Monitored Natural Recovery of Contaminated Sediment in Cottonwood Bay, Texas

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ABSTRACT
Monitored natural recovery (MNR) is a response action for contaminated sediment that relies on natural processes to reduce ecological and human health risks to acceptable levels over time. Multi-media long-term monitoring is necessary to verify remedy effectiveness and success. Whereas MNR has most often been employed together with other engineered remedies to address contaminated sediment, this case study describes an evaluation demonstrating the suitability of MNR with adaptive management as a stand-alone sediment remedial strategy for Cottonwood Bay, located in Dallas, Texas. Current risks at the Cottonwood Bay site are associated with human consumption of fish contaminated with polychlorinated biphenyls (PCBs). The lines of evidence used to evaluate MNR included documentation of source control; review of PCB burial/isolation and surface sediment recovery; evaluation of sediment deposition and sediment stability; and demonstration of decreasing temporal trends in fish tissue PCB concentrations. The multiple lines of evidence demonstrated that source control and natural recovery processes have contributed substantially to reduced concentrations of PCBs and metals in surface sediments and PCBs in fish. MNR has been approved by the Texas Commission on Environmental Quality (TCEQ) and is incorporated as a stand-alone remedy in the revised Remedial Action Plan (RAP) for Cottonwood Bay. A long-term monitoring plan is being implemented as part of an MNR response action to confirm decreasing trends in surface sediment and fish fillet PCB concentrations over time and to demonstrate the continued effectiveness of natural recovery processes.

KEYWORDS
Monitored natural recovery; contaminated sediment; polychlorinated biphenyls; sediment deposition

Introduction
Monitored natural recovery (MNR) is a response action for contaminated sediments that relies on natural processes to reduce ecological and human health risks to acceptable levels and long-term monitoring to verify success of the remedy over time (Magar et al. 2009; Magar and Wenning 2006; USEPA 2005). MNR processes may include physical isolation, reduction in contaminant mobility/bioavailability, chemical transformation, and dispersion. Since ecological and human health are affected by contaminants in surface sediments, the primary recovery processes for MNR are natural sediment burial and contaminant transformation/weathering to less toxic forms (Magar and Wenning 2006). The United
States Environmental Protection Agency (USEPA) (2005, 2014a) recognizes MNR as an appropriate sediment remedial approach, both as a primary remedy and in combination with other remedies.

MNR has been used as a remedy at several large contaminated sediment sites. For example, the following sites implemented remedies that combined multiple remedial technologies including MNR: Study Area 7 in the Lower Hackensack River (Jersey City, NJ) (ITRC 2014), the Sangamo-Weston/Twelvemile Creek/Lake Hartwell Superfund Site (Pickens County, SC) (USEPA 1994a), the Weyerhaeuser Co Plymouth Wood Treating Plant/Lower Roanoke River (OU2) Site (Plymouth, NC), and the Wyckoff/Eagle Harbor Superfund Site (Bainbridge Island, WA) (USEPA 1992, USEPA 1994b). However, at these and other sites, MNR is often combined with other remedies that include multiple technologies (e.g., dredging, capping, enhanced MNR), with MNR simply complementing more aggressive remedies by addressing low-risk areas where natural processes (e.g., natural sedimentation) contribute to reduced surface sediment concentrations.

Bridges and Gustavson (2014) recommend that MNR should be the first remedy considered for contaminated sediments, with more invasive technologies considered only as needed to meet project objectives. Nevertheless, few sites have employed MNR as a primary or sole remedy, without enhancement. The Lake Hartwell Superfund Site was one of the first sites to fully embrace natural MNR (USEPA 1994a) by evaluating and monitoring the deposition of cleaner sediment and polychlorinated biphenyl (PCB) dechlorination (Brenner et al. 2004; Magar et al. 2005a, b). In addition, natural recovery processes have been evaluated at St. Mary’s River (Ontario, Canada) where sediments demonstrated lower contaminant levels and invertebrate toxicity in the upper 5 cm as compared to the upper 10 cm, suggesting the deposition of cleaner sediment is resulting in natural recovery of sediments over time (Milani, Parrot, and Grapentine 2021). Using a multiple lines of evidence approach, the current case study provides a further, unique example of the feasibility of MNR as an effective stand-alone remedial strategy for sediment in Cottonwood Bay, located in Dallas, Texas.

Cottonwood Bay is part of a public reservoir located between Dallas and Grand Prairie, Texas, formed from the damming of Cottonwood Creek, a tributary to the larger Mountain Creek Lake. Cottonwood Bay is approximately 1,650 m long, 215 m wide, and ranges from 0.6 to 2.1 m deep. It is connected to the main body of Mountain Creek Lake via a constructed “diversion” channel (Figure 1). Two former United States Navy-related facilities were located on the northwest shore of Mountain Creek Lake, adjacent to Cottonwood Bay: Naval Air Station (NAS) Dallas and the former Naval Weapons Industrial Reserve Plant (NWIRP). Transformer fluid and plating lines at these facilities likely contributed to elevated concentrations of PCBs and metals in Cottonwood Bay sediments through groundwater migration, overland storm water runoff, and storm water outfall discharges.

Findings from ecological and human health evaluations conducted for Cottonwood Bay (GSI 2016a, 2016b) demonstrate that the only remaining risk at Cottonwood Bay is associated with human consumption of fish tissue contaminated with PCBs. Metals are present, but surface sediment concentrations of metals do not pose adverse risks to human health or the environment. Several lines of evidence are used to demonstrate natural recovery of PCB-contaminated surface sediments at this site, including documentation of source control; PCB contaminant burial/isolation and surface sediment recovery; sediment deposition and
sediment stability; decreasing temporal trends in fish tissue PCB concentrations; and modeling of the time required to achieve target PCB concentrations in edible tissues of various fish species. We also show how natural recovery processes have contributed to historical reductions in surface sediment concentrations of metals. Oversight by the Texas Commission on Environmental Quality (TCEQ) was particularly rigorous in the case of Cottonwood Bay because there are few precedent sites that solely utilize MNR. Multiple lines of evidence, through extensive data and analysis, were used to support MNR within Cottonwood Bay.

Materials and methods

Source control measures for Cottonwood Bay were evaluated through review of site documentation, while natural recovery processes were evaluated through data collection and analysis, as described below.

Sediment cores

Sediment cores were collected in Cottonwood Bay to evaluate chemical concentrations with depth. In 2015, sediment cores were collected from four locations using a Vibracore (Figure 2). Lake elevation was recorded at the time of the survey, and the depth to the top of the surface water/sediment interface was measured using a 200-kilohertz (kHz) depth
Cores were transferred to an onshore processing station, where they were photographed and logged in general accordance with the Unified Soil Classification System. Cores were subsampled at intervals of 0–15 cm, 15–30 cm, 30–60 cm, 60–90 cm, and 90–120 cm, or to the depth of refusal, whichever was shallower. Each sample was analyzed for cadmium, chromium, copper, lead, and zinc (method SW-846 6020A), mercury (method SW-846 7471A/B), PCB Aroclors 1242, 1254, and 1260 (method SW-846 8082A), and grain size (method ASTM D422).

**Surface sediment collection and analyses**

Surface sediment samples were collected from Cottonwood Bay in 1994, 1995, 1996, 1999, 2000, 2001, 2004, 2011, 2014, and 2015 (ENVIRON 2015; NAVFAC 2011; USA Environment 2014; USGS 1997, 2002). Each of these sampling events involved retrieving sediment from the top 6 in of the sediment bed using a box core, gravity core, or vibracore sampler. Samples were analyzed for cadmium, chromium, copper, lead, and zinc (method

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**Figure 2.** Sampling Locations in Cottonwood Bay. Stars mark the 2015 sediment core locations, triangles mark the 2015 geochronology cores locations, open squares mark the historical surface sediment sample locations (1994–1996 and 1999–2001), and closed squares mark the historical surface sediment sample locations (2004, 2011, 2014, and 2015).
SW-846 6020A or similar), mercury (method SW-846 7471A/B or similar), PCB Aroclors 1242, 1254, and 1260 (method SW-846 8082A or similar), and grain size (method ASTM D422). The sediment samples reported by USGS (1997, 2002) were extracted for PCB analysis using organic solvents and analyzed using gas chromatography/electron capture detection. The samples reported by NAVFAC (2011) were extracted for PCB analysis using USEPA method 3546, analyzed by method SW-846 8082A, and underwent data validation with reference to the sampling analysis plan, DoD Quality Systems Manual version 4.1 laboratory requirements (2009), USEPA National Functional Guidelines for Organic Data Review (1999), USEPA National Functional Guidelines for Inorganic Data Review (2004), and TCEQ guidance document RG-366/TRRP-13. The samples reported by USA Environment (2014) were prepared for PCB analysis using method 3541, analyzed by method SW-846 8082A, and validated in accordance with TCEQ guidance document RG-366/TRRP-13. The samples reported by ENVIRON (2015) were prepared for PCB analysis using methods SW-8462541 and SW-846 3665A and underwent a data usability review in accordance with the TCEQ guidance document RG-366/TRRP-13. Surface sediment sampling locations are shown on Figure 2.

**Surface-weighted average concentrations**

To compare surface sediment chemical concentrations over time, surface-weighted average concentrations (SWACs) were calculated using the available surface sediment data. SWACs are calculated as follows:

\[
SWAC = \frac{\sum_{i=1}^{n} (C_i \times A_i)}{\sum_{i=1}^{n} A_i}
\]

(Equation 1)

where:
- \(C_i\) = Sediment concentration at location \(i\) (mg/kg)
- \(A_i\) = Area associated with location \(i\) (hectares)
- \(n\) = Number of locations within the area of interest

Thiessen polygons, a spatial weighting technique which assigns an area to each sample location based on its proximity to neighboring samples, were used to represent the area associated with each sample location. Each polygon contains only one sample location, and the boundaries of the polygon are created by drawing lines perpendicular to and equidistant from the neighboring sample locations.

SWACs were calculated using surface sediment data from 0 to 15 cm for all sample locations. If multiple samples were collected within the 0 to 15 cm interval, a length weighted average concentration for that interval was calculated. Non-detect values were assigned concentrations of one-half the detection limit.

Assessment of surface sediment temporal trends required careful consideration because past sampling was designed for contaminant delineation rather than temporal trend analysis. As a result, the spatial coverage of sampling was not comparable from year to year. To overcome this challenge, data were grouped into two periods for SWAC calculations. The first period (7 years) included data from 1994, 1995, 1999, 2000, and 2001, and the second period (11 years) included data from 2004, 2011, 2014, and 2015. Grouping the data into these periods provided a similar number of samples and spatial
coverage across the two groups. SWACs were calculated for all chemicals that exceeded background in more than 50\% of the samples during the first period for both East and West Cottonwood Bay (Figure 1). Background concentrations were established in a TCEQ (2011) Corrective Action Order based on a detailed review of sediment contaminant concentrations at locations in Mountain Creek Lake and in tributaries that were not impacted by past Navy releases. If less than 50\% of the samples initially exceeded background levels, then significant decreases in surface sediment concentrations were not expected over time. The total number of samples analyzed for each chemical during each time period is provided in Table 1. A statistical analysis was performed on the concentration data using a spatially weighted t-test to determine differences in the SWAC between the first period and the second period. For each area and analyte, the concentrations were weighted according to the average area used for the SWAC calculations (Pasek 2014).

**Geochronology**

Sediment geochronology cores were collected at three locations in Cottonwood Bay, collocated with the sediment chemistry cores in 2015 (Figure 2). After sediment cores were collected and transferred to an onshore processing station, the sediment cores were photographed, divided into 2-cm intervals, and subsampled for lead-210 (Pb-210) and cesium-137 (Cs-137) isotope analyses. Pb-210 is a naturally occurring radioactive element that is part of the uranium-238 radioactive decay series. Pb-210 falls with precipitation and dust at a nearly constant annual rate on earth. The Pb-210 that falls into the lake becomes permanently fixed onto suspended sediment particles, which in turn become part of the sediment bed. The half-life of Pb-210 (22.3 years) can be used to calculate the age of sediments across depth in the sediment column and the rate of sediment accumulation. Pb-210 dating relies on two simplifying assumptions to support the requirement of a constant Pb-210 flux into the sediments: (1) the sediments have a relatively uniform grain size distribution with depth, and (2) the sediments historically have had a relatively constant deposition rate (Brenner et al. 2001, 2004). To the extent possible, these conditions were confirmed visually during the initial inspection of the sediment cores and through examination of particle size distribution, moisture content, and Pb-210 vertical concentration profiles in the cores.

Cs-137 serves as an independent tracer to validate the Pb-210 chronology. The Cs-137 data are interpreted based on the 1959 and 1963 major input peaks associated with atmospheric releases from nuclear weapons testing. Plots of Cs-137 activity against depth are used to estimate average deposition rate and to validate the Pb-210 data.

Pb-210 was analyzed by Flett Research Limited (Winnipeg, Manitoba, CA) based on the method of Eakins and Morrison (1978) and Flynn (1968). In summary, Pb-210 activity was determined by measurement of the polonium-210 (Po-210) daughter, which is in secular equilibrium with Pb-210 within 2 years of Pb-210 deposition. The sample was first spiked with a Po-209 yield tracer, then digested in hot nitric acid. The digest was dried and made up in 1.5 N HCl, and then the Po-210 and Po-209 alpha emitting isotopes were plated out on silver disks followed by alpha spectrometry to determine the activity of the polonium isotopes. The Cs-137 method was modified from EML HASL-300 Method Ga-01-R (EML, DOE (Environmental Measurements Laboratory, United States Department of Energy) 1997). All samples were compressed at ~3000 psi in a hydraulic press that produced
Table 1. Sediment chemical surface-weighted average concentrations (SWACs) at two time periods.

| Area                | Years                        | SWAC (mg/kg)
|---------------------|------------------------------|------------------------
|                     |                              | Cadmium | Chromium | Copper | Lead | Mercury | Zinc | Aroclor-1242 | Aroclor-1254 | Aroclor-1260 |
| East Cottonwood Bay | 1994–1996, 1999–2000, 2001   | 3.9 (n = 8) | 267 (n = 17) | 50 (n = 17) | 75 (n = 17) | 0.16 (n = 15) | 292 (n = 17) | 0.008 (n = 12) | 0.07 (n = 12) | 0.19 (n = 12) |
|                     | 2004, 2011, 2014, 2015       | 7.1 (n = 25) | 296 (n = 25) | 44 (n = 25) | 59** (n = 25) | 0.15 (n = 25) | 223* (n = 25) | 0.002 (n = 25) | 0.04 (n = 25) | 0.08* (n = 25) |
| West Cottonwood Bay | 1994–1996, 1999–2000, 2001   | 3.3 (n = 23) | 142 (n = 37) | 49 (n = 37) | 53 (n = 37) | 0.13 (n = 33) | 212 (n = 37) | 0.019 (n = 22) | 0.080 (n = 22) | 0.12 (n = 22) |
|                     | 2004, 2011, 2014, 2015       | 2.8 (n = 40) | 80* (n = 40) | 33* (n = 40) | 39* (n = 40) | 0.010 (n = 40) | 150* (n = 40) | 0.011 (n = 40) | 0.010 (n = 40) | 0.044* (n = 40) |

a. This analysis included contaminants of concern where more than 50% of samples exceeded background (TCEQ (Texas Commission on Environmental Quality) 2011) during the first time period (1994–2001)

b. Bold font identifies the lower value by area

* indicates a statistically significant decrease (p < 0.05) from the first period to the second period

** indicates a marginally significant decrease (p < 0.1) from the first period to the second period
pancakes of 2.187 in diameter of maximum and relatively similar density. Cs-137 was determined by counting the gamma emissions (from each sample pancake) at 661.6 KeV that are emitted in 82.5% of the decays on high purity HPGe detectors. The method detection limit (MDL) was 0.3 DPM/g for an 80,000 second counting period when measuring 9 g of dry sample at a 95% confidence level.

**Fish**

Fish were collected in 1994, 1995, 2008, and 2015 (USGS (United States Geological Survey) 1995, USGS 1996, TDSHS (Texas Department of State Health Services) 2010, ENVIRON 2015, TDSHS 2016). Fish were collected in West Cottonwood Bay, areas within Mountain Creek Lake, and the diversion channel between Cottonwood Bay and Mountain Creek Lake. Fish sampling was conducted by overnight gill net and electrofishing, and samples were analyzed as fillets. Fish species were collected to represent bottom-feeding fish, such as common carp (Cyprinus carpio), freshwater drum (Aplodinotus grunniens), and small-mouth buffalo (Ictiobus bubalus), and game fish, such as channel catfish (Ictalurus punctatus), largemouth bass (Micropterus salmoides), and white bass (Morone chrysops). Forage fish were also collected in 2015 for ecological risk assessment purposes, but the data could not be evaluated with respect to temporal trends as comparable earlier data were unavailable. Fish tissue samples were analyzed for Aroclors in 1994 and 1995 by gas chromatography with electron-capture detection; the samples were extracted for analysis using organic solvents and gel permeation chromatography, and quality assurance/quality control measures were performed as described in a study plan provided to the Navy, USEPA, and TNRCC (Leiker et al. 1995; USGS 2003).

Fish samples reported by TDSHS were prepared and analyzed for PCB congeners in 2008 and 2015 by established USEPA methods (TDSHS (Texas Department of State Health Services) 2010, GERG (Geochemical and Environmental Research Group) 2013, TDSHS 2016). The 2015 fish tissue analyses reported by ENVIRON (2015) were performed by the same laboratory and followed the same methods as the TDSHS analyses, to maximize comparability. PCBs in fish tissue samples were extracted by macerating a known mass of sample in methylene chloride and sodium sulfate. This extract was then filtered and concentrated. A subsample was taken for lipid content determination, and the remaining sample was further concentrated and reconstituted in hexane for subsequent purification and PCB analysis. Prior to extraction, surrogate standards (PCB103 and PCB108) were added to each sample to monitor method performance.

The lipid content of the sample was determined by evaporating the sample to dryness and reconstituting the sample in a known volume of methylene chloride. A small known volume of this sample was transferred to a pre-weighted filter, the solvent allowed to fully evaporate, and the mass of the remaining lipid content determined by weight.

The sample extract for PCB analysis was purified by silica/alumina chromatography (adapted from EPA Method 3630) followed by gel permeation chromatography (GPC) (adapted from EPA Method 3640). Silica/alumina chromatography removes polar compounds from the sample, while GPC chromatography uses size exclusion chromatography to separate high molecular weight biological molecules (e.g., lipids) from the target analytes. PCB concentrations were quantified by gas chromatography-mass spectrometry (GC/MS) with selected ion monitoring (SIM) using a method adapted from
EPA Method 1668. The gas chromatography used a 30-m x 0.25 mm inner diameter fused silica capillary column with a nonpolar phenyl polymer bonded phase. The SIM mode of identification used three identifying ions for each analyte: one quantitation ion and two confirmation ions. Retention times and instrument response were checked by initial calibration using a standard PCB mix. Additional calibration standards were run as continuing calibration verification at the start of each analytical sequence or every 12 hours of operation (whichever was most frequent). For quantitation of PCBs in samples, tetrachloro-meta-xylene was used as the internal standard. Solvent blanks were run after the initial calibration and after each continuing calibration validation. Method blanks were analyzed at a rate of at least one per 20 samples. A standard reference material (NRC CARP-2) was also analyzed; results met quality control requirements.

**Fish tissue time to recovery**

A provisional analysis was conducted to estimate how much time would be required for fish consumption advisories to be lifted for Mountain Creek Lake and Cottonwood Bay under an MNR remedy. Certain key inputs (e.g., PCB concentrations in newly settling sediments and PCB exposures for fish moving in and out of Cottonwood Bay) were estimated, with additional data collection and model refinement to be conducted during long-term monitoring implementation. For illustration, we report here the time to recovery analysis for largemouth bass within Cottonwood Bay. Because this species is relatively territorial, the fish captured within Cottonwood Bay are likely resident individuals. The target sediment concentration to achieve lifting of a fish consumption advisory specifically for largemouth bass in Cottonwood Bay was identified by dividing the TDSHS Health Assessment Comparison (HAC) value for fish tissue (0.047 mg/kg PCBs as the sum of 43 congeners on a wet weight basis) by an empirical, species- and area-specific biota-sediment accumulation factor (BSAF). Site-specific human health risk assessment, considering factors such as local fish consumption rates and PCB loss during cooking, might have supported a different target sediment concentration but was deemed incompatible with TCEQ’s goal of eliminating fish consumption advisories.

Time to recovery for PCBs in sediment was estimated by comparing modeled future sediment concentrations to the target sediment concentration. The first year where the modeled concentration was less than or equal to the target concentration was considered the time to recovery. Future sediment concentrations were modeled using the SEDCAM sediment attenuation model (Jacobs, Barrick, and Ginn 1988):

\[
C_t = C_b \frac{D_s}{D_m} + C_{t-1} \left(1 - \frac{D_s}{D_m}\right)
\]

(Equation 2)

where

- \(C_t\) = surface sediment concentration at year \(t\) (milligram(s) per kilogram; mg/kg),
- \(C_b\) = concentration in incoming sediments (mg/kg),
- \(C_{t-1}\) = surface sediment concentration at year \(t-1\) (mg/kg),
\[ D_s = \text{sediment deposition rate (cm/year)}, \]
\[ D_m = \text{sediment mixing depth (cm)}. \]

To address uncertainty, the model was run probabilistically using R (R Core Team 2017). Parameters with more than one value were input into the model as distributions, and the model was run using these distributions over 5000 iterations.

**Results and discussion**

**Source control**

The effectiveness of any sediment remedy, including MNR, depends on the control of contaminant sources to prevent sediment recontamination (USEPA 2005). In support of the MNR remedy assessment, source control measures implemented at the site were comprehensively reviewed.

Between 1971 and 2009, many measures to control contaminant sources to Cottonwood Bay were implemented. At NWIRP and NAS, military operations ceased, an oil-spill recovery system was installed, oil was removed from on-site wells, sewage was rerouted to external treatment plants, contaminated soil was removed, groundwater pump-and-treat systems were installed, and a hazardous waste container storage area was closed (NAVFAC (Naval Facilities Engineering Command) 2009). The comprehensive historical source control measures implemented at the site eliminated or controlled chemical sources from migrating into the lake. A long-term monitoring program, included as part of an MNR remedy, will continue to document whether contaminant sources to Cottonwood Bay are effectively controlled by monitoring tributaries and outfalls that contribute sediment into the Bay.

**Sediment deposition and sediment stability**

The deposition of cleaner sediments over time and stability of newly deposited sediments is another line of evidence used to demonstrate the effectiveness of MNR. The cleaner material provides a natural cap over contaminated sediments, isolating them from the biologically active zone, which is typically considered the top 15 cm of sediment (ITRC (Interstate Technology & Regulatory Council) 2011; USEPA 2015). Geochronology (sediment dating across depth) shows the rates at which new sediments are deposited. Sediment grain size analyses provide additional evidence of ongoing sediment deposition. Potential sediment scour via wind-driven resuspension and storm water runoff is also considered in this sediment stability evaluation.

**Geochronology**

Using a constant rate of supply (CRS) model, the age at the 1963 peak of Cs-137 was calculated using geochronology core data from 2015. Sedimentation rates of 0.6 to 1.9 cm/yr were calculated for the past eighteen-year period, further demonstrating that Cottonwood Bay is a depositional environment. The 2015 results agree with a prior sediment accumulation estimate of 1 cm/yr in Cottonwood Bay, which was developed in 2003 by evaluating Cs-137 concentrations and concentrations of DDT and dieldrin compared to their historical uses (USGS 2003).
**Grain size**

The presence of fine-grained sediments is indicative of a low energy, depositional environment (ASCE (American Society of Civil Engineers) 2008). Grain size distribution results presented in the Revised Sediment Background Analysis Report (Newfields 2003) show that sediment deposits in Cottonwood Bay are predominantly composed of silts and clays. Consistent with these earlier findings, sieve analysis results from the 2015 surface sediment samples also show Cottonwood Bay sediments are generally dominated by silts and clays (>97%). Grain size is generally uniform across depth, as the average grain size distribution in subsurface samples (> 0.5 ft in below the sediment surface) from the 2015 cores also was dominated by silts and clays (90%). Likely sources of sediment coming into Cottonwood Bay include Cottonwood Creek, surface drainage channels entering Cottonwood Bay, and overland runoff, as well as endogenous organic material (e.g., decayed algae). These data demonstrate that most of Cottonwood Bay is depositional, further supporting that Cottonwood Bay sediments are stable and will continue to accrue cleaner sediment.

**Potential for sediment resuspension**

Strong wind events could potentially cause turbulence strong enough to scour sediment in a shallow water body. To understand wind-driven sediment resuspension potential at the Site, a Tier 1 analysis of sediment transport processes, including sediment scour, was performed using methods described in Ziegler (2002). Wind data from the Grand Prairie Municipal Airport located approximately four miles southwest of Cottonwood Bay in Grand Prairie, Texas, was gathered from 2009 to 2014. These data indicate that winds are predominantly north-south in the Cottonwood Bay area. 75% of wind speed measurements were less than 16 kilometers per hour (km/hr), 98% of wind measurements were less than 32 km/hr, and 99.97% of measurements were less than 48 km/hr. Wave height and period were estimated using a wave forecasting diagram (Groen and Dorrestein 1976), and a wind-driven sediment scour evaluation was conducted using methods described in Ziegler (2002). Assuming a wind speed of 48 km/hr, which is predicted to occur only 0.03% of the time based on historical data, and water depths as shallow as 60 cm, sediment scour is calculated to be less than 1 mm.

Sediment resuspension is not expected during high flow events. Drainage channels entering Cottonwood Bay are baffled or contain structures that reduce flow prior to entering the bay, which reduces the potential for sediment scour. Potential scour during high flow events would be isolated to areas near storm water drainage outfalls. These outfall areas are relatively small with respect to all of East and West Cottonwood Bay, and potential erosion in the vicinity of the outfalls would have minimal impact on overall SWACs.

The geochronology, sediment grain size, and analysis of potential resuspension events all demonstrate that Cottonwood Bay is a stable and depositional environment.

**Surface sediment recovery**

To understand the ongoing recovery of surface sediments at Cottonwood Bay, temporal trends in surface sediment chemical concentrations and depth profiles were evaluated (Magar et al. 2009).
Surface sediment chemical concentrations over time

SWACs allow the comparison of surface sediment chemical concentrations on a large spatial scale using multiple sample locations, rather than comparing results from a single point location (ITRC 2014; Pelletier et al. 2019; USEPA 2005). For evaluation of uptake from sediment to fish and for comparison to background, the SWAC is the most appropriate statistical representation of the available data, because the SWAC accounts for spatial differences in sample density. In addition, sediment samples were not collected at the same locations during each sampling event, which prohibits direct point-based comparisons over time. SWAC comparisons are used to monitor surface sediment concentrations over time at various sediment sites (e.g., USEPA 2000, 2014b).

Due to the sparse and sometimes biased sampling during some of the pre-2014 sampling events, data were grouped into two periods for the SWAC calculations (see methods). The SWAC results in East Cottonwood Bay showed lower values for the later period than the earlier period for all the evaluated chemicals except cadmium and chromium (Table 1); however, the increases for cadmium and chromium are not statistically significant (p > .05) and are attributed to two distinct samples collected from East Cottonwood Bay during the 2011 sampling event (determined using Rosner’s outlier tests in ProUCL).

West Cottonwood Bay SWACs decreased from the first period to the second for all the evaluated chemicals (Table 1). The SWAC decreases are significant (p < .05) for chromium, copper, lead, zinc, and Aroclor 1260 in West Cottonwood Bay. Maps showing SWACs in Cottonwood Bay are included as Supplemental Information (Figure S1).

Contaminant concentration trends with sediment depth (vertical core profiles)

The recovery of surface sediments over time can also be demonstrated through the evaluation of vertical core profiles (Brenner et al. 2001, 2004; Magar et al. 2005b). If cleaner sediments are deposited over time and sediments are stable, then elevated concentrations of contaminants are buried beneath layers of cleaner material. Thus, sediments contaminants now present at depth represent past surface sediment contaminants (assuming negligible chemical transformation), and vertical trends provide another line of evidence for understanding temporal trends.

Sediment core profiles based on data collected in 2015 for six metals and two Aroclors show sediment concentrations with depth (Figure 3). Although concentrations of metals in Cottonwood Bay sediment do not pose unacceptable risks to human or ecological receptors (GSI 2016a), vertical profiles of metals are presented to support the evaluation of surface sediment recovery in Cottonwood Bay. The 2015 cores show the highest concentrations of metals and Aroclors at depth with decreasing concentrations toward the surface. Peak concentrations of metals and PCBs are typically located at an interval of 30 to 60 cm below the sediment surface at each Cottonwood Bay core location, with higher concentrations of some constituents at a depth of 60 to 90 cm below the sediment surface for the eastern West Cottonwood Bay core (WCB-101). Core profiles are not illustrated for Aroclor 1242 because the concentrations across all depths were similar to background. Concentrations in the western West Cottonwood Bay core (WCB-111) are low in each vertical profile due to its location, which is further from the source of contamination as compared to East Cottonwood Bay and eastern West Cottonwood Bay. Furthermore, clean incoming sediments from Cottonwood Creek, which flows into western West Cottonwood Bay, likely contribute to the low chemical concentrations in this area of the bay.
Decreasing trends in surface sediment contaminant concentrations over time along with the vertical profiles demonstrate historical and ongoing sediment recovery in Cottonwood Bay and isolation of buried contamination.

**Temporal trends in fish tissue PCB concentrations/human health risk reductions**

Temporal trends for contaminants in fish tissue are most appropriately evaluated using data from multiple years for the same fish species. Fish tissue data sets from 1994, 1995, 2008, and 2015 were used to evaluate trends in Cottonwood Bay and Mountain Creek Lake. These data sets were used to evaluate temporal trends in PCB concentrations for fish fillet samples (Figure 4) using lipid-normalized concentrations to control for potential confounding effects of temporal variation in fish lipid content. Data for PCBs are shown as total PCBs, based on the sum of Aroclors (1994 and 1995) or congeners (2008 and 2015), depending on the sampling event. The Texas Department of State Health Services has established a wet-weight PCB fish tissue health assessment comparison value of 0.047 mg/kg; lipid-normalized fish tissue PCB concentrations are not directly comparable to this goal but are used to reduce lipid-related variability when evaluating natural recovery trends.

The lipid-normalized PCB concentration data demonstrated that previously elevated lipid-normalized PCB concentrations (>10 mg/kg lipid) in common carp and largemouth bass from Cottonwood Bay and Mountain Creek Lake have decreased since the mid-1990s (Figure 4). The observed decrease from the mid-1990s to 2015 is supported by trend and nonlinear regression analysis; however, uncertainty in the analyses exists based on a limited
number of sampling years (n = 3–4) and the lack of lipid-normalized total PCB concentration data between 1995 and 2008. Another uncertainty lies in the difference in analytical method between the Aroclor analyses of the 1990s and the later congener analyses. The results of Aroclor analyses may be either greater or less than the sum of congeners for the same samples (e.g., Santini et al. 2015; Sather, Newman, and Ikonomou 2003); lacking split samples analyzed by the applicable laboratories using the applicable methods, direction of bias in this case (if any) is unknown.

Spatial differences in fish PCB concentrations have been observed, with fish captured in and near Cottonwood Bay tending to show higher PCB concentrations than those captured elsewhere in Mountain Creek Lake, although there are exceptions. The spatial coverage of fish tissue sampling has been generally comparable over time, limiting the importance of this variable as a confounding factor in the temporal trend analysis. Maps of fish tissue PCB concentrations over time are provided as Supplemental Information (Figure S2).

During the late 1990s most military operations at the naval air station property ceased and most of the remediation and control measures had been implemented, such that early reductions in fish tissue PCB concentrations likely reflect a combination of source control and natural recovery processes, with natural processes becoming the predominant recovery processes over time. Although concentrations do not differ much between the 2008 and 2015 sampling events, data from additional time points are needed (and will be collected) to further clarify recovery trends. Typically, an observed trend in fish tissue concentration lags compared to changes in surface sediment concentrations (Weston et al. 2002). Overall, the

![Figure 4. Lipid-Normalized Total PCB Concentrations over time. Bars represent the average lipid-normalized total PCB concentration in fillet skin off samples by fish species and sampling event. The hashed, white, gray, and black bars represent the 1994, 1995, 2008, and 2015 sampling events, respectively. Error bars represent 1 standard deviation. Sample counts (n) are shown beneath each bar.](image-url)
lipid-normalized data demonstrate that PCB concentrations in fish tissue have generally decreased since the mid-1990s, which is supported by the SWAC trends and vertical core PCB profiles.

**Fish tissue time to recovery**

An empirical BSAF of 1.5 for PCBs in largemouth bass within Cottonwood Bay was identified based on a sediment SWAC of 0.082 mg/kg dry weight and an average bass fillet PCB concentration (based on 43 congeners) of 0.12 mg/kg wet weight. Combining this BSAF and the TDSHS goal of 0.047 mg/kg in edible fish, the target sediment concentration to allow removal of the fish advisory for Largemouth Bass was estimated as 0.03 mg/kg. This approach assumes that the site-specific bioaccumulation factor (BAF) for total PCBs is constant over time and is independent of PCB concentrations in sediment. This assumption is conservative, as there is evidence that BAFs may decrease with decreasing sediment contaminant concentrations in systems where contaminant sorption to black carbon is significant (Cornelissen and Gustafsson 2005). Other factors that can cause bioaccumulation rates to change over time include changes in food web structure and variability in growth dilution.

The time required for the average largemouth bass PCB concentration in Cottonwood Bay to reach 0.047 mg/kg was estimated based on PCB concentrations in newly depositing sediment, the sediment deposition rate, and the depth of sediment mixing. Total PCB concentrations in newly depositing sediment were approximated as a uniform distribution of 0.0005 mg/kg to 0.0083 mg/kg dry weight, based on surface sediment concentrations in Cottonwood Creek (a Cottonwood Bay tributary). The incoming sediment total PCB data set will be updated for model refinement as data are collected during long-term monitoring.

The sediment deposition rate was represented as a uniform distribution of 0.6 to 1.9 cm/year, based on results representing approximately the last 18 years for each of the three geochronology cores. Site-specific sediment mixing depth information was inferred from the vertical distribution of excess Pb-210 in shallow (<10 cm) sediments from the geochronology cores. Under conditions of sediment accumulation without mixing, excess Pb-210 would be expected to be highest at the surface and decrease with depth. However, in two of the three sediment cores, excess Pb-210 in the shallow sediments was either constant with depth or increased slightly, indicating sediment mixing to depths of 0, 2.0, and 7.1 cm in the three cores, respectively. To be conservative, and considering non-site-specific data on the vertical distribution of organisms in lake sediments (USEPA 2015), we used a uniform distribution of 2.5 to 15 cm to represent sediment mixing depth for modeling purposes. These inputs yielded a median estimate of nine years (range = 2 to 19 years) for largemouth bass PCB tissue concentrations in Cottonwood Bay to decline to 0.047 mg/kg.

While largemouth bass were predicted to recover within a reasonable time, certain fish species were predicted to maintain fillet PCB concentrations greater than the TDSHS target for the foreseeable future. Specifically, the PCB concentrations estimated in newly depositing sediment (i.e., background PCB inputs) were predicted to result in ongoing fish consumption advisories for bottom-feeding species with high lipid content in edible fillets (e.g., carp, smallmouth buffalo).
The actual course of fish tissue PCB recovery over time will be monitored, with refinements to the recovery model to include characterization of PCBs in newly depositing sediments. If recovery does not proceed as expected, one possible explanation would be incomplete source control, with PCB uptake from surface water potentially playing a more important role than expected. Ghosh, Bokare, and Gobas (2021) demonstrated that surface water PCBs can contribute substantially to PCB bioaccumulation in fish, depending on food web structure and site-specific PCB mixture composition. Thus, investigations to understand possible remaining PCB sources and/or to develop a mechanistic understanding of site-specific PCB bioaccumulation could be considered if PCB concentrations in fish do not recover adequately over time. Depending on the outcome of such investigations, additional measures such as source control or thin cover placement may be warranted. This type of adaptive management planning is consistent with sediment risk management recommendations (Bridges and Gustavson 2014) and was an important contributor to the acceptability of MNR as a remedy for Cottonwood Bay.

Conclusions

Current risks at the Cottonwood Bay site are associated with human consumption of PCB-contaminated fish, and these risks are currently managed through institutional controls such as fish consumption advisories. Findings of the MNR evaluation for Cottonwood Bay sediments are summarized as follows. First, source control measures have eliminated or controlled sources of metals and PCBs contributing to affected sediments in Cottonwood Bay. Second, decreasing chemical trends in surface sediment concentrations over time confirmed ongoing surface sediment recovery in Cottonwood Bay. Third, geochronology evaluations verified that cleaner sediments are accumulating at a sufficient rate (0.6–1.9 cm/yr) to effectively isolate PCB-contaminated sediments at depth. Fourth, the presence of fine-grained sediments and modeling of negligible sediment scour through high wind or high flow events indicated minimal potential for sediment erosion and contributed to the understanding of Cottonwood Bay as a depositional environment. Fifth, results from sediment core sampling and analysis demonstrated the natural recovery of surface sediments through evidence that the highest metals and PCB concentrations in Cottonwood Bay are located at depth, covered by layers of cleaner sediments. Sixth, fish tissue concentration trends indicated reduced PCB concentrations in fish tissue from the 1990s through 2008, while changes in concentrations were negligible between 2008 and 2015. Finally, a time to recovery analysis predicted that PCB concentrations in edible tissue of most fish species would reach target levels within a reasonable period of time.

The results of the MNR evaluation and current PCB concentration trends in surface sediment and fish show that MNR is expected to be an effective and appropriate response action for Cottonwood Bay. Long-term monitoring is planned as part of the MNR response action that was approved by TCEQ in a Remediation Action Plan (RAP) (Ramboll 2018). Monitoring will be used to confirm that natural recovery processes are continuing, such that surface sediment and fish tissue PCB concentrations continue to decrease over time. The 20-year monitoring program for Cottonwood Bay includes the implementation of (a) bathymetric surveys to identify potential scour areas, (b) Bay-wide, surface-sediment composite sampling for PCB analysis, (c) sampling of incoming suspended sediment to evaluate incoming background conditions, and (d) fish fillet sampling for PCB analysis. Decreasing trends in surface sediment
and fish fillet PCB concentrations over time (if observed) will demonstrate the continued effectiveness of natural recovery processes and will further support estimates of the time required for PCB concentrations in fish fillet to decline enough for fish consumption advisories to be removed. If surface sediment and corresponding fish concentrations show ongoing reductions over time, but are not anticipated to achieve Site-specific remediation goals within a reasonable period, a contingency action may be implemented, which would include (a) additional investigations to determine why PCB concentrations are not adequately decreasing, such as the evaluation of possible remaining PCB sources or a further understanding of site-specific PCB bioaccumulation, and (b) if appropriate based on those findings, application of a thin cover (e.g., nominally 15 cm) to accelerate MNR in areas where decreasing PCB concentration trends are slowest.

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Availability of Data and Material
Data can be made available upon request or accessed at http://dx.doi.org/10.17632/rs2vx884cd.1.

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