pritchard (1993); Larsson *et al.* (1996); Ion *et al.* (1997); Stow *et al.* (1997); Jones and de Voogt (1999); Gobas and Morrison (2000); Stapleton *et al.* (2001a); Johnston *et al.* (2002); Fernandez *et al.* (2004)
Contaminant uptake and elimination processes can be influenced by environmental factors such as contaminant levels in food items, trophic position and length of the food chain, seasonal variation in contaminant availability, water column contaminant concentration, and sediment contaminant concentration and bioavailability.	Smith *et al.* (1988); Rasmussen *et al.* (1990); Farrington (1991); Pritchard (1993); Vander Zanden and Rasmussen (1996); Zaranko *et al.* (1997); Kidd *et al.* (1998a,b); Jones and de Voogt (1999); Gobas and Morrison (2000)
Tissue PCB concentrations are influenced by differences in congener bioavailability, equilibrium time, and susceptibility to uptake, biotransformation, and elimination.	Hansen *et al.* (1976); Wszolek *et al.* (1979); Niimi and Oliver (1983); Schmitt *et al.* (1985); Oliver and Niimi (1988); van der Oost *et al.* (1988); McFarland and Clarke (1989); Sijm *et al.* (1992); de Boer *et al.* (1994); Madenjian *et al.* (1999); Gundersen *et al.* (2000); Zhou and Wong (2000); Erickson (2001); Trowbridge and Swackhamer (2002)
PCB congener profiles can vary greatly among different ages and species of fish.	Gerstenberger *et al.* (1997); Fernandez *et al.* (2004); Connor *et al.* (2005)
PCB congeners are cleared from tissues at very different rates, and the same congener may be cleared at different rates in different species.	Matthews and Dedrick (1984); Schmitt *et al.* (1985)
The rate of decline of the more toxic coplanar PCB congeners may be much less than that observed for total PCBs.	Tanabe *et al.* (1987); Kannan *et al.* (1989); Coristine *et al.* (1996); Trowbridge and Swackhamer (2002)
Published uptake and elimination rates derived from laboratory studies may not reflect field conditions, limiting their use for the prediction of contaminant behavior.	Swackhamer and Hites (1988); de Boer *et al.* (1994)

Moring (2002) collected five smallmouth buffalo and ten specimens each of channel catfish, common carp, freshwater drum, largemouth bass, and white crappie (55 total fish). Analyses were conducted on skinless fillets for Aroclor mixtures 1248, 1254, and 1260. Aroclor 1248 was not detected in any of the samples, and the subsequent risk assessment (TDH 2000b) was based on tissue concentrations of Aroclor 1254, Aroclor 1260, and total PCBs. That assessment resulted in the issuance of the fish consumption advisory.

In November 2003, Giggleman and Lewis (2004) collected five to ten specimens each of the same species collected by Moring (2002), as well as five white bass specimens (48 total fish). Analyses were conducted on skinless fillets for total PCBs, 96 individual congeners, and Aroclor mixtures 1242, 1248, 1254, and 1260.

The largest number of individual PCB congeners (92) was found in smallmouth buffalo, while the smallest numbers were found in white crappie (67) and freshwater drum (68) (Giggleman and Lewis 2004). Nineteen congeners, consisting of penta- (5-6), hexa- (6), hepta- (7-8), and octachlorobiphenyls (1-2), were detected in all 48 fish. Three of these nineteen (PCB 105, 118, and 180) are coplanar congeners. Two additional coplanar heptachlorobiphenyl congeners (PCB 167 and 156) were detected in 41 and 42 fish, respectively. Five other coplanar congeners were detected in 14-25 fish specimens.

Table 6. Mean and range of total PCB concentrations (mg/kg) in Lake Worth fish tissue samples collected in March-April 1999 (Moring 2002) and November 2003 (Giggleman and Lewis 2004). Total PCB values for the Moring (2002) data were calculated as the sums of Aroclors 1254 and 1260. Sample values less than the detection limit were treated as one-half of the detection limit. Mean values from Giggleman and Lewis (2004) were calculated using the total PCB values provided for each fish in their Table 7. N = fish sample size. $\bar{x}_{03}/\bar{x}_{99}$ = mean of 2003 PCB concentration (Giggleman and Lewis 2004) divided by the mean of 1999 PCB concentration (Moring 2002).

	Total PCB Concentration in Fish Tissue (mg/kg)						
	Moring (2002)			**Giggleman and Lewis (2004)**			
Fish Species	**N**	**Range**	**Mean**	**N**	**Range**	**Mean**	$\bar{x}_{03}/\bar{x}_{99}$
Channel Catfish	10	0.11 – 0.30	0.19	10	0.010 – 0.099	0.040	0.21
Common Carp	10	0.075 – 2.1	0.69	5	0.089 – 0.23	0.15	0.22
Freshwater Drum	10	0.047 – 0.33	0.11	5	0.008 – 0.024	0.013	0.12
Largemouth Bass	10	0.050 – 0.22	0.12	10	0.011 – 0.082	0.034	0.28
Smallmouth Buffalo	5	0.10 – 0.72	0.22	5	0.076 – 2.5	0.61	2.8
White Crappie*	10	<0.050 – <0.10	0.065	8	0.006 – 0.020	0.012	0.18
White Bass	0	na	na	5	0.026 – 0.076	0.048	na
All Fish	55	<0.050 – 2.1	0.24	48	0.006 – 2.5	0.10	0.42

*Moring (2002) white crappie values were all less than the detection limit.

The common occurrence of the penta-, hexa-, and heptachlorobiphenyl congeners is consistent with the finding of other studies (McFarland and Clarke 1989; Kuehl *et al.* 1994; Willman *et al.* 1997; USEPA 1999; Ellis 2001; Sethajintanin *et al.* 2004; Connor *et al.* 2005). In 44 of the 48 total fish, the congener measured at the largest concentration was a hexachlorobiphenyl (PCB 153/132 or 138/160 coelutions). PCB 180, a coplanar heptachlorobiphenyl, was measured at the largest concentration in two carp specimens.

For the six fish species collected in both studies, the mean tissue concentrations for total PCBs were smaller in 2003 for all except smallmouth buffalo, where the mean was 2.8 times larger in 2003 than in 1999 (see Table 6 and Figure 3). Giggleman and Lewis (2004) suggested this may be at least partly a result of the smallmouth buffalo collected in 2003 being generally larger than those collected in 1999. The 2003 concentration means in the other species ranged from 12 to 28 percent of the 1999 means (Table 6). The mean total PCB concentration for all fish in 2003 (0.10 mg/kg) was less than half of that observed in 1999 (0.24 mg/kg).

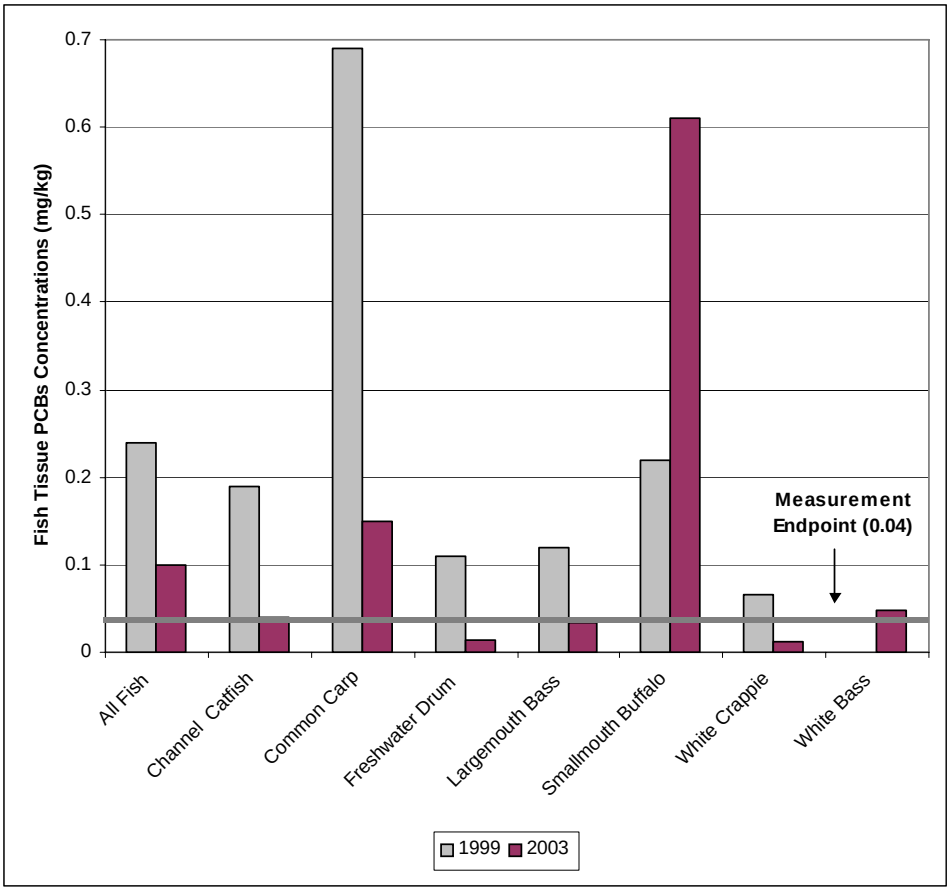

Figure 3. Mean total PCB concentrations in Lake Worth fish tissue samples collected in March-April 1999 (Moring 2002) and November 2003 (Giggleman and Lewis 2004). The TMDL measurement endpoint is indicated for comparison. See Table 6 and text for additional information.

The two data sets suggest that fish tissue PCB concentrations are generally declining in Lake Worth, although the 2003 mean concentration for all fish was still greater than the TMDL measurement endpoint concentration (<0.04 mg/kg) (Figure 3). Among the individual species examined, the 2003 mean PCB concentrations in channel catfish, freshwater drum, largemouth bass, and white crappie were at or less than the measurement endpoint, and white bass exceeded the endpoint by only 0.008 mg/kg.

Seasonality

Although there may be seasonal fluctuations in contaminant uptake and elimination, critical conditions such as low flow and seasonal variability in loading are not major influences on the long-term process of reducing PCBs in Lake Worth fish tissue. In addition, site remediation activities at AFP4 are expected to permanently mitigate major PCB sources, while any best management practices undertaken will likely be in place throughout the year to control any remaining sources.

Margin of Safety

A margin of safety is required in a TMDL in order to account for any uncertainty about the pollutant load and its association with water quality. The margin of safety may be an explicit component that leaves a portion of the potential assimilative capacity of a water body unallocated, or an implicit component established through the use of conservative analytical assumptions.

This TMDL uses an implicit margin of safety. USEPA (2000b) guidance on the assessment of contaminant data for use in fish advisories contains an extensive discussion of the assumptions and uncertainties present in the calculation of fish consumption limits. Conservative assumptions and calculations are used throughout the guidance to provide a margin of safety for the various uncertainties. Strict criteria exist concerning the types of studies and the data required to support assumptions and calculations. Numeric adjustments are made for the extrapolation of study results from animals or humans to the general population, and to provide a conservative upper bound on cancer risk values and a conservative oral reference dose (RfD) for noncarcinogens (see also USEPA 2005 and <www.epa.gov/iris/subst/0294.htm>). Adjustments are designed to provide a safe margin between observed toxicity and potential toxicity in a sensitive human.

USEPA assumes no safe threshold for exposure to carcinogens. Any exposure is assumed to pose some cancer risk. Noncarcinogenic effects occur with chronic exposure over a significant period of time. The oral reference dose (RfD) is defined in USEPA (2000b) as "an estimate (with uncertainty perhaps spanning an order of magnitude) of a daily exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime." Calculated RfDs reflect the assumption that, for noncarcinogens, a threshold exists below which exposure does not cause adverse health effects. RfD calculations use modifying and uncertainty factors to account for variables such as the variability of responses in human populations, differences in responses between animal study species and humans, and gaps in available data. The RfD is calculated

so there is little probability of an adverse health effect due to chronic exposure to concentrations below the RfD (USEPA 2000b).

Use of the most protective target concentration for a contaminant provides additional assurance that protection from both carcinogenic and noncarcinogenic effects will be achieved. Because the goal of this TMDL is removal of the fish consumption advisory through reduction of the consumption risk, the margin of safety inherent in the USEPA guidance, combined with the conservative use of endpoint targets, will provide an adequate margin of safety for the protection of human health. The decline of tissue PCB concentrations to an acceptable level of risk will allow TDSHS to remove the fish consumption advisory, which will restore the fish consumption use to this water body.

Pollutant Load Allocation

Restrictions on the use of PCBs generally have resulted in a slow but steady decline in environmental residues (Smith *et al.* 1988). PCB levels in lake sediment cores have shown good agreement with production and usage history, with peak concentrations appearing at the times of peak use (Ricci *et al.* 1983; Oliver *et al.* 1989; Van Metre and Callender 1997; Van Metre *et al.* 1997a,b, 1998, 2003a,b; Ging *et al.* 1999; Van Metre and Mahler 1999; Imamoglu *et al.* 2002). This trend is also apparent in Lake Worth cores (Harwell *et al.* 2003; Besse *et al*. 2005). Higher concentrations generally appeared deeper in the cores, indicating that input and accumulation were decreasing with time. Although residues continue to persist in deeper parts of the cores, burial by more recently deposited sediments may result in effective exclusion of the contaminants from bioavailability to aquatic life (Ricci *et al.* 1983; Pakdeesusuk *et al.* 2003).

Decreases in fish and human tissue concentrations of PCBs have been observed where no major additional inputs are occurring (Brown *et al.* 1985; Hovinga *et al.* 1992; Bremle and Larsson 1998; Sjödin *et al.* 2004). Reviews of tissue data collected from a variety of water bodies in northern Europe between 1967 and 1995 have found a significant decrease in organochlorine concentrations over time (Skåre *et al.* 1985; Bignert *et al.* 1998). Declining tissue PCB concentrations have also been reported in the Great Lakes (Scheider *et al.* 1998; Chernyak *et al.* 2005). Less consistent trends in tissue PCB levels may be a reflection of the congener-specific nature of PCB metabolism and degradation. In addition, strong oscillations in PCB levels influenced by food web interactions can be superimposed over a gradual declining trend (see Borgmann and Whittle 1992).

Continuing decreases in environmental PCB levels are expected, although the necessary time frame is subject to debate (Loganathan and Kannan 1994; Jönsson *et al.* 2003). Within the context of this TMDL, legacy PCBs are considered background sources that reflect the site-specific release history and loss rates of the subject area. There are no existing TPDES permits within the Lake Worth watershed that specifically authorize the discharge of PCB-containing wastewater or storm water. Continuing loading may occur from nonpoint source runoff, leaching, or erosion of the various sinks that may exist within the watershed. Any

residual contamination at AFP4 may be a source to Lake Worth while site remediation continues.

Most of the Lake Worth sediment samples collected as part of the investigations at AFP4 exhibit PCB concentrations similar to or less than the typical background concentrations found in other urban water bodies, and considerably less than values measured in more highly contaminated water bodies. Only one sample value exceeded the ROD remediation target for Meandering Road creek sediment. This indicates that recent and current loading is similar to that typically encountered in urban areas. Elevated fish tissue PCB concentrations are likely to be predominately a reflection of higher loadings to Meandering Road Creek and Woods Inlet in the past, rather than of current loading. Available data suggest that fish tissue PCB concentrations in Lake Worth are generally declining, although the most recent mean value for all fish species still exceeded the TMDL measurement endpoint. Mean concentrations for four individual species were at or less than the measurement endpoint.

Remediation of any remaining PCB contamination associated with AFP4 through the Air Force Installation Restoration Program, the Superfund Program and the Record of Decision for the site, or the TCEQ Defense and State Memorandum of Agreement Program, will eliminate the major source of PCBs to Lake Worth. Additional planning and sampling, as well as any decision to modify the ROD to take more recent findings into account, will be addressed during the implementation phase of the TMDL. Continuing natural attenuation in the lake is also expected, via processes such as degradation and metabolism of PCBs in tissue (Matthews and Dedrick 1984) and sediment, mixing and dispersion of contaminated sediments, irreversible sorption of PCBs to sediment particles, and continuing deposition of clean sediment that results in the burial of contaminated sediments and isolation from the food chain (Jones and de Voogt 1999; Pakdeesusuk *et al.* 2003; Brenner *et al.* 2004; Apitz *et al.* 2005).

Natural attenuation is generally a preferred option for the elimination of legacy pollutants within a water body. More drastic alternatives, such as sediment removal by dredging, can result in considerable habitat disturbance and destruction. Considerable uncertainty and variability exists in the risk of trophic transfer of PCBs during the dredging of contaminated sediments (von Stackelberg *et al.* 2002). Sediments resuspended during dredging further expose aquatic life to contaminants and the potential for additional uptake, cause abrasive damage to gills and sensory organs of fish and invertebrates, and interfere with fish prey selection (O'Brien 1990; Waters 1995; Weston *et al*. 2002).

Morrison *et al.* (2002) suggest that substantial improvements in biota contaminant levels could be achieved by reducing bottom sediment PCB concentrations, but that remediation of contaminated sediments that are dispersed over large areas may not be ecologically sound, technologically feasible, or cost effective. Weston *et al.* (2002) studied the impact of dredging in San Francisco Bay on total DDT body burdens in ten species. Surficial sediment DDT was largely unaffected, probably due to incomplete dredging and the dispersal of previously-buried contaminated sediments, allowing continuing exposure. Dredging

activities remobilized large amounts of DDT and dramatically increased body burdens in the biota. The authors determined that post-remediation conditions were comparable with or slightly worse than conditions before dredging, and suggested capping of sediment in place as an alternative. In another case, however, Bergen *et al.* (1998) found no extensive redistribution of PCBs as a result of dredging in a shallow estuarine system. Alternatives such as dredging and/or eradication of contaminated fish communities and restocking (Bergen *et al.* 1998; Bremle and Larsson 1998; O'Meara *et al.* 2000; Steuer 2000; Renner 2001; USEPA 2002) may be better justified at sites heavily contaminated by point source discharges and major spills. Decisions concerning any additional remediation activities such as these, as well as any additional mitigation measures or best management practices will be addressed during the implementation phase of the TMDL.

Public Participation

The public and stakeholder participation process in TMDL development is described in detail in the agency guidance document, *Developing Total Maximum Daily Load Projects in Texas: A Guide for Lead Organizations* (TNRCC 1999). Additional information about the public and stakeholder participation process in TMDL development and implementation can be found on the Web at **www.tnrcc.state.tx.us/water/quality/tmdl/tmdl_guidance.html**.

Notice concerning the release of the draft Lake Worth TMDL for public comment and the scheduling of a public meeting was mailed to potential stakeholders in mid-April 2005. Notice was also published in the *Texas Register* and the *Fort Worth Star-Telegram*. Both the notice and the draft TMDL were posted on the TCEQ web site during the comment period. A public meeting to accept oral comments was held in Fort Worth on May 12, 2005. The 30-day public comment period ended on May 20, 2005. The Response to Public Comment and the revised TMDL to be submitted to USEPA will be posted on the TCEQ web site and made available to all interested parties. The Response to Public Comment is a separate document that summarizes TCEQ responses and any modifications made to the TMDL document as a result of the public comment process.

Implementation and Reasonable Assurance

The TMDL development process involves the preparation of two documents (1) a TMDL, which determines the amount of pollutant a water body can receive and meet applicable water quality standards, and (2) an implementation plan, which is a detailed description and schedule of regulatory and voluntary management measures necessary to achieve the pollutant reductions identified in the TMDL. It is the policy of the TCEQ to develop implementation plans for all TMDLs adopted by the Commission (TNRCC 1999; TCEQ 2002), and to assure the plans are implemented.

During TMDL implementation, the TCEQ works with stakeholders to develop the management strategies needed to restore water quality to an impaired water body. This information is summarized in the TMDL Implementation Plan (TMDL IP), which is separate

from the TMDL document. Preparation of an implementation plan is critical to ensure water quality standards are restored and maintained.

Preparation of the implementation plan for Lake Worth will be initiated upon Commission approval of the TMDL. The TMDL IP will detail any activities such as mitigation measures, permit actions, best management practices, and additional sampling and monitoring determined to be necessary to restore water quality. Additional fish tissue sampling at appropriate intervals will allow progress toward the assessment endpoint to be tracked and evaluated. The TMDL IP will also discuss available information on the effect of natural attenuation on environmental PCB concentrations. These steps will provide reasonable assurances that the regulatory and voluntary activities necessary to achieve the pollutant reductions will be implemented.

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