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Modeling water and sediment contamination of Lake Pontchartrain following pump-out of Hurricane Katrina floodwater

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Abstract

Levee failure and overtopping as a result of Hurricane Katrina caused major flooding of New Orleans, Louisiana. Floodwaters, which were contaminated with heavy metals, organic chemicals, and fecal coliform bacteria (FCB), were pumped into neighboring Lake Pontchartrain during dewatering. The impact of levee failure on water and benthic sediment concentrations in the lake was investigated by applying a numerical water quality model coupled to a three-dimensional, numerical hydrodynamic model. The model was used to compute water and benthic sediment concentrations throughout the lake for lead, arsenic, benzo(a)pyrene (BaP), and 1,1-dichloro-2,2-bis(p-chlorophenyl)ethylene (DDE), and water concentrations for FCB. Computed concentrations resulting from actual pumped discharges with levee failure and overtopping were compared to computed concentrations resulting from pumped discharges without levee failure or overtopping, and concentrations from both sets of conditions were compared to ecological water and sediment quality screening guideline values. The model indicated that incremental increases above pre-Katrina benthic sediment concentrations are about a factor of 10 greater with dewatering of the floodwaters than with dewatering of storm water without flooding. However, these increases for the metals are small relative to pre-Katrina concentrations. The results showed that the ecological screening-level sediment quality guideline values were exceeded for BaP and DDE in areas near the south shoreline of the lake as a result of floodwater pump-out, whereas, this was not the case for storm water removal without flooding. The model showed that lake water column concentrations should be about the same during both dewatering conditions regardless of whether there is flooding or not.

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Keywords: Lake Pontchartrain; Hurricane Katrina; Floodwaters; Levee failures; Modeling; Contaminants; Sediments

1. Background

On August 29, 2005, the storm surge associated with Hurricane Katrina resulted in the failure and overtopping of levees protecting New Orleans. Floodwater was pumped out of the city into surrounding marshes and Lake Pontchartrain over a period of 37 days, introducing contaminants to the surrounding environment and raising environmental concerns. Trace metals, organic chemicals, and bacterial contamination were measured in New Orleans floodwater shortly after the hurricane and flood event (Pardue et al., 2006; Presley et al., 2006; US EPA, 2006). The Interagency Performance Evaluation Task Force (IPET) led by the US Army Corps of Engineers

was charged to study and understand the behavior of the New Orleans hurricane protection system in response to Katrina. The IPET was in part tasked with evaluating the socio-economic consequences of levee failures, including environmental consequences. This paper describes the numerical modeling for IPET to evaluate contamination of Lake Pontchartrain resulting from dewatering following Katrina as contrasted with dewatering without levee failures and overtopping.

2. Approach

The three-dimensional (3D) hydrodynamic model CH3D-WES (Johnson et al., 1991, 1993) and the 3D water quality model CE-QUAL-ICM (ICM) (Cercio and Cole, 1995) were used to model conditions in Lake Pontchartrain for a period of 90 days starting on September 1, 2005.

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Contaminant concentrations were modeled for two conditions: (1) dewatering of floodwaters for the “*actual*” conditions that occurred with levee breaches and overtopping; and (2) dewatering for conditions of the system performing as designed and without levee overtopping (“*baseline*” conditions). The *baseline* conditions served as a basis for comparison with the *actual* conditions. Both the *actual* and *baseline* conditions include the period during which pumps were operating to remove flood and rain water from the city following Hurricanes Katrina and Rita.

2.1. Hydrodynamic model

The *z*-plane version of CH3D-WES was used. This version has a varying number of layers with the total number along a water column depending on the depth. A plan view of the model computational grid is shown in Fig. 1. The grid contained 6038 computational cells in the surface layer and 21,018 cells in total over all layers. Each layer thickness was 1.52 m except the surface layer, which varied depending on the water surface elevation. The maximum number of layers was six for a maximum depth of about 9.14 m. A typical grid cell size in plan form was on the order of 300 m × 700 m. The grid was generated using a grid generation tool within the Surface Water Modeling System (<http://chl.erdc.usace.army.mil/chl.aspx?p=s&a=Software;4>). National Oceanographic and Atmospheric Administration (NOAA) navigation charts were used to develop the bathymetry with 5 ft intervals.

Although the CH3D-WES model includes baroclinic terms to simulate stratified flows resulting from water density differences caused by temperature and salinity, this feature was not activated for this study since the floodwaters had about the same salinity as the lake water in the vicinity of the pumped discharges. In the month following the storm, salinity measurements taken in the lake at

Station 4 (south shore at Pontchartrain Beach) by the Lake Pontchartrain Basin Foundation averaged 7.4 parts per thousand (ppt), whereas, salinity measurements taken in the New Orleans floodwaters averaged 5.6 ppt. Floodwater salinity was higher near the lake shore (ranging approximately from 7 to 9 ppt) and decreased moving south towards downtown away from the shore (ranging approximately from 2 to 6 ppt).

The hydrodynamic model was run for both the *actual* and *baseline* conditions, where the differences between the two conditions were the pumping rates and durations. Wind speed and direction from the New Orleans International Airport was applied to the model. Tide records from NOAA were used as boundary conditions at the open sea boundary, which was located at the Rigolets (see Fig. 1), the inlet between Lake Pontchartrain and Lake Borgne, which connects to the Gulf of Mexico. Additionally, the pumped flows were applied as boundary conditions along the south shore and were a major forcing function for the lake hydrodynamics during *actual* dewatering. Freshwater stream flows entering the lake were not included in the model since most streams enter along the north shore and are a minor forcing component compared with wind and tidal forcing and pumped flows, except when Mississippi River floodwaters are diverted through the Bonne Carre Spillway.

The hydrodynamic model used the estimating pumping rates obtained from IPET task team 8 (Pumping Station Performance) for *actual* conditions of post Katrina, including the pumping during and following Hurricane Rita. These data show that the pumps started operating on September 11, 2005, and ended on October 17 for pump-out to the lake. Obtaining pumping rates for the *baseline* conditions was more problematic. Other IPET tasks will estimate the pumping rates for conditions of the levees performing as designed, but this information was not available at the time of this study. In the absence of these

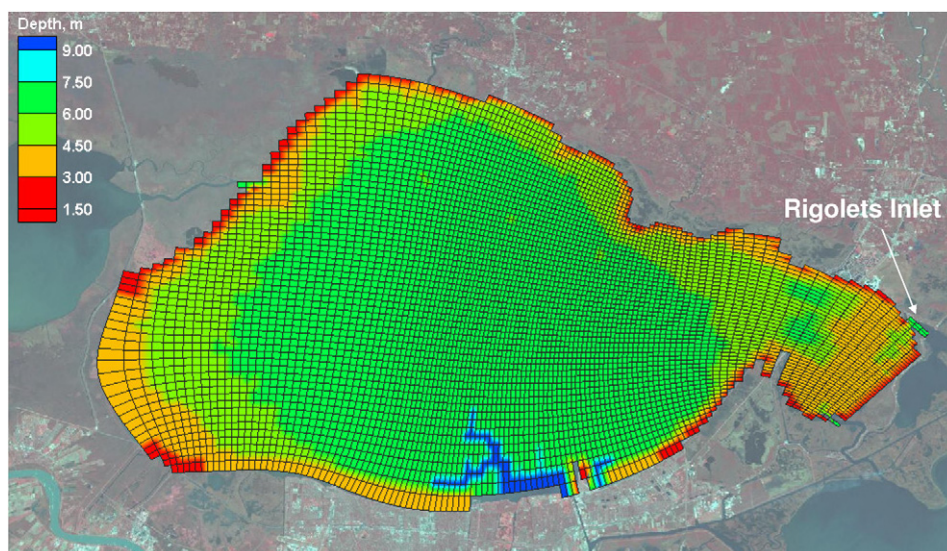


Fig. 1. Model grid for Lake Pontchartrain with depth contours.

data, assumptions were used to establish a *baseline* condition. The rainfall amounts recorded for Katrina and Rita were multiplied by the approximate rainfall collection area to produce rainfall volumes. The known pump flow capacities were divided into the rainfall volumes to estimate the time required to pump out the rainfall. The capacity of each pump was used with the duration required to dewater the rainfall to establish the *baseline* pumped flows. This approach does not include water that overtopped the levees. Thus, this approach assumes that a levee system was in place that fully protected New Orleans from any storm surge flooding by these two hurricanes.

2.2. Water quality model

The water quality model was originally developed during an eutrophication model study of Chesapeake Bay (Cerco and Cole, 1993), but has subsequently been applied to other systems in the US (Cerco et al., 1994, 1999, 2000) and the San Juan Bay and Estuary in Puerto Rico (Bunch et al., 2000). The version of the model used for Lake Pontchartrain was based on a recent version used for Lake Washington, WA (Cerco et al., 2004, 2006; Kim et al., 2006).

During the Lake Washington model study, the ICM model code was modified for simulating a pathogen and two toxicants. The two toxicants were coded in a generalized format and were intended to provide a rapid screening tool and a foundation for more detailed formulations that could be added later. The two toxicants are subject to identical processes including degradation, volatilization, partitioning to solids, settling, and burial. A mass balance of toxicant within the water column and sediment bed is maintained. Toxicant 1, which can represent a metal, partitions to inorganic solids, and toxicant 2, which represents an organic chemical, partitions to particulate organic carbon (POC). Degradation can occur in both the water column and sediments. Dissolved toxicant diffuses in either direction across the sediment–water interface. Linear, reversible, equilibrium partitioning to solids is assumed, and the fractions of dissolved and particulate toxicant are computed accordingly. The sediment bed for toxicant mass balance is treated as a single, well-mixed layer. Degradation in the bed is applied to the bulk (total) contaminant concentration. Degradation rates and partition coefficients within the sediments may vary from those specified for the water column. The mass balance and partitioning equations for toxicants within the water column and sediment are presented by Cerco et al. (2004).

The water quality model used the same grid resolution as the hydrodynamic model of Lake Pontchartrain. The hydrodynamic model was executed, flow fields were saved, and these data were subsequently used to provide flow fields required by the water quality model.

The water quality model was applied for five constituents, arsenic (As), lead (Pb), benzo(a)pyrene (BaP), 1, 1-dichloro-2,2-bis(*p*-chlorophenyl)ethylene (DDE), which

is a degradation product of DDT, and fecal coliform bacteria (FCB). Chemicals from various classes (e.g., pesticides, polycyclic aromatic hydrocarbons, and metals) were selected to be representative of other compounds. The chemicals selected for the modeling effort were identified from existing analytical chemistry data for soils and waters in New Orleans during the flooding event. Chemicals were selected based on their frequency and magnitude of detection, association with suspended particles, and potential to cause adverse ecological impacts. The model state variable for each contaminant was treated as total concentration (i.e., dissolved and particulate) with fractions of dissolved and particulate calculated from equilibrium partitioning to suspended solids and sediment bed solids. Fate processes of sorption to solids, settling of particulate contaminant, volatilization of dissolved contaminant, and die-off of FCB were modeled. Degradation of the organic chemicals was ignored given the relatively short simulation period and long half-life of these compounds in natural waters. These constituents provide a wide range in adsorption behavior for inorganic and organic contaminants with arsenic having a relatively low sorption distribution coefficient and DDE having a high value.

The water quality model requires constituent loading (mass/time) for each effluent location. Loading is the product of discharge rate (volume/time) and concentration (mass/volume) of the effluent. With pumping rates given, the concentrations of the effluent had to be determined for both scenarios. Fortunately, for the *actual* scenario, many floodwater and flood sediment samples had been collected and analyzed for a host of contaminants. These data were collected by the US Environmental Protection Agency (EPA), Louisiana State University (LSU), US Geological Survey (USGS), and Louisiana Department of Environmental Quality (DEQ) and were assembled into an EPA database (US EPA, 2006). Other sources of data included peer-reviewed publications of studies reporting similar information (Pardue et al., 2006; Presley et al., 2006). Data for the constituents assessed in this study were assembled in a database and analyzed to yield a median concentration and a 95% upper confidence limit (95UCL) concentration for each constituent for three different areas of interest, Orleans Parish Metro, Orleans Parish East, and St. Bernard Parish. The first two areas involve floodwaters that were pumped into the lake, and the latter represents floodwaters that were pumped into Violet Marsh. Given the extensive difficulty in trying to sort out which specific sub-areas of floodwater (and associated sample concentrations) to assign to each pump, the decision was made to process all measured values within each of the three broad areas to produce a single median and 95UCL concentration to use for the pumped effluents of that area. Any measurements that were below detection limit were set equal to 1/2 the detection limit rather than zero for the analysis. In summary, measured concentrations were used to establish the pumped loadings discharged into the lake

for the *actual* conditions for Orleans Metro and Orleans East, and the concentrations were held constant for the duration of the pumping to establish those loadings.

The original plan for estimating pumped concentrations for the *baseline* conditions was to use data from a previous pumping event that endured rainfall of the amount that fell during Katrina. However, no such observed data could be found for the constituents being studied. [Pardue et al. \(2006\)](#) stated that the metal concentrations in the Katrina floodwaters of New Orleans were typical of storm water with a few exceptions of elevated concentrations of lead. [Jin et al. \(2004\)](#) reported FCB concentrations on the order of 40,000 MPN/100 mL measured during 1998 in pumped storm water in drainage canals that empty into the lake. This concentration is within the range of values measured in the Katrina floodwaters. Given these previous findings and the lack of any better information, the assumption was made that the pumped concentrations for the *baseline* conditions would be the same as those for the *actual* conditions. However, the total constituent mass discharged for the *baseline* conditions is considerably less since the pumps operate for a much shorter period of time to remove far less water.

Output from the 3D lake water quality model consists of time-varying concentrations in the water column for each computational cell and time-varying sediment concentrations in the benthic sediment layer beneath each bottom water cell. Such a large amount of data requires reduction to render presentations that are useful for interpretation. Two-dimensional contour plots of maximum concentrations in the water surface layer and benthic sediment layer were used to collapse data for manageable interpretation.

3. Model input

3.1. Parameters and basic data

In addition to boundary conditions and model control variables, several parameters and basic data are required to apply the lake model, which include:

- total suspended solids (TSS) (mg/L),
- TSS settling rate (m/day),
- FCB die-off rate (day^{-1}),
- fraction of total organic carbon (TOC) to total sediment dry weight, f_{oc} , for the water column and sediment bed,
- sedimentation variables, either burial rate or resuspension rate (m/day),
- benthic surficial layer porosity,
- sediment–water sorption distribution coefficient, K_d (L/kg).

TSS was calculated from turbidity using a regression developed from data collected from the New Orleans Inner Harbor Navigation Channel (IHNC) as part of the study ([Estes et al., 2006](#)) of the Corps Dredged Material Management Units (DMMU). Abundant turbidity data

existed for both the floodwaters and the lake. TSS settling rate was set to 1 m/day based on experience in modeling similar systems and particle settling studies conducted for the IHNC. The total coliform bacterial die-off rate is typically around 1.0 day^{-1} ([Thomann and Mueller, 1987](#)) for freshwater and 1.4 day^{-1} for seawater. [Jin et al. \(2004\)](#) measured FCB die rates of approximately 2.8 day^{-1} in the lake. The relatively conservative value of 1.0 day^{-1} was used in the model. Data from the DMMU studies indicated f_{oc} values of about 0.02. Resuspension was assumed to be zero; using the settling rate, TSS, and sediment porosity to compute sediment bulk density, a burial rate of 0.026 m/yr was calculated based on a steady-state solids balance. Benthic surficial layer porosity was assumed to be 0.9, which is a value typical of most unconsolidated surficial sediments. Estimates of K_d for arsenic and lead were obtained from the literature and consisted of 300 and 4000 L/kg, respectively. The organic carbon–water partition coefficient, K_{oc} , was computed for the organic constituents BaP and DDE using the relationship ([Karickhoff et al., 1979](#)):

$$K_{oc} = 0.6 K_{ow}, \quad (1)$$

where K_{ow} is the octanol to water partition coefficient for organic chemicals. A database of chemical properties within the Risk Assessment Information System (RAIS) (<http://risk.lsd.ornl.gov/>) was used to obtain values of K_{ow} for BaP and DDE of $1.0 \text{ E}6$ and $3.24 \text{ E}6$, respectively. The K_d for organic chemicals is usually the product of K_{oc} and f_{oc} when total solids are used to partition, but in the case of the ICM model, inorganic suspended solids (ISS) and suspended POC are used for inorganic and organic contaminants, respectively, thus, care must be taken in defining K_d for use in ICM as explained below. The POC concentration is the product of f_{oc} and TSS for the water column and the product of f_{oc} and sediment bulk density for the sediment bed.

Since K_d is an important parameter that can be affected by ambient conditions and can vary from system to system for the same chemical, testing was conducted with the RECOVERY model ([Ruiz and Gerald, 2001](#); [Boyer et al., 1994](#)) to validate the selected values for K_d and K_{ow} . RECOVERY is a time-varying model that treats the water column as a single, fully mixed cell of known area, depth, and flushing rate and represents the bottom sediments as a series of layers over the vertical dimension. Thus, this model, like the lake model, produces time-varying concentrations for the water column and bottom sediments. The model assumes a constant flushing rate or flow through the system. RECOVERY was run to quasi-steady-state assuming no settling, resuspension, degradation, volatilization, or flushing (i.e., a large volume with very small through-flow, thus, a very large residence time); only equilibrium sediment–water partitioning was included. Sediment concentrations measured from the floodwaters were input to the model, and overlying equilibrium water concentrations were computed by the

model and compared with measured water values taken concurrent with the sediment measurements in the floodwaters. The model indicated that the value of 4000 L/kg for lead K_d was representative of conditions in the New Orleans floodwaters. However, the value of K_d for arsenic had to be adjusted slightly to 500 L/kg to match observed water concentration. The K_{ow} for BaP and DDE also had to be decreased to 0.5 E6 and 1.0 E6 L/kg, respectively, to match observations. Since Eq. (1) is programmed into the RECOVERY model code, and f_{oc} is a measured input variable, it was easier and more rational to adjust K_{ow} for BaP and DDE. More than likely, adsorption to dissolved organic carbon (DOC), which is manifested as partitioning to water since DOC sorption is not included in the model, is the reason that K_{ow} had to be decreased to match observations. In reality, K_{ow} is a chemical property that should not require adjustment if DOC partitioning is included. These tests with RECOVERY resulted in relatively minor model adjustments that gave increased confidence in the modeled sorption process.

3.2. Hydrodynamic model boundary conditions

The hydrodynamic model was started with quiescent conditions on September 1, 2005, and run for 90 days applying boundary conditions for winds, water levels at the Rigolets Inlet, and pumped discharges from New Orleans. Wind velocities were based on observations from the New Orleans International Airport. However, wind data were not available until September 7 due to Katrina, so the winds applied to the model between September 1 and 7 were linearly ramped from 0.0 to the values observed on September 7. Thus, the period between September 1 and 7 were considered a model spin-up period and were not be used for model analysis and comparisons.

The water level boundary condition at the Rigolets Inlet included both meteorological and astronomical tide. For the astronomical tide, hourly predictions from NOAA at Waveland (station number 8747766) were used. There is a predicted tide station at Long Point, LA, in Lake Borgne, which is closer to the Rigolets entrance, but only high and low tides are available for that location. Measured water levels were available for the Waveland gage prior to Hurricane Katrina, but the gage was destroyed during the hurricane. Thus, predicted hourly tides at Waveland were used. The sub-tidal signal was obtained from water level recordings at East Bank 1 gage, Norco, Bayou LaBranche, LA (gage number 8762372) by using a 48-h moving average to filter out higher frequency signals. The sub-tidal signal was moved forward by 24 h and added to the astronomical tidal signal for Waveland to form the boundary condition at the Rigolets Inlet.

The pumped discharges and loadings to the lake were separated into Orleans Metro and Orleans East. Orleans Metro includes all the pump stations in Orleans Parish that are west of the IHNC and that pump water into the lake or into canals that empty into the lake, whereas, Orleans East

includes all the pump stations in Orleans Parish east of the IHNC that pump into the lake. Records indicate that Jefferson Parish pumps that discharge into the lake were not operated. Fig. 2 shows time-series plots of the combined, estimated pumping rates into the lake for Orleans Metro and Orleans East following Katrina (*actual* conditions). The flows for each pump station have been combined for the two areas for presentation, but the flow for each pump constituted a separate time series of discharge input for the model. The pump stations included in the model are shown in Fig. 3 with the exception of two temporary pumps in Orleans East that pumped into the lake. Since the lake model grid did not include canals that are connected to the lake, the discharges of any pump stations that are not located on the shoreline were assumed to be located at the confluence of the lake and the canal they pump into. Pumped discharges were used for inflows to the hydrodynamic model and for calculating loading inputs for the water quality model. The combined, time-integrated pumped flows of Fig. 2 represent a pumped volume of 0.54 E9 m³, which is approximately 5–10% of the lake volume.

As discussed in the Section 2, rainfall and pump capacities were used to establish the *baseline* pumping conditions. The rainfall reported at Slidell, LA, was approximately 20 cm for Katrina; other gages in the area did not report. The rainfall reported at the New Orleans International Airport for Rita was 5.84 cm. Given the approximate, combined collection basin area for Orleans Metro and Orleans East of 2.8 E8 m², the rainfall volumes for Katrina and Rita were about 5.7 E7 and 1.7 E7 m³, respectively, or roughly 1/10 of the actual pumped volumes. With the Orleans Parish pump-to-lake total discharge capacity of approximately 1076 m³/s, rainwater could have been pumped out in less than a day following each hurricane. Of course, this pumping period assumes no overtopping of the levees. The pump-out time for Katrina under these assumptions is about 0.6 days. The capacity of

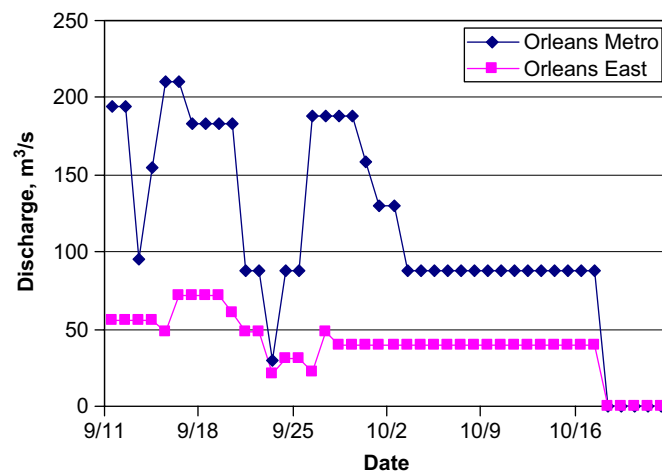


Fig. 2. Time series of combined *actual* pump discharges for Orleans Metro and Orleans East that were introduced into the lake.

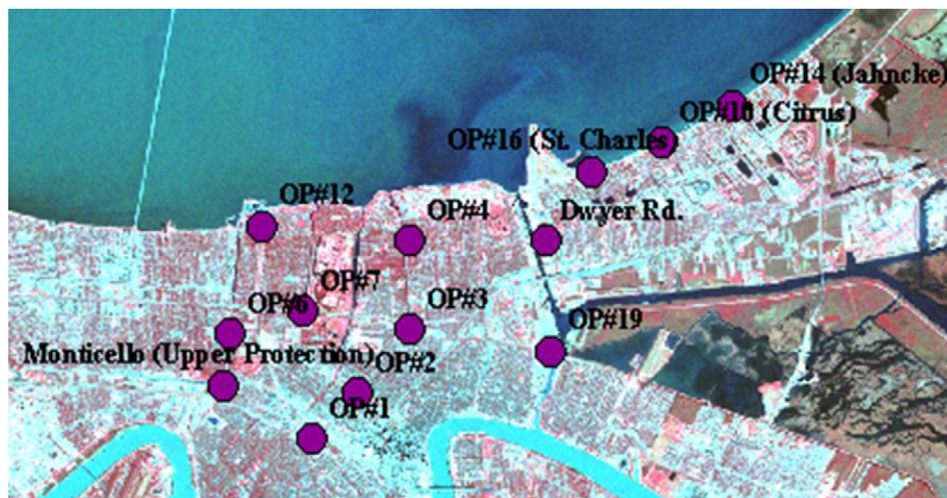


Fig. 3. Locations of pump stations for Orleans Parish included in the model that pumps into the lake.

each pump is known, and it was assumed that each pump would have been run at capacity for the *baseline* condition. Examination of pump records during tropical storm Isadora in September 2002 indicated this was a reasonable assumption. However, the model can accept only daily inputs for flows and loads (i.e., flow and load is assumed constant over each day, but can change from day to day). Therefore, the pump capacities were adjusted to provide a daily flow equal to the amount of rainwater to be emptied. For example, if a pump capacity is $100 \text{ m}^3/\text{s}$, then the flow used for the model pump was $60 \text{ m}^3/\text{s}$ based on the Katrina pump-out time of 0.6 days.

3.3. Water quality model inputs

The ICM model includes ISS and suspended POC as modeled state variables, rather than TSS. The current version of the model allows simulation of one inorganic contaminant that sorbs to ISS and one organic contaminant that sorbs to POC. As discussed earlier, TSS was estimated based on turbidity measurements, but there were no data for ISS and POC that are needed by the model for simulating the fate of particulate contaminants in the lake. POC constitutes about 40% of the volatile suspended solids (VSS), where VSS represents suspended particulate organic matter. It was possible to estimate ISS using estimated TSS and measured f_{oc} data and recognizing that TOC is primarily made up of POC, thus

$$\text{TSS} = \text{ISS} + \text{VSS} = \text{ISS} + 2.5f_{oc}\text{TSS}. \quad (2)$$

Rearranging Eq. (2) yields

$$\text{ISS} = \text{TSS}(1 - 2.5f_{oc}). \quad (3)$$

POC is calculated from

$$\text{POC} = f_{oc}\text{TSS}. \quad (4)$$

Concentrations of ISS and POC were held constant as background values throughout the lake by setting the

initial conditions and all boundary conditions to the same constant values.

The fraction of particulate inorganic contaminant concentration to total inorganic contaminant concentration can be determined through reversible equilibrium partitioning from

$$F_{pi} = \frac{K_d \text{ISS}}{1 + K_d \text{ISS}}. \quad (5)$$

Likewise, the fraction of particulate organic contaminant concentration to total organic contaminant concentration can be determined from

$$F_{po} = \frac{K_d \text{POC}}{1 + K_d \text{POC}}. \quad (6)$$

The units for K_d in ICM are m^3/g , thus, the values presented previously were converted from L/kg to m^3/g by multiplying by $1.0 \text{ E-}6$. The K_d values input for lead (Pb) and arsenic (As) were $0.5 \text{ E-}3$ and $4.0 \text{ E-}3 \text{ m}^3/\text{g}$, respectively. For ICM, K_d and K_{oc} are operationally the same since f_{oc} is taken into account by using POC instead of TSS to compute F_{po} . Thus, the K_d values for BaP and DDE (after applying Eq. (1) to get K_{oc}) used in ICM were 0.3 and $0.6 \text{ m}^3/\text{g}$, respectively.

The ICM model requires input of the volatilization rate (K_{vol} , m/day) rather than computing it from chemical properties, wind, and hydrodynamic flow conditions. Wind is the predominant forcing rather than flow for volatilization in lakes, thus, volatilization rate was computed based on wind speed. Using a wind speed of 5 miles per hour (3 m/s), Henry's law constants for BaP and DDE of $4.5 \text{ E-}7$ and $4.0 \text{ E-}5 \text{ atm m}^3/\text{mole}$, respectively, and respective molecular weights of 252 and $318 \text{ g}/\text{mole}$, K_{vol} was computed using the algorithm within the RECOVERY model. Volatilization within RECOVERY is based on the Whitman two-film theory as described by Chapra (1977), where the gas and liquid side mass transfer rates are computed from wind speed and molecular weight based

upon relationships presented by Mills et al. (1982). The resulting values of K_{vol} computed for BaP and DDE were 0.005 and 0.19 m/day, respectively. The ICM model multiplies K_{vol} by the dissolved organic chemical concentration in the surface layer of the water column to calculate the volatilization flux ($\text{g}/\text{m}^2/\text{day}$). The dissolved organic chemical concentration is the product of the total organic chemical concentration and the quantity $(1-F_{po})$.

Turbidity measurements for Lake Pontchartrain are routinely measured. Lake turbidity values obtained during the fall of 2005 following Katrina were analyzed over time and for all recording stations to obtain a lake-wide median value of 8.2 nephelometric turbidity units (NTU) (LPBF, 2006; Pardue et al., 2006). The lake median turbidity was converted to a median TSS value of 19.2 mg/L for use in the lake model for background suspended sediment. Benthic sediment resuspension rate was set to zero, and the burial rate was set to 0.026 m/yr in the model; the surficial sediment bed layer thickness was set to 0.2 m. Degradation rates were set to zero for all constituents except FCB.

The concentrations used to establish the lake water quality model loadings are shown in Table 1. The loading concentrations were categorized by median and 95UCL, which were determined from statistical analysis of all the post-Katrina floodwater measurements taken in the two areas (Orleans Metro and Orleans East) using the database compiled as described earlier. For some cases, the median and 95UCL are the same due to lack of variability in the measured concentrations. Loading concentrations for *baseline* conditions were set equal to those for *actual* conditions.

4. Calibration/validation

Given the observational data limitations, it was not possible to calibrate and validate the lake hydrodynamic

and water quality models to the extent normally desired. The flow fields and transport computed with these models have been found to be quite accurate in other studies if sufficient boundary conditions for inflows, water levels, winds, and constituent loadings are provided. A limited level of model calibration/validation was undertaken using water surface elevations and FCB concentrations measured during the later summer/fall of 2005. The hydrodynamic model was executed for *actual* conditions following Katrina so the model could be calibrated against observed water surface elevations in the lake. The water quality model was applied for FCB during *actual* conditions following Katrina to validate the model against observed FCB in the lake using the calibrated hydrodynamic model output.

The computed and observed water surface elevations during September and early October 2005 at the Norco gage of Lake Pontchartrain are shown in Fig. 4. This was the only water level observation gage available in Lake Pontchartrain for model comparison, and this gage was not reported between October 10 and December 2, 2005. The model compares closely with the observations throughout the observation period with the exception of the first 4 days, which was the model spin-up period when the model was started with quiescent conditions at mean sea-level elevation. The large spike in water level around September 24 was due to Hurricane Rita. The model performs exceptionally well given that the boundary conditions at the open sea boundary were synthesized from the combination of predicted astronomical tides and filtered sub-tidal meteorological forcing. Measured water levels at the seaward boundary are often available for estuarine and coastal hydrodynamic model applications. It was not necessary to make any adjustments in model parameters, such as bottom roughness and wind drag coefficient.

Pumped flows were a dominant factor in the lake currents under the *actual* conditions as can be seen in Fig. 5, which shows the surface layer velocity vectors computed for September 12, 2005, near the end of the day. Animation of the currents showed that the speeds increase and decrease dramatically near the south shore when pumping begins and ends.

The lake water quality model output for FCB with *actual* conditions was compared to lake measurements of FCB obtained by the Lake Pontchartrain Basin Foundation (LPBF, 2006). The Foundation's water sampling station locations are shown in Fig. 6. Model and observed concentrations versus time are plotted in Fig. 7 for stations 1–4 where data were available during the fall of 2005 along the south shore. Data for station 5 were not available during those months, and data at stations along the north shore were not included since the model did not include any FCB loadings from the north shore.

The observed data in Fig. 7 are less than ideal. There are no observations for September when the highest loadings and greatest computed concentrations occur. Model loadings end on October 18 when pump-out was

Table 1
Lake loading concentrations (total) by region for *baseline* and *actual* conditions

Constituent	Median ($\mu\text{g}/\text{L}$)	95UCL ($\mu\text{g}/\text{L}$)
<i>Orleans Metro</i>		
Arsenic	20	20
BaP	5	5
DDE	0.05	0.05
Lead	5	44
Fecal coliform bacteria	2200 ^a	70,041 ^a
<i>Orleans East</i>		
Arsenic	20	26
BaP	5	5
DDE	0.05	0.38
Lead	2.5	12
Fecal coliform bacteria	200 ^a	32,869 ^a

^aUnits are cfu/100 mL or MPN/100 mL.

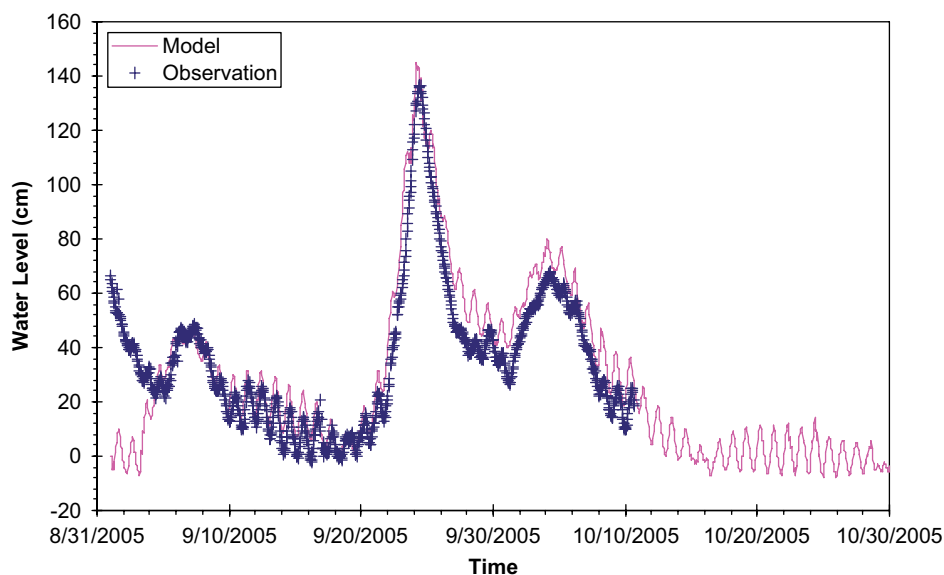


Fig. 4. Computed and observed water level in Lake Pontchartrain for tide gage 8762372 East Bank 1, Norco, Bayou LaBranche, LA.

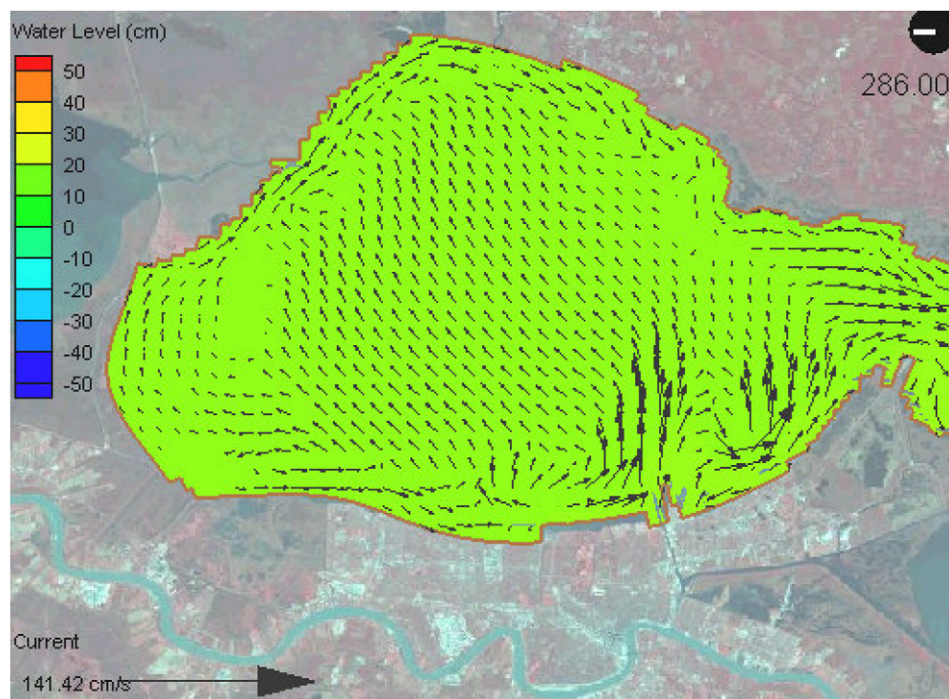


Fig. 5. Computed surface layer currents at the end of September 12, 2005, *actual* conditions.

completed, but observations indicate that there must have been other source loadings into the lake after that date that were not accounted for in the model. Thus, there is a window of only about 18 days in early October with which to meaningfully compare model and observations. During that window, the model can be compared with the observations. Comparisons after October 18 should be disregarded since there are no model loadings after that date. There is not enough information in Fig. 7 to evaluate model validation. It should be noted that the model loading concentration was constant over time and equal to

the median concentrations in the floodwaters, whereas the actual loading concentrations probably varied due to variations in pumped floodwater concentration over time and space.

The USGS collected lake sediment samples near the south shore and the causeway during September and October 2005 that were analyzed for a host of constituents (Demas, 2006¹), including those modeled. Measured BaP

¹Demas, C.R., 2006. Personal communication. US Geological Survey Louisiana Water Science Center, Baton Rouge, LA.

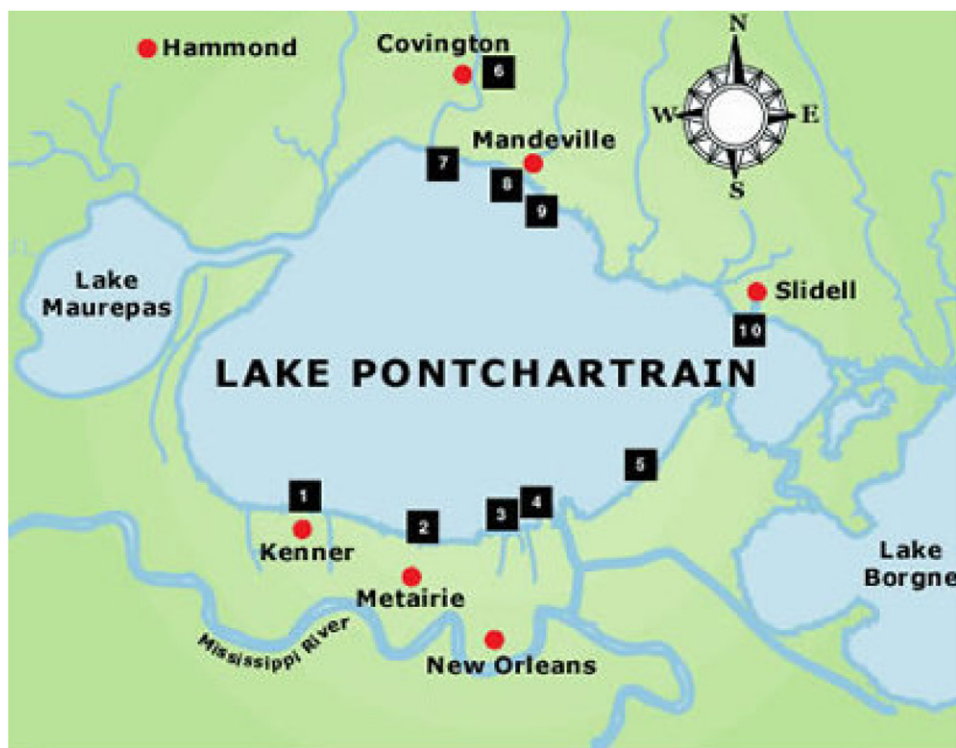


Fig. 6. Lake Pontchartrain Basin Foundation water quality sampling station locations (from Lake Pontchartrain Basin Foundation, http://www.saveourlake.org/water_quality.asp).

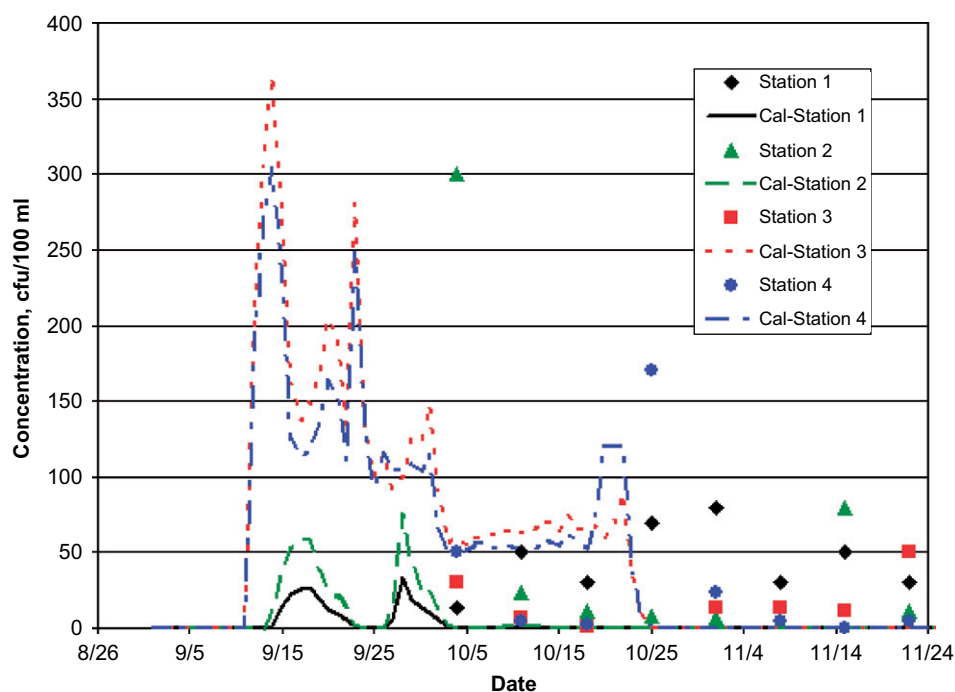


Fig. 7. Model-computed (Cal-Station and solid lines) FCB concentrations for *actual* conditions and median loading concentration and measured (symbols) FCB concentrations following Katrina at four stations along the south shore of Lake Pontchartrain.

ranged between 17 and 290 $\mu\text{g}/\text{kg}$, where model computations for sediment BaP with *actual* conditions were approximately 40 $\mu\text{g}/\text{kg}$ in the sampled area of the lake. The fact that the model result falls within the range of

observations for this chemical increases the level of confidence in the model. Computed sediment concentrations for DDE were also about the same order of magnitude as those measured. However, computed

sediment concentrations for As and Pb were two-orders of magnitude less than measured. This result is not surprising since only two loading events (Katrina and Rita) were included in the model. The prototype sediments have experienced many loading events over the years, and metals do not degrade; thus, the sediments can have a long memory for metal contaminant loading events.

5. Scenario results

5.1. Comparison of actual versus baseline conditions

The lake water quality model was executed for both scenarios (*baseline* and *actual* hydrodynamic conditions and loadings) using median and 95UCL loading concentrations. Computed maximum concentrations for the model surface layer and for the surficial benthic sediments were determined for each computational cell and saved for plotting. Example plots are shown in Figs. 8 and 9 for As

for both *actual* and *baseline* conditions, respectively. In general, the outermost extent of the maximum concentration contours for *actual* conditions extend further out from the shore and cover a larger area, whereas the *baseline* contours are more compact. Also, the outermost contours have higher concentrations for the *actual* conditions. The greater spread is due to the longer duration of pumping and the overall greater total mass loading of the *actual* conditions associated with the larger water volumes pumped. However, the pump discharge rates of the *baseline* conditions are near pump capacity, which results in a larger flow rate and mass loading rate, but for a much shorter duration. The short-term bursts of higher loading rates of the *baseline* conditions result in slightly higher overall maximum concentrations near the shore. The highest maximum sediment concentrations tend to be concentrated along the southeast shore of the lake, out from Orleans East, for both conditions as shown in Figs. 10 and 11 for As. This is due to the proximity to

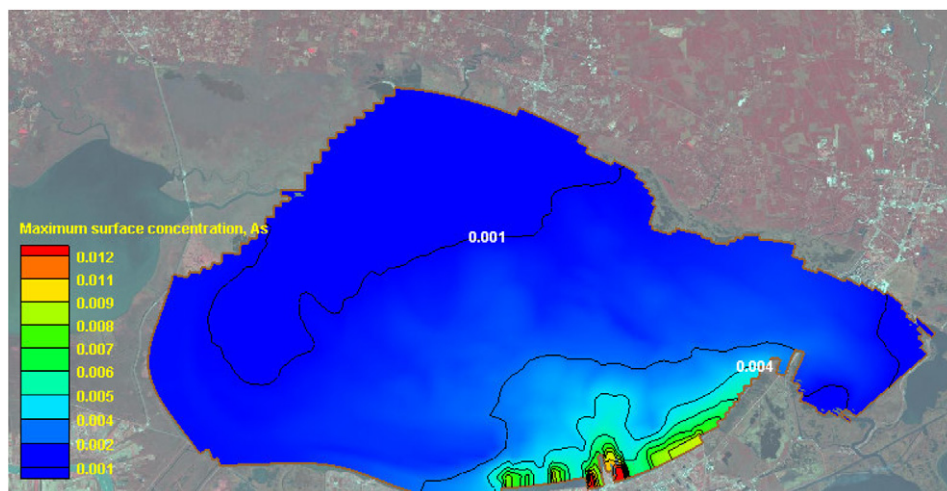


Fig. 8. Computed incremental maximum As water surface concentrations (mg/L total) in Lake Pontchartrain for *actual* conditions, median loading concentrations.

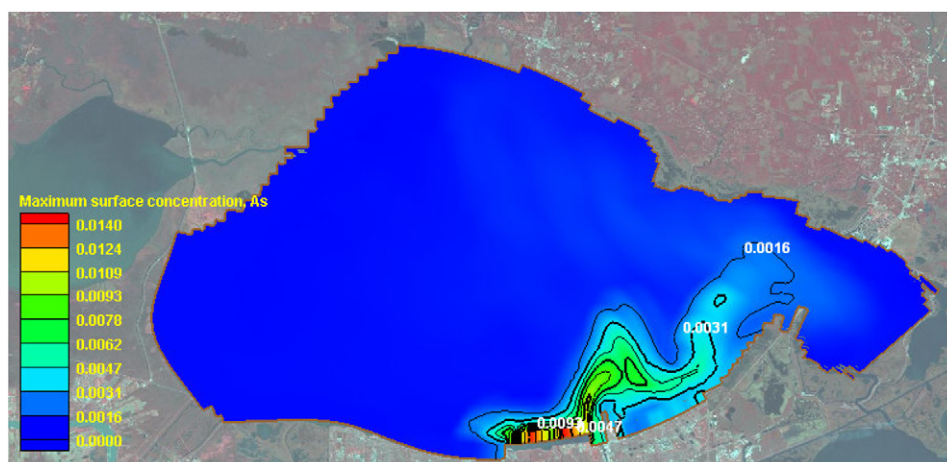


Fig. 9. Computed incremental maximum As water surface concentrations (mg/L total) in Lake Pontchartrain for *baseline* conditions, median loading concentrations.

pumps operating near capacity) which results in less time for settling of particulate matter and die-off for FCB. However, the maximum change in sediment concentrations for the *baseline* condition are roughly an order of magnitude less than those of the corresponding *actual* condition for all constituents, which is due to the much lower mass loading for the *baseline* condition.

5.2. Comparisons to ecological screening values

Dissolved water concentrations were needed for comparison to water quality criteria and standards, which are stated as dissolved. Dissolved concentrations were obtained by multiplying the fraction of dissolved to total contaminant concentrations in the water column (F_{dw}) times the maximum total concentrations in water in Table 2. The dissolved concentrations are reported in Table 3 for each constituent along with the dissolved fractions.

Computed maximum dissolved water column concentrations and maximum sediment concentrations were compared with ecological water and sediment quality screening values in Table 4. US Environmental Protection Agency (US EPA, 1986) recommends primary contact protective limits for FCB of 400 MPN/100 mL for a single sample. The computed maximum water concentrations for As and BaP were less than the EPA water quality criteria and LA

water quality standards for both conditions and both loading concentrations. The computed maximum water column concentrations for Pb exceeded EPA and LA water quality standards for both conditions with the 95UCL loading concentrations, and the concentrations for DDE exceeded the LA criteria for the *actual* conditions with 95UCL loading concentrations. Maximum concentrations for FCB in water exceeded EPA criteria for both conditions and both loading concentrations, but FCB exceedence in the lake frequently occurs during dewatering of New Orleans storm water (Jin et al., 2004).

The computed maximum sediment concentrations for As and Pb were less than the sediment quality guideline values for both conditions and both loading concentrations. The computed maximum sediment concentrations for BaP and DDE were less than the sediment screening criteria for both loading concentrations of the *baseline* conditions, but sediment concentrations for both constituents exceeded the guideline values for both loading concentrations under the *actual* conditions.

6. Discussion

Considering the uncertainties in the loading concentrations for *actual* and *baseline* conditions, one cannot conclude that dewatering will definitely result in specific concentrations or criteria exceedence. Model results should be viewed as relative, not absolute. One of the strengths of models rests in the ability to examine incremental changes and make relative comparisons.

It should be recognized that incremental increases in sediment concentration for some constituents due to the Katrina–Rita loading events may be miniscule relative to concentrations already in the prototype sediments resulting from years of accumulation from previous events. The increase in lake sediment metal concentrations computed by the model for *actual* conditions are very small compared with pre-Katrina concentrations. Pre-Katrina mean sediment concentrations reported by Penland et al. (2002) were 7.02 parts per million (ppm) and 17.5 ppm for As and Pb, respectively; thus, the maximum increase in sediment concentrations above pre-Katrina (background) values computed by the model for *actual* conditions and median loading concentrations was 0.7% for As and 0.3% for Pb. Limited sampling near the south shore of the lake following Katrina revealed that sediment concentrations for As and Pb were about the same as pre-Katrina values, with As values of 3–11 ppm and Pb values of 12–33 ppm (Demas, 2006; see footnote 1), thus, confirming the model that showed dewatering of New Orleans following Hurricanes Katrina and Rita had little impact on lake sediment concentrations of metals. Pre-Katrina sediment concentrations for BaP and DDE were not found, but post-Katrina concentrations of 17–290 ppb for BaP and non-detection—5.3 ppb for DDE were measured (Demas, 2006; see footnote 1). Sediment concentrations computed by the model for *actual* conditions and median loading concen-

Table 3

Dissolved fractions in the water column (F_{dw}) for each constituent and computed maximum water ($\mu\text{g/L}$) concentrations (dissolved) for *actual* and *baseline* conditions and median and 95UCL loading concentrations

Condition	As	BaP	DDE	Pb
F_{dw}	0.99	0.89	0.80	0.93
Actual	12.9	3.3	0.029	3.4
Actual95	15.8	3.3	0.167	23.6
Base	13.9	3.3	0.030	3.4
Base95	13.9	3.3	0.042	29.8

Table 4

Screening-level ecologically protective benchmarks

Criteria	As	BaP	DDE	Pb
EPA, water ($\mu\text{g/L}$ dissolved)	36 ^a	300 ^b	14 ^c	8.1 ^a
LA, water ($\mu\text{g/L}$ dissolved) ^d	36 ^d	NA	0.14 ^d	4.0 ^e
Sediment (mg/kg dry) ^f	7.24	0.0888	0.00207	30.2

^aChronic EPA Water Quality Criteria (US EPA, 2002).

^bMarine lowest observable effects level (NOAA, 1999).

^cAcute EPA Water Quality Criteria (US EPA, 2002).

^dChronic LA Water Quality Standards (LADEQ, 2000).

^eChronic LA Water Quality Standard for lead hardness adjusted (120 ppm CaCO_3) (LADEQ, 2000).

^fMarine threshold effects limit (TEL) from MacDonald et al. (1996) for coastal and marine waters in the Gulf of Mexico.

trations in the same vicinity as those measured were 40 ppb for BaP and 0.3 ppb for DDE. The increase in sediment concentration computed by the model for BaP is within the range of measured sediment concentrations, rather than much less as for the metals, which is reasonable given that pre-Katrina BaP sediment concentrations may have been relatively low due to the degradation and volatilization potentials of this compound. DDE would tend to persist in the sediments longer than BaP but not as long as the metals; thus, the percentage of computed sediment concentration increase to measured post-Katrina concentration of DDE seems reasonable as compared with the results for BaP and the metals.

The assumption was made for the modeling that storm water and floodwater concentrations were the same under *baseline* and *actual* conditions. Any efforts to improve the model should focus on obtaining water quality measurements in storm water under normal, *baseline* conditions without flooding through or over the levees.

At one point, consideration was given to trying to estimate the source terms that resulted in the floodwater contamination. Models are much more robust if the source terms can be quantified. However, such an undertaking would have required a tremendous effort with very high uncertainty of the results. Therefore, this idea was dropped from further consideration and is most likely not a viable goal for future studies. Furthermore, Mielke et al. (2004) reported high soil concentrations of polynuclear aromatic hydrocarbons (PAH) and metals in urban New Orleans, especially near busy city streets. These data are pre-Katrina and represent a common condition in urban areas with heavy traffic. Thus, a substantial portion of floodwater contamination may have been caused by flooding of already contaminated soils rather than rupturing or leaking chemical storage tanks. Flooding and the subsequent dewatering resulted in exposing the environment surrounding New Orleans to these contaminants. However, such exposure most likely occurs even during normal (*baseline*) dewatering, but to a less degree due to less storm water and shorter pumping durations.

7. Conclusions

The Lake Pontchartrain model indicated that incremental increases in lake sediment concentrations of contaminants as a result of the *actual* dewatering event were about an order of magnitude greater than for *baseline* removal of storm water without levee failure and overtopping. However, increases in concentration of As, Pb, and DDE in sediment were small relative to measured and suspected pre-Katrina sediment concentrations for these constituents, whereas the increases in sediment concentrations of BaP were about the same order of magnitude of post-Katrina measurements. Thus, dewatering of floodwaters is expected to have a negligible to minor impact on lake sediments relative to background concentrations for metals and long-life organic compounds; but a measurable impact is

expected for more rapidly degrading compounds, such as PAHs. Model-computed maximum sediment concentrations for BaP and DDE exceeded ecologically protective sediment quality guideline values for *actual* conditions, whereas concentrations of these chemicals did not exceed these screening values for *baseline* conditions. However, the sediment areas that exceeded sediment quality criteria are relatively small areas close to the southeast shore. Exceedence of guidelines should be interpreted with caution because the intended use of sediment quality guideline values is to screen sediments for potential adverse effects on benthic invertebrate organisms, not to quantify risk or an impact (Wenning et al., 2005).

Elevated concentrations of FCB in water are expected to occur, and do occur (Jin et al., 2004), in the lake during dewatering regardless of levee conditions (i.e., levees failures/overtopping or not). In fact, water concentrations of all constituents should be about the same, or even less, with levee failures since fewer pumps may be operating and may be operating below capacity. Lower pump discharge rates result in lower water concentrations due to larger residence times in ambient waters with greater opportunity for settling and dilution. The reason that incremental increases in sediment concentrations are expected to be higher with levee failures is that more floodwater volume must be removed, thus, dewatering takes longer and more contaminant mass is discharged to receiving waters, which is manifested as higher sediment concentrations.

Lead may present water quality concerns under any dewatering conditions regardless of the levee failures/overtopping or not. Computed maximum water concentrations of Pb exceeded water quality criteria for both *actual* and *baseline* conditions when loading concentrations are high (i.e., 95UCL). However, elevated concentrations of metals and PAHs existed in urban New Orleans soils before Katrina. Thus, the presence of these constituents is expected for urban storm water and in the pumped effluents, with or without levee failures.

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References

- Boyer, J.M., Chapra, S.C., Ruiz, C.E., Dortch, M.S., 1994. RECOVERY, a mathematical model to predict the temporal response of surface water to contaminated sediments. Technical Report W-94-4, US Army Engineer Waterways Experiment Station, Vicksburg, MS.
- Bunch, B.W., Cerco, C.F., Dortch, M.S., Johnson, B.H., Kim, K.W., 2000. Hydrodynamic and water quality model study of San Juan Bay

- Estuary. ERDC TR-00-01, US Army Engineer Research and Development Center, Vicksburg, MS.
- Cerco, C., Cole, T., 1993. Three-dimensional eutrophication model of Chesapeake Bay. *Journal of Environmental Engineering* 119 (6), 1006–1025.
- Cerco, C., Cole, T., 1995. User's guide to the CE-QUAL-ICM three-dimensional eutrophication model, release version 1.0. Technical Report EL-95-15, US Army Engineer Waterways Experiment Station, Vicksburg, MS.
- Cerco, C.F., Bunch, B., Cialone, M.A., Wang, H., 1994. Hydrodynamic and eutrophication model study of Indian River and Rehoboth Bay, Delaware. Technical Report EL-94-5, US Army Engineer Waterways Experiment Station, Vicksburg, MS.
- Cerco, C.F., Bunch, B., Letter, J., 1999. Impact of flood-diversion tunnel on Newark Bay and adjacent waters. *Journal of Environmental Engineering* 125 (4), 328–338.
- Cerco, C.F., Bunch, B.W., Teeter, A.M., Dortch, M.S., 2000. Water quality model of Florida Bay. ERDC/EL TR-00-10, US Army Engineer Research and Development Center, Vicksburg, MS.
- Cerco, C.F., Noel, M.R., Kim, S.C., 2004. Three-dimensional eutrophication model of lake Washington, Washington State. ERDC/EL TR-04-12, US Army Engineer Research and Development Center, Vicksburg, MS.
- Cerco, C.F., Noel, M.R., Kim, S.C., 2006. Three-dimensional management model for Lake Washington, Part II: eutrophication modeling and skill assessment. *Lake and Reservoir Management* 22, 115–131.
- Chapra, S.C., 1977. *Surface Water-Quality Modeling*. McGraw-Hill, New York.
- Estes, T., Schroeder, P., Steevens, J.A., 2006. Interim data report for physical characterization and column settling tests for sediments collected from IHNC, New Orleans, LA. March 13, 2006.
- Jin, G., Englande, A.J., Bradford, H., Jeng, H., 2004. Comparison of *E. coli*, Enterococci, and fecal coliform as indicators for brackish water quality assessment. *Water Environment Research* 74 (4), 245–255.
- Johnson, B.H., Heath, R.E., Hsieh, B.B., Kim, K.K., Butler, H.L., 1991. User's guide for a three-dimensional numerical hydrodynamic, salinity, and temperature model of Chesapeake Bay. Technical Report HL-91-1, US Army Engineer Waterways Experiment Station, Vicksburg, MS.
- Johnson, B.H., Kim, K., Heath, R., Hsieh, B., Butler, L., 1993. Validation of a three-dimensional hydrodynamic model of Chesapeake Bay. *Journal of Hydraulic Engineering* 119 (1), 2–20.
- Karickhoff, S.W., Brown, D.S., Scott, T.A., 1979. Sorption of hydrophobic pollutants on natural sediments. *Water Research* 13, 241–248.
- Kim, S.C., Cerco, C.F., Johnson, B.H., 2006. Three dimensional management model for Lake Washington, Part I: Introduction and hydrodynamic modeling. *Lake and Reservoir Management* 22, 103–114.
- Lake Pontchartrain Basin Foundation (LPBF), 2006. Weekly Water Quality Report. Available online at: http://www.saveourlake.org/water_quality.asp.
- Louisiana Department of Environmental Quality (LADEQ), 2000. Part IX. Water Quality Regulations: Chapter 11, Surface Water Quality Standards.
- MacDonald, D.D., Carr, R.S., Calder, F.D., Long, E.R., Ingersoll, C.G., 1996. Development and evaluation of sediment quality guidelines for Florida coastal waters. *Ecotoxicology* 5, 253–278.
- Mielke, H.W., Wang, G., Gonzales, C.R., Powell, E.T., Le, B., Quach, V.N., 2004. PAHs and metals in the soils of inner-city and suburban New Orleans, Louisiana, USA. *Environmental Toxicology and Pharmacology* 18 (2004), 243–247.
- Mills, W.B., Dean, J.D., Porcella, D.B., Gherini, S.A., Hudson, R.J.M., Frick, W.E., Rupp, G.L., Bowie, G.L., 1982. Water quality assessment: a screening procedure for toxic and conventional pollutants, Part I. EPA-600/6-82-004a, Environmental Research Laboratory, Office of Research and Development, USEPA, Athens, GA.
- National Oceanic and Atmospheric Administration (NOAA), 1999. Screening quick reference tables (SQUIRT), HAZMAT Report 99-1, Seattle, WA.
- Pardue, J.H., Moe, W.M., Mcinnis, D., Thibodeaux, L.J., van Heerden, I., Korevec, N., Yuan, Q.Z., 2006. Chemical and microbiological parameters in New Orleans floodwater following Hurricane Katrina. *Journal of Environmental Science and Technology* 39, 8591–8599.
- Penland, S., Beale, A., Kindinger, J. (Eds.), 2002 (On line posting). Environmental Atlas of Lake Pontchartrain Basin, USGS Open File Report 02-206 <<http://pubs.usgs.gov/of/2002/of02-206/index.html>>.
- Presley, S.M., Rainwater, T.R., Austin, G.P., Platt, S.G., Zak, J.C., Cobb, G.P., Marsland, E.J., Tian, K., Zhang, B., Anderson, T.A., Cox, S.B., Abel, M.T., Leftwich, B.D., Huddleston, J.R., Jeter, R.M., Kendall, R.J., 2006. Assessment of pathogens and toxicants in New Orleans, LA following Hurricane Katrina. *Journal of Environmental Science and Technology* 40, 468–474.
- Ruiz, C.E., Gerald, T., 2001. RECOVERY version 2.0, a mathematical model to predict the temporal response of surface water to contaminated sediments. ERDC/EL TR-01-3, US Army Engineer Research and Development Center, Vicksburg, MS.
- Thomann, R.V., Mueller, J.A., 1987. *Principles of Surface Water Quality Modeling and Control*. Harper & Row, New York.
- US Environmental Protection Agency (US EPA), 1986. Ambient water quality criteria for bacteria. EPA-440/5-84-002, Cincinnati, OH.
- US Environmental Protection Agency (US EPA), 2002. National Recommended Water Quality Criteria. EPA-822-R-02-047. <<http://www.epa.gov/waterscience/criteria/wqcriteria.html#D>>, Washington, DC.
- US Environmental Protection Agency (US EPA), 2006. Environmental assessment summary for areas of Jefferson, Orleans, St. Bernard, and Plaquemines Parishes flooded as a result of Hurricane Katrina. Available online at: <http://www.epa.gov/katrina/testresults/katrina_env_assessment_summary.htm>.
- Wenning, R.J., Batley, G.E., Ingersoll, C.G., Moore, D.W., 2005. Use of Sediment Quality Guidelines and Related Tools for the Assessment of Contaminated Sediments. SETAC Press, Pensacola, FL.