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Post-Katrina Interagency Performance Evaluation Taskforce (IPET)

Contaminant Fate/Transport Modeling for Environmental Consequences of IPET Task 9

Mark S. Dortch, Mansour Zakikhani, and Sung-Chan Kim

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Abstract: This report documents mathematical model studies conducted to evaluate the environmental impacts of Hurricane Katrina. One of the primary environmental concerns associated with Hurricane Katrina was the impacts to ecological resources stemming from contaminants that were released into the floodwaters and subsequently pumped into surrounding water bodies outside the levee system. Contaminant concentrations within the water column and sediment bed were computed for two environmentally important water bodies, Lake Pontchartrain and Violet Marsh, both of which received pumped floodwater effluents following Hurricane Katrina. Two different mathematical models were used to simulate contaminant concentrations within the lake and marsh. Contaminant concentration information in these two systems was used to draw conclusions regarding the environmental consequences of contaminant releases. The models were used to determine the consequences of dewatering the floodwaters of Hurricane Katrina, with or without levee failure.

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Preface

This report documents the application of numerical contaminant transport and fate models to Lake Pontchartrain and Violet Marsh to evaluate any potential impacts to these water bodies resulting from dewatering of contaminated floodwaters in New Orleans following Hurricane Katrina. The models were used to help understand the environmental consequences of dewatering by comparing results for the actual flood conditions against those projected without the severe flooding associated with levee failures and overtopping.

This study was conducted as part of Interagency Performance Evaluation Task Force (IPET) performance evaluation of the New Orleans and Southeast Louisiana hurricane protection system. This study fell under Task 9, Consequences Analysis, which dealt with environmental, economic, human health and safety, social, cultural, and historic consequences of the event. The U.S. Army Corps Engineers was responsible for executing the IPET, and the Corps' Institute for Water Resources (IWR) was responsible for Task 9. The study was funded by IWR.

This study was conducted by Drs. Mark Dortch, Mansour Zakikhani, and Sung-Chan Kim of the Water Quality and Contaminant Modeling Branch (WQCMB), Environmental Processes and Engineering Division (EPED), Environmental Laboratory (EL), of the U.S. Army Engineer Research and Development Center (ERDC). The work was conducted under the general supervision of Dr. Barry Bunch, Chief, WQCMB, Dr. Richard E. Price, Chief, EPED, and Dr. Beth Fleming, Director, EL. Dr. Barbara Kleiss of the Wetlands and Coastal Ecology Branch, Ecosystem Evaluation and Engineering Division, EL, was the ERDC point of contact for the environmental consequences work of IPET Task 9. This report was prepared by Drs. Dortch, Zakikhani, and Kim. The report was reviewed by Ms. Dorothy Tillman and Mr. Mark Noel of the WQCMB.

COL Richard B. Jenkins, was Commander and Executive Director of ERDC. Dr. James R. Houston was Director.

Unit Conversion Factors

Multiply	By	To Obtain
feet	0.3048	meters
inches	0.0254	meters

1 Introduction

Background

Mathematical models fill data gaps and provide information to relate impacts to different operational scenarios. One of the primary environmental concerns associated with Hurricane Katrina was the impacts to ecological resources stemming from contaminants that were released into the floodwaters and subsequently pumped into surrounding water bodies outside the levee system. Models were used within Task 9 to provide information with which to more fully evaluate such environmental consequences. This technical report describes these model studies and the results obtained from them.

Objective

The objective of the Task 9 environmental modeling was to compute contaminant concentrations within the water column and sediment bed for two environmentally important water bodies, Lake Pontchartrain and Violet Marsh, which received pumped floodwater effluents following Hurricane Katrina. These two systems were selected for study since they were primary recipients of pumped floodwaters, they contain valuable ecological resources, and they are representative of natural ecosystems that are adjacent to New Orleans. The goal was to provide more complete information on contaminant concentrations in these two systems so that more definitive conclusions regarding environmental consequences resulting from contaminant releases could be drawn.

2 Approach

Two different mathematical models were used to simulate contaminant concentrations within the lake and marsh. Both models and their applications are described below. Contaminant concentrations were modeled for two conditions: 1) dewatering of floodwaters for the "*actual*" conditions that occurred with levee breaches, and 2) dewatering for conditions of the system performing as designed ("*baseline*" conditions). The *baseline* conditions serve as a basis for comparison with the *actual* conditions.

Lake Pontchartrain Model

The three-dimensional (3D) hydrodynamic model CH3D and the 3D water quality model CE-QUAL-ICM (ICM) were used to model conditions in Lake Pontchartrain for a period of 90 days starting on September 1, 2005. Thus, both the *actua*l and *baseline* conditions include the period during which pumps were operating to remove flood and rainwater from the city following Hurricanes Katrina and Rita.

The z-plane version of CH3D was used (Johnson et al. 1991, 1993). This version has a varying number of layers with the total number along a water column depending on the depth. Figure 1 is a plan view of the model computational grid. The grid contains 6038 computational cells in the surface layer and 21018 cells in total over all layers. Each layer thickness was 5 ft except the surface layer, which varied depending on the water surface elevation. The maximum number of layers was 6 for a maximum depth of about 30 ft. A typical cell size in plan form is on the order of 300 m by 700 m.

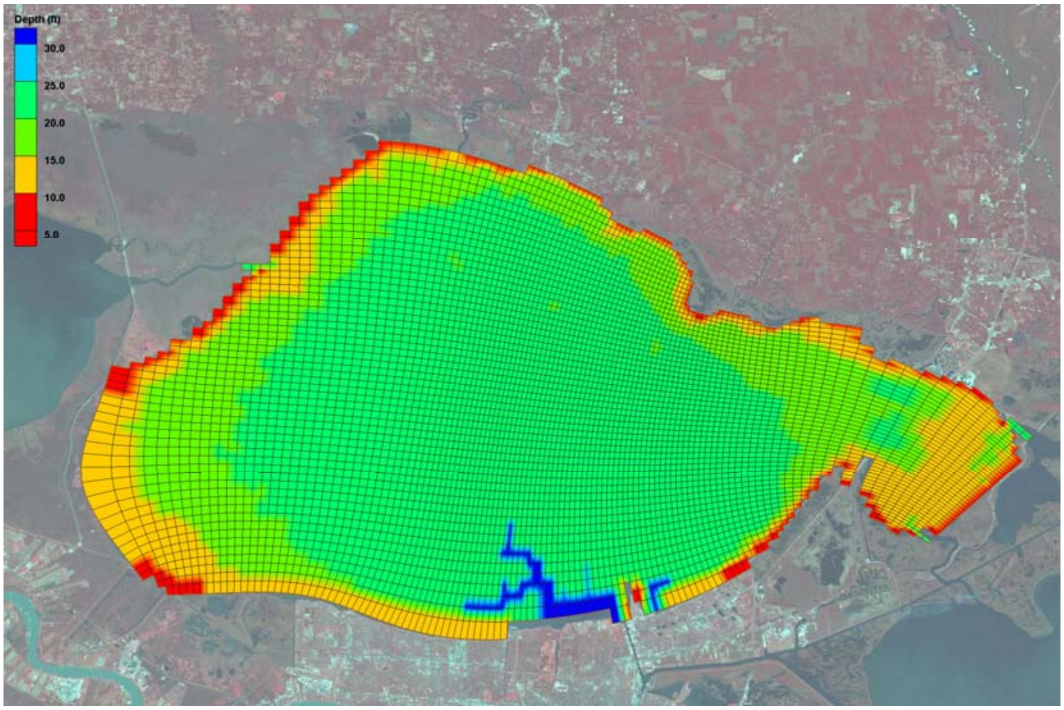

Figure 1. Model grid for Lake Pontchartrain with depth contours.

Although the CH3D model includes baroclinic terms (i.e., it can 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. Activating this feature would have increased the input data and modeling requirements substantially, so it did not seem warranted given the paucity of data. Also, given the fact that the pumped discharges following Katrina dominated the lake currents and thus the salinity along the south shore, ignoring salinity differences is a reasonable approach. 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) in the fall of 2005, whereas salinities across the lake near Slidell at Station 10 (North Shore Beach) were higher, averaging 14 ppt. Salinity measurements taken throughout October 2005 in the New Orleans floodwaters averaged 5.6 ppt, where floodwater salinity was higher near the lake shore (ranging from approximately 7 to 9 ppt) and decreased moving south towards downtown away from the shore (ranging from approximately 2 to 6 ppt). These data indicate that the pumped water near the shore was nearly the same salinity as the lake water near the shore. Therefore, model output from the surface layer should be representative of expected lake concentrations resulting from pumping (since pumped water enters along the surface of the lake, and with about the same salinity, it would remain near the water surface).

The hydrodynamic model was run for both *actual* and *baseline* conditions. Wind speed and direction from the New Orleans International Airport were applied to the model. NOAA tide records were used as boundary conditions at the open sea boundary, which was located at the Rigolets inlet to the lake. Additionally, the pumped flows were applied as boundary conditions along the south shore and were the major forcing function for lake hydrodynamics during *actual* dewatering. Freshwater stream flows entering the lake were not included in the model since such flows are small with respect to wind and tidal forcing except when Mississippi River floodwaters are diverted through the Bonne Carre Spillway.

The hydrodynamic model used the estimated pumping rates obtained from IPET Task 8 for *actual* post-Katrina conditions, including pumping during and following Hurricane Rita. These data show that the pumps started operating on September 11, 2005 and ended on October 20 (October 17 for pumpout 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 this study was conducted. In the absence of these 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. Known pump flow capacities were divided into 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. The biggest problem with this approach is that it did 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 flooding by these two hurricanes, with the exception of rainwater.

The water quality model code was originally developed during a eutrophication model study of Chesapeake Bay (Cerco and Cole 1993), but has subsequently been applied to a variety of systems throughout the United States and the San Juan Bay and Estuary in Puerto Rico. The Lake Pontchartrain version is based on a model recently used for Lake Washington, WA (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 run the water quality model.

The water quality model was applied for five constituents: arsenic (As); lead (Pb); benzo(a)pyrene (BaP); DDE, a degradation product of DDT; and fecal coliform bacteria (FCB). 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 or 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. These constituents provide a wide range in adsorption behavior, with arsenic having a relatively low sorption distribution coefficient and DDE having a high value. Fecal coliform bacteria can serve as a tracer if die-off is set to 0.0.

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 were collected and analyzed for a host of contaminants. These data were collected by the U.S. Environmental Protection Agency (EPA), Louisiana State University (LSU), and Louisiana Department of Environmental Quality (DEQ) and were assembled into an EPA database. Data for the constituents assessed in this study were extracted from the database and analyzed. This analysis produced concentrations to calculate the model pumped loadings and resulted in a median concentration and a 95-percent 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 the 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, it was decided 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. 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.

Estimating pumped concentrations for the *baseline* conditions was problematic. The plan was to use data from a previous pumping event that endured rainfall of the amount that fell during Katrina. However, no such data could be found for the constituents being studied. Pardue et al. (2006) state that the metal concentrations in the Katrina floodwaters of New Orleans were generally typical of storm water with a few exceptions of elevated concentrations of lead. Jin et al. (2004) reported FCB concentrations of 40,000 most probable number (MPN) /100 ml measured during 1998 in pumped storm water in canals that drain into the lake. This concentration is within the range of values measured in the Katrina floodwaters. Given this information, it was assumed that the pumped concentrations for the *baseline* conditions are the same as those for the *actual* conditions. However, total constituent mass discharged for the *baseline* condition is much 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 of the benthic sediments beneath each bottom water cell. Such a large amount of data requires reduction to render presentations that are useful for interpretation. Two-dimensional surface layer and sediment contour plots of maximum concentrations were used to collapse data down for manageable interpretation.

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. However, some water surface elevation and fecal coliform bacteria measurements were available for the lake during the fall of 2005, which were used as discussed in the Calibration/Validation section. The flow fields and transport computed with these models have been found to be relatively accurate in other studies if sufficient boundary conditions for inflows, water levels, winds, and constituent loadings are provided.

Violet Marsh Model

The upper portion of Violet Marsh was selected for the model study, since this area directly received pumped discharges and is well-defined geometrically, bounded on all four sides by a bayou, a highway, a levee, and a wastewater treatment facility. This simplified the modeling approach while focusing on an area of environmental interest that received significant pumped water. Given the simplicity of this water body and the rapid flushing rate, which was on the order of half a day during Katrina dewatering, it was possible to use a much simpler model than for the lake; thus, the RECOVERY model (Ruiz and Gerald 2001) was used. RECOVERY was first developed in 1994 (Boyer et al. 1994) but has been modified and improved over the years; the most recent version (4.3) was used for this study. 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; however, it can accept time-varying loadings. The surface area of upper Violet Marsh is 9.76E6 m^2, and the mean depth is about 0.175 m, which results in a very short residence time of less than a day for typical pumped discharges.

The assumptions made for the lake model were also used for the marsh model. Thus, the *baseline* pump flows were based on rainfall volume and pump capacity, and the *baseline* pumped concentrations were set equal to the concentrations obtained for the *actual* conditions. Pumped flows for *actual* conditions were based on the Task 8 estimates, and the associated pumped concentrations were based on the analysis of sample measurements taken in St. Bernard Parish floodwaters.

The same five constituents selected for the lake model were modeled for the marsh. The marsh model simulated a year for each run with the loading starting on day 1 and ending on day 2 for *baseline* conditions and starting on day 1 and ending on day 37 for *actual* conditions. The RECOVERY model assumes a constant background flushing flow rate, but allows time-varying loading of constituents. During pumping, the flushing flow should be equal to the pumped flow. However, when pumping and loading cease, the flushing flow should drop to an unknown, but much lower, background flow. The background flow was assumed to be 0.1 m^3/sec for this small isolated wetland system. In order to account for two different flushing flows (during pumping and after pumping ceases), two separate runs were required. The computed peak water and sediment concentrations, which occurred when pumping ceased, were taken from the first model run (i.e., flushing flow equal to pumping flow) and used as initial conditions for a second run with the flushing flow set to a nominal background flow of 0.1 m^3/sec. This two-step flushing condition should yield a much more reasonable prediction of peak sediment concentrations, which should be slightly higher than those computed from the first model run since the water column concentrations are flushed out much more slowly after pumping ceases, thus raising sediment concentrations.

3 Model Inputs

Parameters

In addition to boundary conditions and model control variables, several other parameters and basic data are required to apply the lake and marsh models. These include:

- total suspended solids (TSS), mg/L
- TSS settling rate, (m/day)
- fecal coliform bacterial die-off rate, day^{-1}
- fraction of total organic carbon (TOC) to total sediment by 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 Inner Harbor Navigation Channel (IHNC) for studies of the Corps Dredged Material Management Units (DMMU). Abundant turbidity data existed for both the floodwaters and the lake. There were no turbidity data for the marsh; thus, the floodwater values were used. TSS settling rate was set to 1 m/day based on experience in modeling similar systems and particle settling studies conducted for the DMMUs of 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 for seawater. Jin et al. (2004) measured FCB die rates of approximately 2.8 day^{-1} in the lake. A relatively conservative value of 1.0 day^{-1} was used in the model. Data from Corps DMMU studies indicated f_{oc} values of about 0.02 (2 percent), which were used for the lake and marsh. Resuspension was assumed to be zero, and using the settling rate, TSS concentration, and sediment porosity to compute 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 values of 300 and 4,000 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)

$$K_{oc} = 0.6\ K_{ow} \tag{1}$$

where K_{ow} is the octonol to water partition coefficient for organic chemicals. Databases of chemical properties were searched to obtain values of K_{ow} for BaP and DDE of 1.0E6 and 3.24E6, 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 and particulate organic suspended solids are used for inorganic and organic contaminants, respectively; thus, care must be taken in defining K_d for use in ICM as explained in the next section. The TOC concentration is the product of f_{oc} and TSS for the water column or 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 other ambient conditions and can vary from system to system for the same chemical, testing was conducted with the RECOVERY model to validate the literature values for K_d and K_{ow}. RECOVERY was run to steady state assuming no flushing, settling, resuspension, degradation, or volatilization; 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. The model indicated that the value of 4,000 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 0.5E6 and 1.0E6 L/kg, respectively, to match observations. Since Equation 1 is programmed into the RECOVERY model code, and f_{oc} is a measured 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 it 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 resulted in relatively minor model adjustments that gave increased confidence in the modeled sorption process.

Lake Model

As discussed in the approach section, the hydrodynamic model of the lake was driven with winds, tides at the Rigolets Inlet, and pumped discharges from New Orleans. The hydrodynamic model was started with quiescent conditions on September 1, 2005, and run for 90 days. Observed wind data from the New Orleans airport were used as input. However, wind data were not available until September 7 due to Katrina, so the wind vectors 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 is considered a model spin-up period and should not be used for model analysis and comparisons.

The water level boundary conditions at the entrance of Lake Pontchartrain (Rigolets Inlet) included both meteorological and astronomical tides. For the astronomical tide, hourly predictions from NOAA at Waveland, MS, (station number 8747766) were used. A predicted tide station at Long Point, LA, in Lake Borgne 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 were used for this location. 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-hr moving average to filter out higher frequency signals. The sub-tidal signal was moved back in time by 24 hr 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 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. Figure 2 shows time series plots of the combined, estimated pumping rates into waterways emptying 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 report presentation, but the flow for each pump constituted a separate discharge input for the model. The pump stations included in the lake model are shown in Figure 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 are pumped into. Pumped discharges were used for inflows to the hydrodynamic model and for calculating loading inputs for the water quality model. The combined pumped flows of Figure 2 represent a total pumped volume of about 19 E9 ft^3, which is approximately 6 to 8 percent of the lake volume.

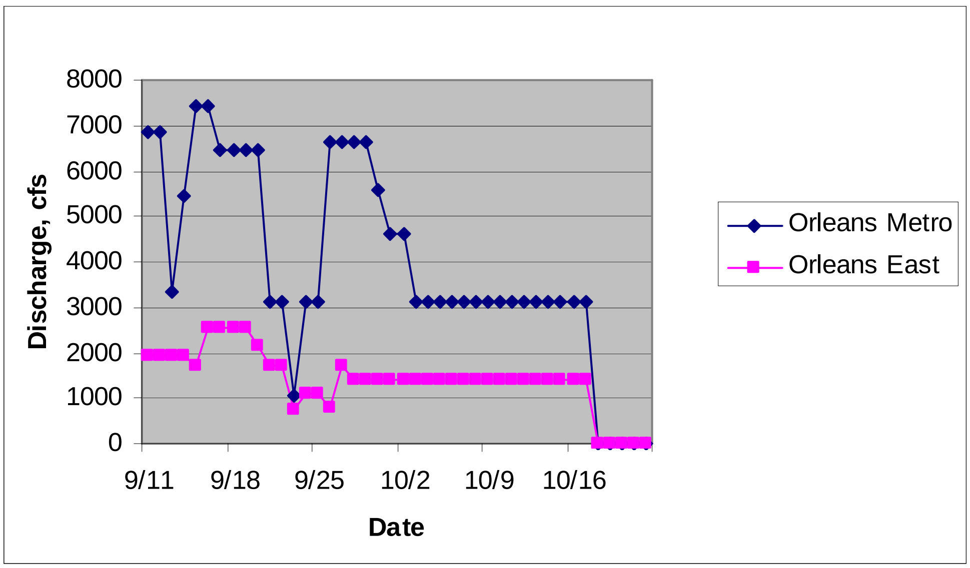

Figure 2. Time series of combined *actual* pump discharges for Orleans Metro and Orleans East that pumped into the lake.

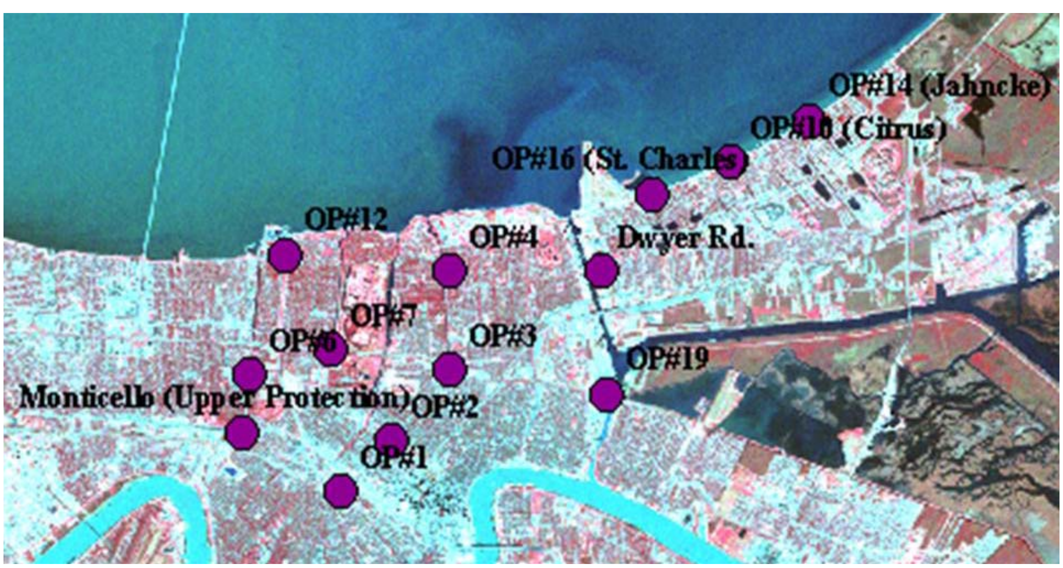

Figure 3. Locations of pump stations for Orleans Parish included in the model that pump into the lake.

As explained in the Approach section, rainfall and pump capacities were used to establish the *baseline* pumping conditions. The rainfall reported at Slidell (other gages in the area did not report) was approximately 8 in. for Katrina, and the rainfall reported at the New Orleans International Airport for Rita was 2.3 in. Given the approximate, combined collection basin area for Orleans Metro and Orleans East of 3 E9 ft^2, the rainfall volumes for Katrina and Rita were about 2 E9 ft^3 and 0.6 E9 ft^3, respectively. The pump-to-lake total discharge capacity in Orleans Parish is approximately 38,000 ft^3/sec (cfs). This means that in Orleans Parish, 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 pumpout time for Katrina under these assumptions is about 0.6 day. The capacity of 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 (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 are 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 1,000 cfs, then the flow used for the model pump was 600 cfs based on the Katrina pumpout time of 0.6 day.

The ICM model includes inorganic suspended solids (ISS) and suspended particulate organic carbon (POC) as modeled state variables, rather than TSS. 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. POC constitutes about 40 percent of the volatile suspended solids (VSS), where VSS represents suspended particulate organic matter. It is possible to estimate ISS using TSS and f_{oc} data and recognizing that TOC is primarily made up of POC; thus,

$$TSS = ISS + VSS = ISS + 2.5 f_{oc} TSS \tag{2}$$

Rearranging Equation 2 yields

$$ISS = TSS\left(1 - 2.5 f_{oc}\right) \tag{3}$$

POC is calculated from

$$POC = f_{oc}\, TSS \tag{4}$$

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

$$F_{pi} = \frac{K_d\, ISS}{1 + K_d\, ISS} \tag{5}$$

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

$$F_{po} = \frac{K_d\, POC}{1 + K_d\, POC} \tag{6}$$

The units for K_d in ICM are m^3/g; thus, the values presented in the previous section were converted from L/kg to m^3/g by multiplying by 1.0 E-6. The K_d values input for lead (Pb) and arsenic (As) were 0.5 E-3 and 4.0 E-3 m^3/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 Equation 1 to get K_{oc}) used in ICM were 0.3 and 0.6 m^3/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 factor over flow for lakes; thus, the volatilization rate was computed based on wind speed, using an average speed of 5 mph (3 m/sec). This wind speed, Henry's law constants for BaP and DDE of 4.5E-7 and 4.0E-5 atm-m^3/mole, respectively, and respective molecular weights of 252 and 318 g/mole were used to compute K_{vol} using the algorithm within the RECOVERY model. Volatilization within RECOVERY is based on the Whitman two-film theory presented by Chapra (1997) where the gas and liquid side mass transfer rates are computed from wind speed. The resulting values of K_{vol} 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 (g/m²/day). The dissolved organic chemical concentration is the product of the total organic chemical concentration times the quantity ($1-F_{po}$). Table 1 summarizes values for the various parameters for partitioning and volatilization.

Table 1. Partitioning and volatilization parameters used for the lake model.

Chemical	K_{ow} (L/kg)	K_d (L/kg)	K_d (m³/g)	K_{vol} (m/day)
As	NA	500	0.5E-3	NA
Pb	NA	4000	4.0E-3	NA
BaP	0.5E6	0.3E6	0.3	0.005
DDE	1.0E6	0.6E6	0.6	0.19

The concentrations used to establish the lake water quality model loadings are shown in Table 2. The loadings were categorized by median and 95UCL concentrations, which were determined from statistical analysis of the floodwater measurements taken in the two areas (Orleans Metro and Orleans East).

Table 2. Lake loading concentrations (total) by region for b*aseline* and *actual* conditions.

Constituent	Median, μg/L	95UCL, μg/L
Orleans Metro		
Arsenic	20	20
BaP	5	5
DDE	0.05	0.05
Lead	5	44
Fecal coliform bacteria	2,200[1]	70,041[1]
Orleans East		
Arsenic	20	26
BaP	5	5
DDE	0.05	0.38
Lead	2.5	12
Fecal coliform bacteria	200[1]	32,869[1]
[1] Units are colony forming units (cfu) /100 ml or MPN/100 ml.		

Turbidity for Lake Pontchartrain is 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. 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. Although the ICM model transports sediment, it was possible to hold the value constant by setting initial conditions and all boundary conditions to the background value. Resuspension rate was set to zero in the lake model, and the surficial sediment bed layer thickness was set to 0.2 m. The burial rate was set to 0.026 m/yr, which was computed from a steady-state solids balance in the bed and a settling rate of 365 m/yr. Degradation rates were set to zero for all constituents except FCB.

Marsh Model

Using an 8-in. rainfall for the New Orleans area during Katrina and an approximate collection area of 3.4 E8 ft^2, the approximate rainfall volume for the area of St. Bernard Parish that was pumped into the marsh was estimated to be 6.4 E6 m^3. Using the combined pump capacities for pumps 1 and 6 of 70 m^3/sec, the estimated dewatering time without levee failures or overtopping is about 26 hr, or about a day. Thus, a pump flow of 70 m^3/sec over one day was used for the pump operations to establish the loading and background flushing during dewatering for *baseline* conditions. It was assumed that pump 4 would not be used, as it was not used following Katrina. Also, rainfall from Rita was not considered for the marsh modeling for either condition.

The estimated *actual* pump flows for pumps 1 and 6 and concentration measurements taken from St. Bernard Parish were used for the *actual* conditions. The estimated flows through pumps 1 and 6 were combined and averaged over the 37-day pumping period, yielding an average pump flow rate of 31 m^3/sec for 37 days. This flow was used to establish the loading and to set the modeled system background flushing flow rate during dewatering for *actual* conditions. The volume of upper Violet Marsh is about 1.7 E6 m^3. With a flow of 31 m^3/sec, the flushing time for the marsh is 0.63 day.

The RECOVERY model was run for all five constituents, for *actual* and *baseline* conditions, and for two loadings based on median and 95UCL concentrations for each of the two conditions. These combinations constituted four runs, since all five constituents could be included in a run.

These runs are referred to as Actual and Base and Actual95 and Base95 for the *actual* and *baseline* conditions with median and 95UCL loading concentrations, respectively. As described in the Approach section, the results from these runs were used as initial conditions for subsequent runs with a low-level background flushing flow following pumped flow cessation. Loading concentrations for *baseline* and *actual* conditions are shown in Table 3.

Table 3. Marsh loading concentrations (total) for *baseline* and *actual* conditions.

Constituent	Median, μg/L	95UCL, μg/L
Arsenic	12.0	14.0
BaP	5.0	5.0
DDE	0.05	0.1
Lead	2.5	4.9
Fecal coliform bacteria	90[1]	1708[1]
[1] Units are cfu/100 ml or MPN/100 ml.		

The RECOVERY model required TSS as an input parameter for calculating water column particulate contaminant concentrations. Turbidity measurements obtained from the floodwaters following Katrina were analyzed for median concentration, which was converted to a TSS concentration of 19.8 mg/L. This value was used in the model since the short flushing time of the marsh will result in marsh TSS concentrations equal to that of the floodwater pumped into it.

The RECOVERY model uses K_{ow} and Equation 1 to compute K_{oc} and the product of K_{oc} and f_{oc} to compute K_d for organic chemicals. Then TSS is used in place of POC in Equation 6 to calculate the fraction of particulate organic chemical to total organic chemical concentration. For inorganic chemicals, K_d values are input directly into the model.

RECOVERY requires several other inputs, including the sediment dry density, which was 2.65 g/ml, and surficial layer thickness, which was set to 0.2 m. Sediments are typically found to be fairly well mixed over a depth of 0.2 m. The surficial sediment layer thickness does affect computed sediment concentrations. The average wind speed of 3.0 m/sec was applied to the marsh model. Sediment resuspension rate was set to 0.0. A burial rate of 0.026 m/yr was computed by the model from a steady-state solids balance in the bed and a settling rate of 365 m/yr. Degradation rates were set to 0.0 for all constituents except FCB, which had a die-off rate of 1.0 day^{-1}.

4 Calibration/Validation

A limited level of model calibration and validation was undertaken for the lake model, but due to lack of data, calibration and validation were not conducted for the marsh model. Model calibration/validation was less important for the marsh given the simplicity of the marsh system and its modeling approach. The preferred approach is to adjust model parameters (i.e., calibrate) to match observations as well as possible for one set of conditions, then validate how well the model can reproduce observations using a different, independent set of conditions. Given the data limitations and short time available to conduct this study, it was not possible to adhere to the usual protocol for model calibration/validation. Observational data collected during September and October 2005 were used to conduct concurrent model calibration and validation. The hydrodynamic model was executed for *actual* conditions following Katrina. Model inputs were adjusted to bring the model into agreement with observed water surface elevations in the lake. The water quality model was applied for FCB during *actua*l conditions following Katrina to validate the model against observed FCB in the lake using the calibrated hydrodynamic model output.

Computed and observed water surface elevations during September and early October 2005 at the Norco gage of Lake Pontchartrain are shown in Figure 4. This was the only water level observation gage available in Lake Pontchartrain for model comparison. This gage was not operational between October 10 and December 2, 2005. The model compares closely with data collected throughout the observation period with the exception of the first 4 days (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 usually available for most estuarine and coastal hydrodynamic model applications. As stated previously, this model has been found to perform quite well if boundary conditions are adequately prescribed. Such was the case here, as it was not necessary to make any adjustments in model parameters, such as bottom roughness and wind drag coefficients.

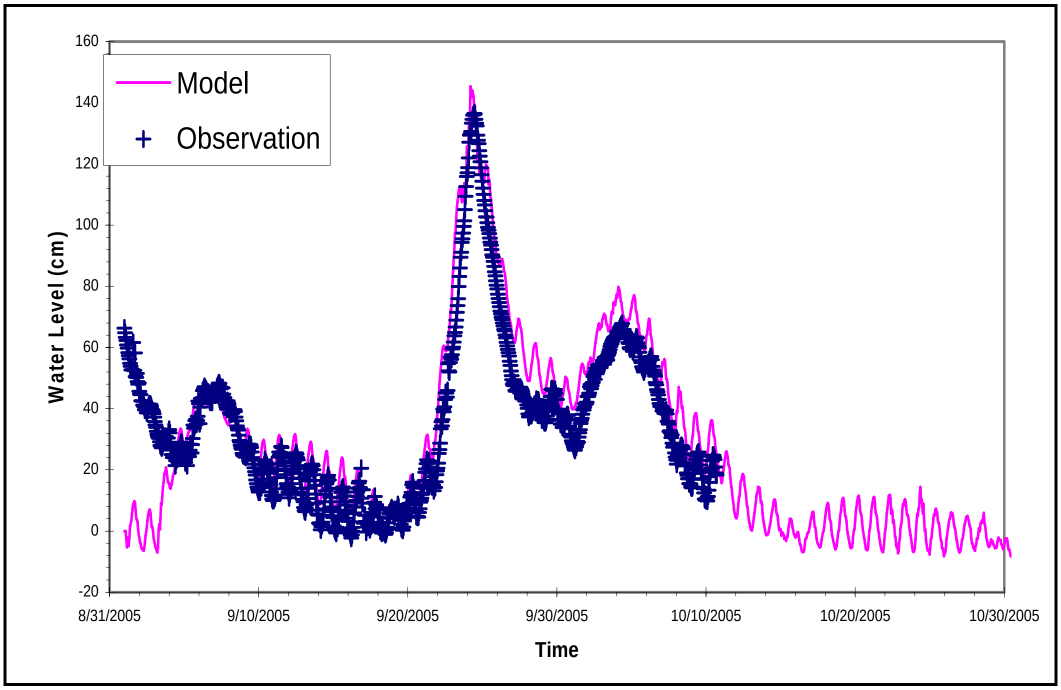

Figure 4. Computed and observed water level in Lake Pontchartrain for tide gage 8762372 East Bank 1, Norco, Bayou LaBranche, LA.

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

The lake water quality model output for FCB and *actual* conditions were compared to lake measurements of FCB obtained by the Lake Pontchartrain Basin Foundation following Katrina. The Foundation's water sampling station locations are shown in Figure 6. Model and observed data are compared in Figure 7 for stations 1-4 where data were available during in the fall along the south shore. Data for station 5 were not available during those months, and data at stations along the north shore were not compared, since the model did not include any FCB loadings from the north shore.

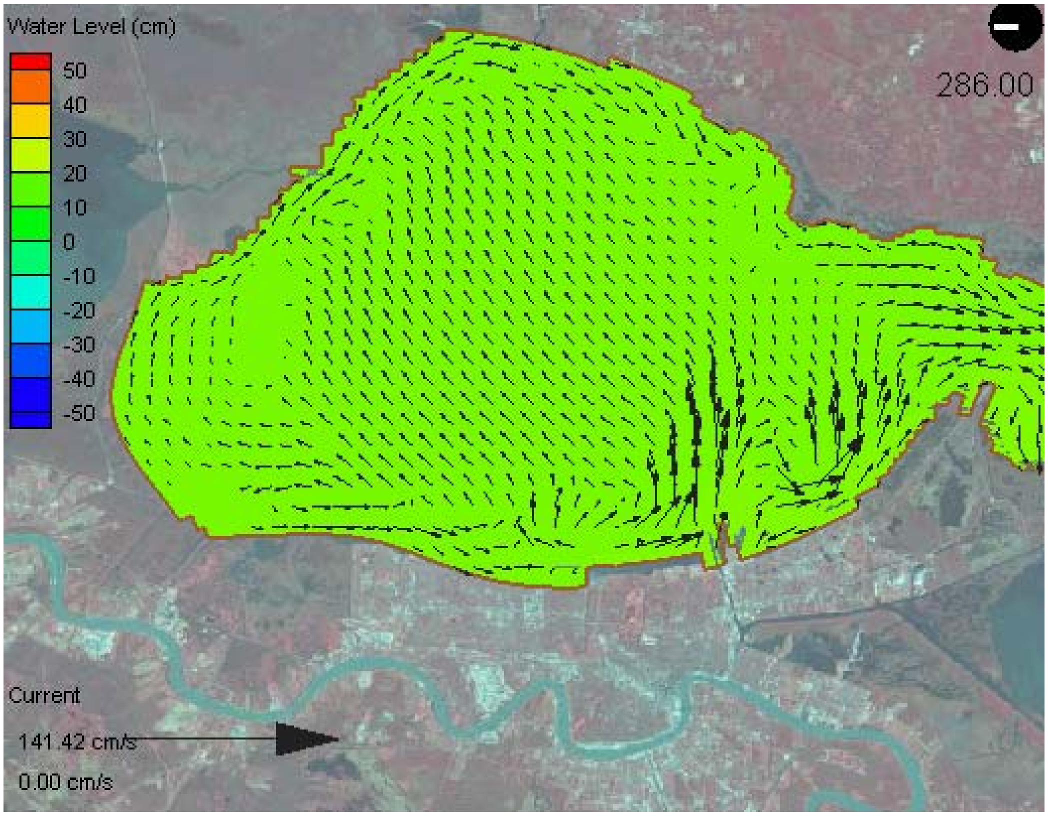

Figure 5. Computed surface layer currents at the end of September 12, 2005, *actual* conditions.

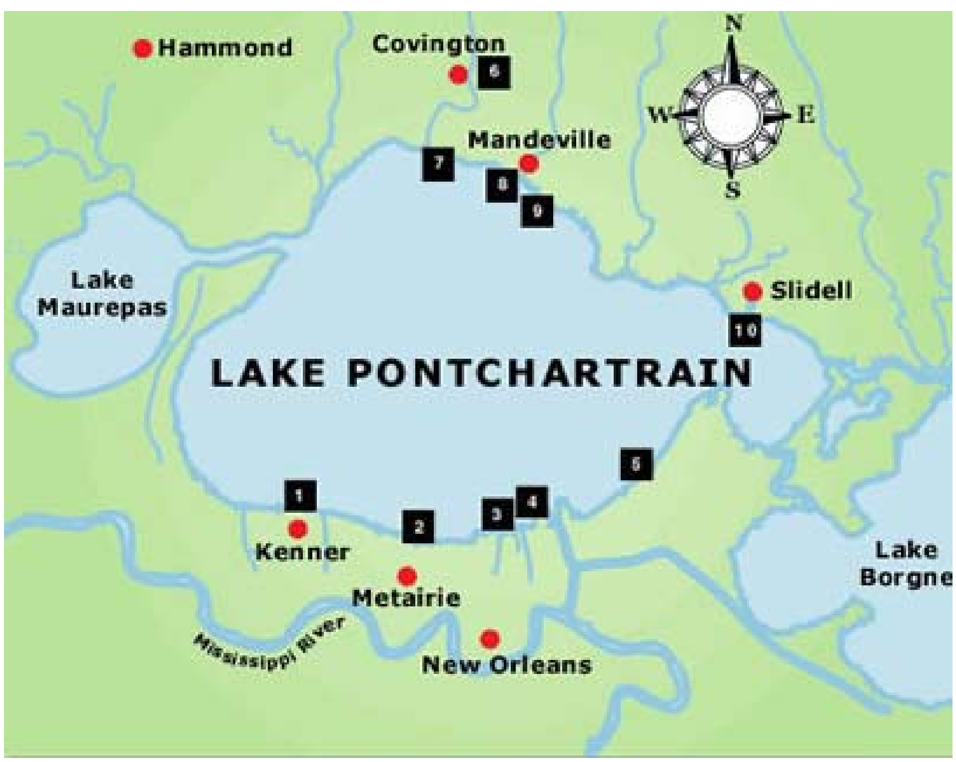

Figure 6. Lake Pontchartrain Basin Foundation water quality sampling station locations.

The observed data in Figure 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 pumpout was completed, but observations indicate that there must have been other source loadings into the lake after that date that are not accounted for in the model. Thus, there is a window of only about 18 days in early October with which to meaningfully compare the model and observations. During that window, the model is in general agreement with observations, with the exception of one outlier observed at station 2. Comparisons after October 18 should be disregarded since there are no model loadings after that date. There is not enough information in Figure 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. Further work would be required to find data adequate for use in model validation and to conduct the model validation applications.

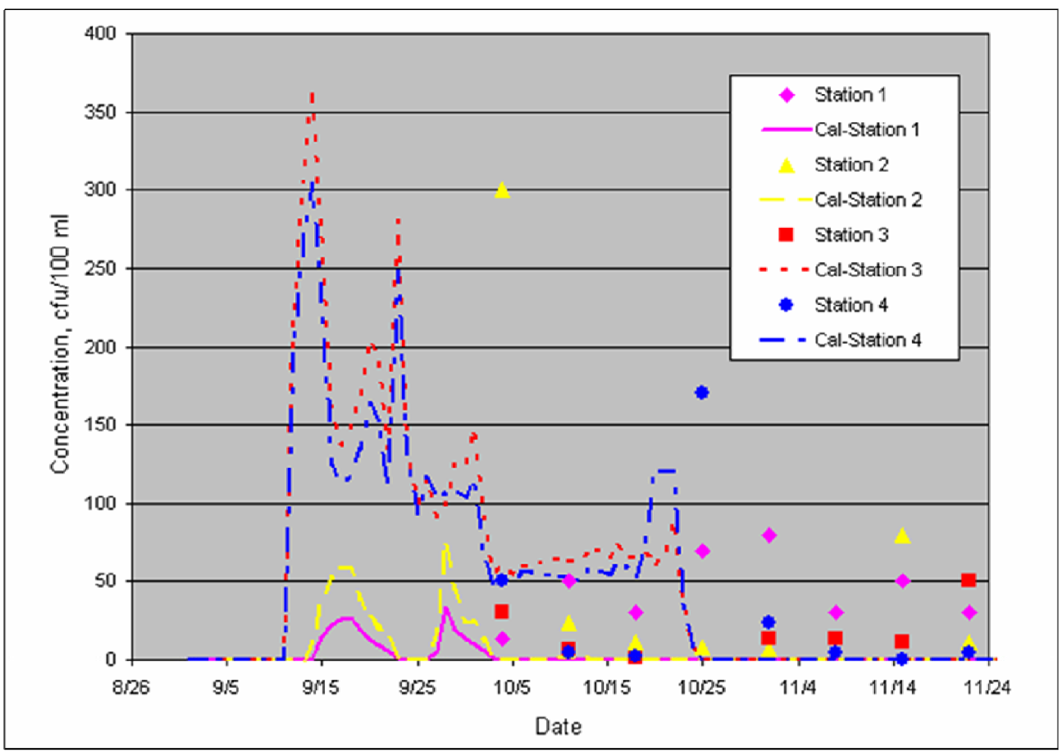

Figure 7. Model-computed (Cal-Station and solid lines) FCB concentrations for *actual* conditions and median loading concentration and measured (symbols) FCB concentrations following Katrina at four stations along the south shore of Lake Pontchartrain.

The U.S. Geological Service (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 2006), including those modeled. Measured BaP ranged between 17 and 290 µg/kg, where model computations for sediment BaP with *actual* conditions were approximately 40 µg/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 metals not included in the model (zero initial concentrations were input).

5 Scenario Results

The rather large uncertainty in loading concentrations should be recognized before interpreting model results or comparing results to protective benchmarks. Loading concentrations were based on statistical analysis of measured floodwater concentrations that were obtained over large spatial areas and a rather long time frame. In the analysis, non-detection values were set to half the detection limits rather than zero, which can substantially affect the estimated loading concentrations. It is also important to recognize that the models were started with zero initial sediment concentrations of contaminants. Thus, apparent large increase in sediment concentration due to a loading event may be miniscule relative to concentrations already in the prototype sediments resulting from years of accumulation from multiple events. For this reason, model results are often referred to herein as incremental maximum values, since the concentrations are actually the maximum incremental change in concentration above a background value of zero.

Lake Model

Three-dimensional models can generate voluminous output that can be viewed in a wide variety of formats, but two-dimensional concentration contour plots are one good way to view results. An example of this type of plot is illustrated in Figure 8, where maximum concentrations for the 90-day simulation are stored for every cell of the lake surface layer, then plotted as concentration contours. The results in Figure 8 are for arsenic with *actual* conditions and a median loading concentration of 20 μg/L. The third contour line from the top is 4 μg/L, which is a fivefold reduction in effluent concentration. The red color shading along the south shore is 12 μg/L or about half the effluent concentration.

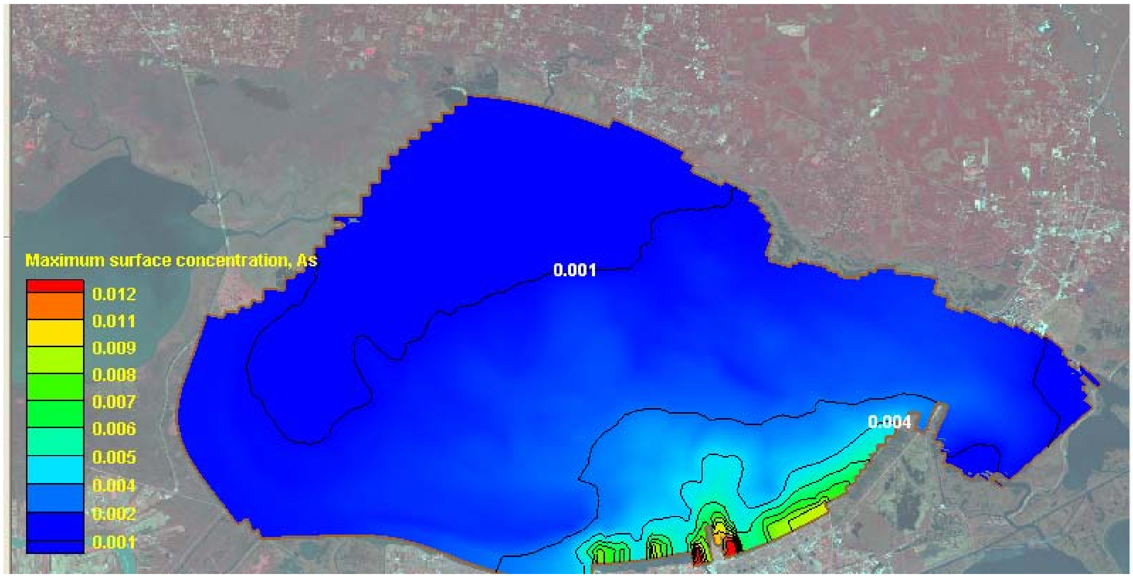

Figure 8. Maximum arsenic concentrations (µg/L total) in the surface layer of the Lake Pontchartrain model for *actual* conditions and median loading concentration.

The lake water quality model was executed for each scenario (*baseline* and *actual*) and for median and 95UCL loading concentrations. All five constituents were modeled, but not all could be included in the same model run since the model can presently handle only two contaminant constituents at a time. Incremental maximum concentrations in the model surface layer for the median loading concentrations were plotted and are provided in Appendix A for all five constituents and for both *baseline* and *actual* conditions. Similarly, for both scenario conditions, incremental maximum concentrations in surficial benthic sediments for the median loading concentrations are provided in Appendix A for the two metals and two organic chemicals.

Results for the 95UCL loading concentrations are not plotted since these plots would look similar to the median loading concentration results except that the concentration values along each contour would be increased in proportion to the product of the ratio of the 95UCL to median concentrations. However, the ratios of median and 95UCL loading concentrations are different for Orleans Metro and Orleans East for all constituents except BaP. Thus, the amount of change in the contour concentration depends on how close the contour is to each loading source and the source's change in loading concentration. The results can be used to estimate receiving water concentrations resulting from other loading concentrations if all source loading concentrations are adjusted by the same factor. For example, if someone wanted to determine the maximum lake concentration at a point of interest for 10 times the concentration used in the model for all sources, they would simply multiply the computed model concentration at that location times 10. Normally the same could be said for scaling the loading rate, but since the loading flow into the lake is a major component of the hydrodynamics, a linear scaling is not appropriate.

The effects of the levee failures on the lake environment can be related by comparing the figures in Appendix A for *actua*l and *baseline* conditions. In general, 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 larger total mass loadings of the *actual* conditions associated with the greater water volumes pumped. However, the pump discharge rates of the *baseline* conditions are at 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 (see Table 4) and even a larger impacted area for FCB; but as soon as pumping stops, the concentrations in the impacted area rapidly dissipate. This behavior occurs for the other constituents as well, as is evident by comparing the As results in Appendix A plotted for September 12 (Figures A19 and A20). The behavior is more apparent for FCB because of the higher concentrations.

The highest incremental maximum sediment concentrations tend to be located along the southeast shore of the lake, out from Orleans East, for both conditions (see Figures A11-A18). This is believed to be due to the currents and the shallow water in this area. More material can settle to the bottom in shallow water than in deep water. It should be noted that resuspension was set to zero, and resuspension can reduce sediment concentrations over time. However, it is doubtful that much resuspension and transport would occur during the 90-day simulation.

Table 4. Computed incremental maximum water (µg/l) and sediment (mg/kg) concentrations (total) for Lake Pontchartrain for *actual and baseline* conditions and median and 95UCL loading concentrations.

Condition	As water	As sed	BaP water	BaP sed	DDE water	DDE sed	Pb water	Pb sed	FCB water[1]
Actual	13	0.048	3.7	0.173	0.036	0.0024	3.7	0.053	1,055
Actual95	16	0.066	3.7	0.173	0.209	0.0171	25.4	0.384	42,214
Base	14	0.0052	3.7	0.014	0.037	0.000172	3.7	0.0062	1,413
Base95	14	0.0054	3.7	0.014	0.053	0.000598	32.1	0.051	44,780

[1]Units for FCB are cfu/100ml or MPN/100ml.
Note: Actual and Base are median loading concentrations, and Actual95 and Base95 are 95UCL loading concentrations.

From Table 4, it is apparent the maximum water concentrations for the *actual* conditions are about the same or a little less than those for the *baseline* conditions. This is because the *baseline* condition has a higher flow rate (due to more pumps operating at capacity) during pumping, which results in less time for settling of particulate matter and die-off of FCB, thus slightly greater water column concentrations. However, the maximum sediment concentrations for the *baseline* condition are roughly an order of magnitude less than those of the corresponding *actual* condition for all constituents, due to the fact that the sediment for the *baseline* condition has a much shorter exposure duration to constituents in the water column because the pumped loading period is much shorter.

Responses are not all linear with respect to loading concentrations, as expected. Linear response means that if the loading concentration doubles, then the corresponding water column and sediment concentrations also double, as long as the flow conditions do not change. However, if the loading flow doubles, then the corresponding concentrations do not necessarily double, since this system is flow dominated. The results in Table 4 do have a linear response for some constituents and conditions, such as for As with *actual* and *actual95* conditions/loadings, but others do not, especially for sediment concentrations. The nonlinear response may be due to the differences in median and 95UCL loading concentrations that differ by loading location (i.e., Orleans Metro and Orleans East) and the effects of the shallow waters along the shore of Orleans East.

Marsh Model

The marsh is dominated by the loadings; thus, the water concentrations rapidly reach a constant value and remain constant over the loading period, then rapidly drop when pumping and loading cease as shown in Figure 9 for arsenic with a fictional loading concentration of 1,000 µg/L. Sediment concentrations increase more gradually during loading, but then drop off gradually after loading ceases as shown in Figure 10. However, the results in Figure 10 are for a flushing rate equal to the pumping rate that continues after pumping ceases. Figure 11 shows results for the same conditions but using a flushing rate of 0.1 cms after pumping ceases and with peak concentrations of Figures 9 and 10 as initial conditions for the run that produced Figure 11. It can be seen by comparing Figures 10 and 11 that the two-step flushing procedure extends concentrations over time with higher peak sediment concentrations, which are considered to be more representative of what is expected to occur.

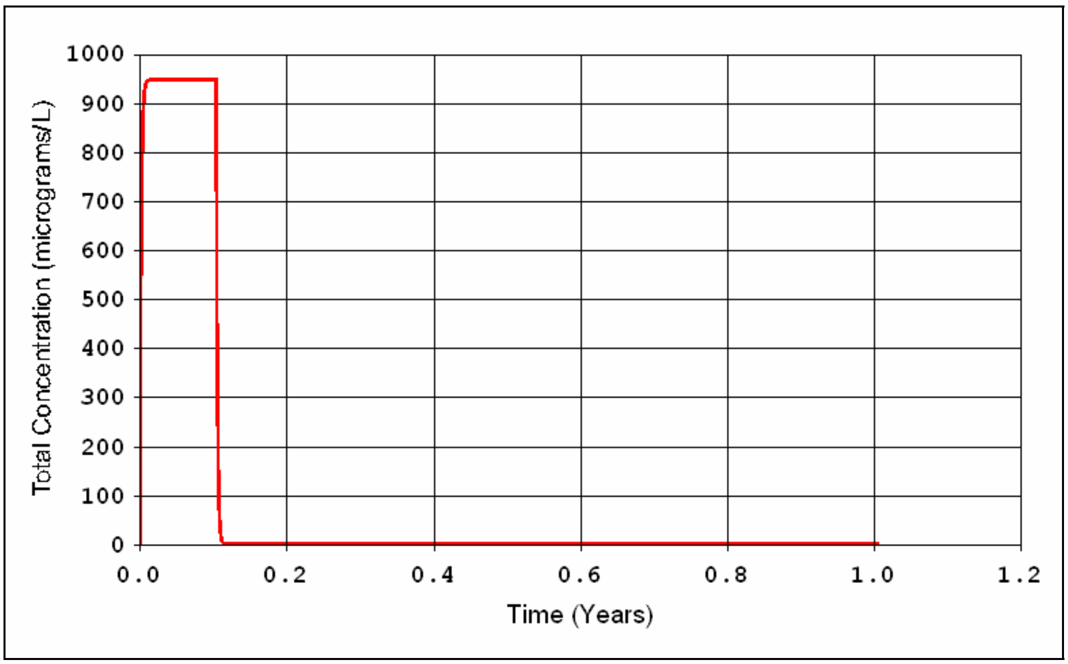

Figure 9. Computed arsenic concentrations (total) for water column of upper Violet Marsh for *actual* conditions using a pumped effluent concentration of 1,000 µg/L.

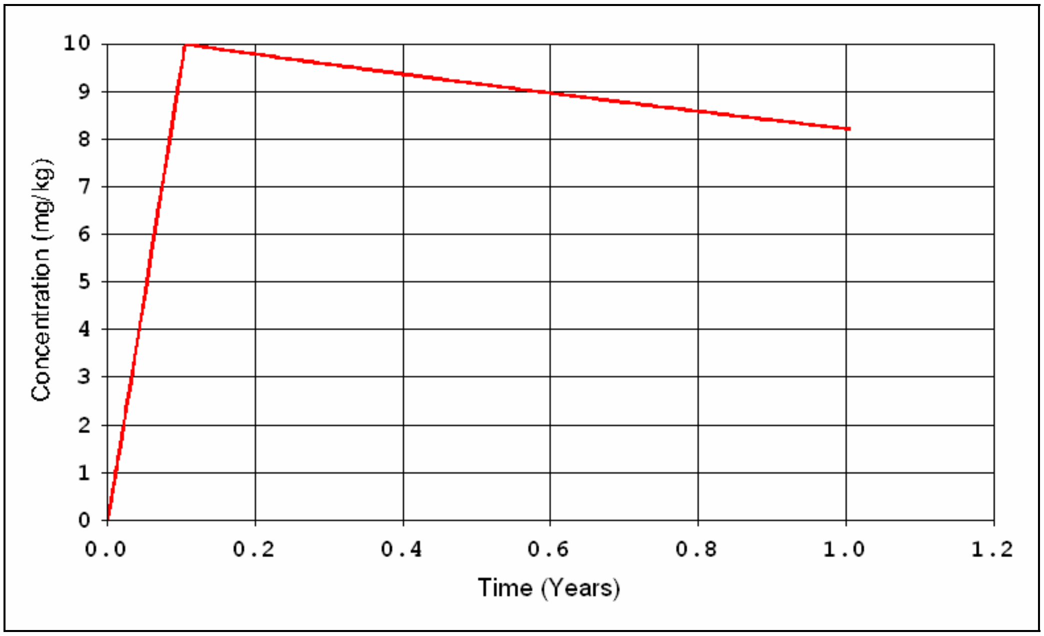

Figure 10. Computed arsenic concentrations (total) for benthic sediment of upper Violet Marsh for *actual* conditions using a pumped effluent concentration of 1,000 µg/L and background flushing equal to the pumped discharge flow.

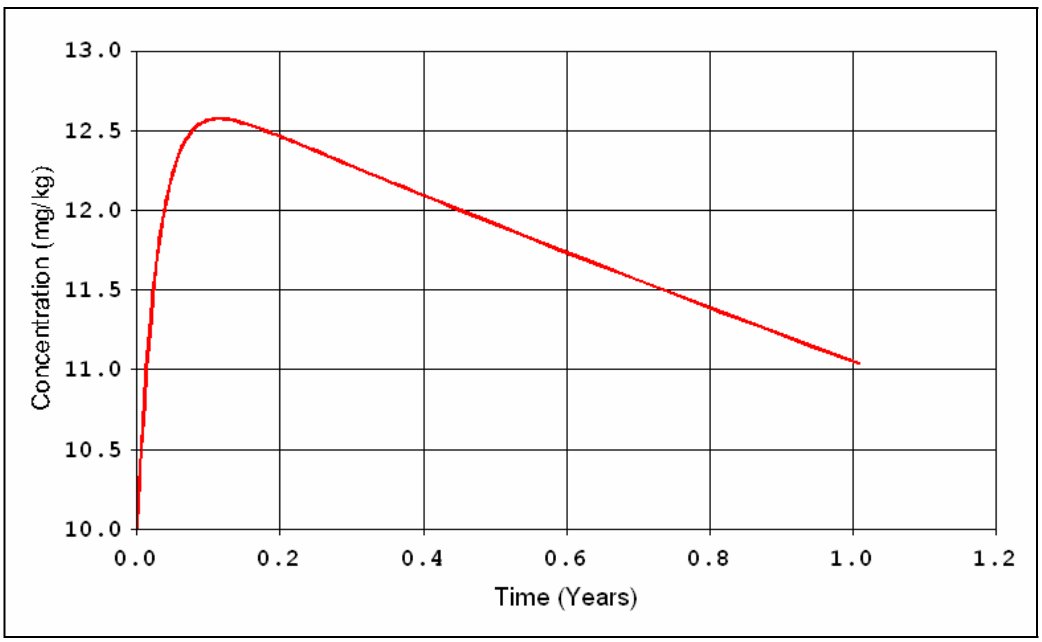

Figure 11. Computed arsenic concentrations (total) for benthic sediment of upper Violet Marsh for *actual* conditions using a pumped effluent concentration of 1,000 µg/L and background flushing flow of 0.1 cms following pumping cessation.

Marsh model results for incremental peak water and sediment concentrations (total) are presented in Table 5 for both conditions and for the loading concentrations shown in Table 3. Peak sediment concentrations were obtained from the runs with the low background flushing rate after pump cessation.

Table 5. Computed incremental maximum water (µg/l) and sediment (mg/kg) concentrations (total) for upper Violet Marsh for *actual and baseline* conditions and median and 95UCL loading concentrations.

Condition	As water	As sed	BaP water	BaP sed	DDE water	DDE sed	Pb water	Pb sed	FCB water[1]
Actual	11.5	0.15	3.5	0.28	0.022	0.003	1.95	0.111	55.0
Actual95	13.4	0.18	3.5	0.28	0.044	0.006	3.82	0.22	1040
Base	11.5	0.038	4.2	0.026	0.032	0.00024	2.2	0.012	69
Base95	13.4	0.044	4.2	0.026	0.065	0.00048	4.3	0.023	1310

[1] Units for FCB are cfu/100 ml or MPN/100 ml.

Note: Actual and Base are median loading concentrations, and Actual95 and Base95 are 95UCL loading concentrations.

Several interesting features can be observed from Table 5. One feature is that the *baseline* condition results in maximum water concentrations that are either equal to or slightly greater than those for the corresponding *actual* condition. The reason for this is that the *baseline* condition has a higher flow rate through the system (due to more pumps operating at capacity) during pumping, which results in less time for settling of particulate matter and die-off for FCB, thus slightly greater water column concentrations. However, the maximum 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 fact that the sediment for the *baseline* condition has a much shorter exposure duration to constituents in the water column because the pumped loading period is much shorter.

Maximum sediment concentrations at the end of the initial runs (i.e., background flushing flow equal to pumped flow) are close to the maximum sediment concentrations for the subsequent runs (i.e., background flushing flow set to 0.1 m^3/sec) for the *actual* conditions; however, for the *baseline* condition, the sediment concentrations increased substantially above the initial concentrations during the subsequent runs. This is due to the short duration of initial loading relative to the follow-on settling period associated with the *baseline* condition.

Responses are linear for all conditions and loadings; for example, if the median loading concentration doubles, then the corresponding water column and sediment concentrations also double. However, if the loading flow doubles, then the corresponding concentrations do not necessarily double since the system is flow dominated. Thus, the results in Table 5 can be easily extended to other loading conditions (i.e., loading concentrations) as long as the loading discharges and durations and background flows do not change.

Comparisons to Protective Benchmarks

Dissolved water concentrations were needed for comparison to water quality criteria, 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 total concentrations in water in Tables 4 and 5. The dissolved concentrations are reported in Table 6 for each constituent along with the dissolved fractions.

Table 6. Dissolved fractions in the water column (F_{dw}) for each constituent and computed maximum water (µg/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
Lake				
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
Marsh				
Actual	11.4	3.11	0.018	1.81
Actual95	13.3	3.11	0.035	3.55
Base	11.4	3.74	0.026	2.05
Base95	13.3	3.74	0.052	4.0

The maximum dissolved water column concentrations and maximum sediment concentrations were compared with ecologically protective water quality criteria and sediment screening values shown in Table 7. U.S. Environmental Protection Agengy (USEPA) (1986) recommended primary contact protective limits for FCB of 400 MPN/100 ml for a single sample.

Table 7. Screening-level ecologically protective benchmarks.

Criteria	As	BaP	DDE	Pb
EPA, water (μg/L dissolved)	36[1]	300[2]	14[3]	4.0[5]
LA, water (μg/L dissolved)	36[1]	NA	0.14[4]	1.2[5]
Sediment (mg/kg dry)[6]	5.9	0.0319	0.00142	35.0

[1] chronic, marine
[2] acute, marine
[3] acute, marine
[4] chronic, marine
[5] chronic, fresh, adjusted for hardness.
[6] freshwater TEL

Lake

The computed maximum water concentrations for As and BaP were less than the EPA and LA water quality criteria for both conditions and both loading concentrations. The computed maximum water column concentrations for Pb exceeded the LA criteria for both conditions and both loading concentrations, and the concentrations for DDE exceeded the LA criteria for both conditions with 95UCL loading concentrations. Maximum concentrations for FCB in water exceeded EPA criteria for both conditions and both loading concentrations, but FCB exceedence frequently occurs during stormwater dewatering (Jin et al. 2004) of New Orleans.

The computed maximum sediment concentrations for As and Pb were less than the sediment screening criteria 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 criteria for both loading concentrations under the *actual* conditions.

In summary, the model indicated that the only degradation resulting from *actual* conditions as compared to *baseline* conditions most likely occurs for organic contaminants in relatively small areas of bottom sediments near the south shore. Any contaminant concentrations above water quality standards for the water column (e.g., FCB and Pb) can occur for almost any dewatering condition, with or without levee failures.

Marsh

The computed maximum water concentrations for As, BaP, and DDE were less than the EPA and LA water quality criteria for both conditions and both loading concentrations. The computed maximum water column concentrations for Pb exceeded the LA standards for both conditions and both loading concentrations and equaled the EPA standard for the Base95 condition. Maximum concentrations for FCB in water exceeded EPA criteria for both conditions and the 95UCL loading concentrations, but were below the criteria for both conditions and the median loading concentrations. This result is different from the lake results because the FCB loading concentrations for the lake were considerably higher than for the marsh (see Tables 2 and 3).

The computed maximum sediment concentrations for As and Pb were less than the sediment screening criteria 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 criteria for both loading concentrations under the *actual* conditions.

In summary, the model indicated that the only degradation resulting from *actual* conditions as compared to *baseline* conditions most likely occurs for organic contaminants in the sediment of upper Violet Marsh. Any contaminant concentrations above water quality standards for the water column (e.g., FCB and Pb) can occur at times for almost any dewatering condition, with or without levee failures.

6 Discussion

The lake and marsh respond in a similar manner to loadings. However, the marsh tends to have a greater sediment response to loadings than does the lake due to the marsh being a confined system with the loadings being the only flow in the system. Also, lake and marsh concentrations differ due to differences in loading concentrations, such as for Pb.

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 multiple events. Added to the uncertainties in the loading concentrations for *actua*l 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.

Increases in lake sediment metal concentrations computed by the model for *actual* conditions are very small compared with background concentrations for some contaminants. Pre-Katrina mean sediment concentrations were 7.02 parts per million (ppm) and 17.5 ppm for As and Pb, respectively (Penland et al. 2002); thus, the maximum increase in sediment concentrations computed by the model for *actual* conditions and median loading concentrations was 0.7 percent for As and 0.3 percent for Pb of the pre-Katrina (background) values. 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), thus confirming that dewatering of New Orleans following Hurricanes Katrina and Rita had little impact on lake sediment concentrations of metals as shown by the model. 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). Sediment concentrations computed by the model for actual conditions and median loading concentrations 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 concentration increase to measured post-Katrina concentration seems reasonable as compared with the results for BaP and the metals.

The greatest area for improvement in the model would be to obtain a better representation of pump flow rates for the *baseline* scenario. The present *baseline* scenario approach ignores any water entering the city by levee overtopping, whereas data indicate that overtopping would have occurred even if the levees had functioned fully as designed. The second highest priority for model improvement should focus on obtaining water quality measurements in stormwater under normal, *baseline* conditions with the levees functioning as designed. The assumption was made for modeling that stormwater and floodwater concentrations were the same under *baseline* and *actual* conditions, an assumption that is highly questionable due to limited measured water quality data for normal dewatering operations. Additionally, given more time and funding, it would be good to conduct additional model calibration/validation for the lake model.

At one point early in this study, consideration was given to trying to estimate the source terms that resulted in 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 PAHs and metals in the urban area of 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 sources. Flooding and subsequent dewatering resulted in exposing the New Orleans environment to these contaminants. However, such exposure occurs even during normal (*baseline*) dewatering, but to a lesser degree, due to less stormwater and shorter pumping durations.

In retrospect, the use of 95UCL loading concentrations and sampling of the model maximum water and sediment concentrations may have been an excessively conservative approach. A better approach would have been to use a statistical distribution of loading concentrations observed in the floodwaters and then process the output distribution to determine the 95UCL water and sediment concentrations. However, this approach would have required many more computer runs and post-processing, which would have required substantially more time and funding.

7 Conclusions

Models were applied to Lake Pontchartrain and upper Violet Marsh to determine the consequences of dewatering the floodwaters of Hurricane Katrina, with or without levee failure. Results can be summarized as follows:

- Incremental increases in lake and marsh sediment concentrations of contaminants as a result of the *actual* dewatering event are about an order of magnitude greater than *baseline* removal of stormwater without levee failure, but increases relative to background concentrations were small for metals and long-life organic compounds.
- Model-computed maximum sediment concentrations for the organic chemicals BaP and DDE exceeded ecologically protective sediment quality criteria, whereas concentrations of these chemicals did not exceed these criteria following removal of stormwater for the same event but without levee failure or overtopping. However, it should be recognized that the sediment area that exceeded sediment quality criteria is relatively small for the lake and isolated to areas near the southeast shore.

Other water quality impacts, such as elevated concentrations of FCB in water, occur in the lake regardless of dewatering conditions (i.e., with or without levee failures). Elevated FCB concentrations may or may not occur in the marsh, depending on pump effluent concentrations, with little or no dependence on dewatering conditions. 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 those that are operating may be functioning below capacity. Lower pump discharge rates can result in lower water concentrations due to greater residence times in ambient waters with greater opportunity for settling and dilution. The reason that incremental changes in sediment concentrations are expected to be higher with levee failures is that more floodwater volume must be removed, thus, dewatering takes longer and much more contaminant mass is discharged to receiving waters, which is manifested as larger increases in sediment concentrations.

Some constituents, such as Pb, may present water quality concerns under any dewatering conditions regardless of the levee failures and overtopping or not. Maximum water concentrations of Pb computed by the models for the lake and marsh exceeded ecologically protective water quality criteria used herein for both *actual* and *baseline* conditions. Elevated concentrations of metals and PAHs existed in urban New Orleans soils before Katrina. Thus, the presence of these constituents is expected for both urban floodwater and urban stormwater runoff and in the subsequent pumped effluents, with or without levee failures/overtopping.

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Appendix A: Contour Plots of Computed Incremental Maximum Water Concentrations in Surface Layer and Incremental Maximum Benthic Sediment Concentrations for Lake Pontchartrain

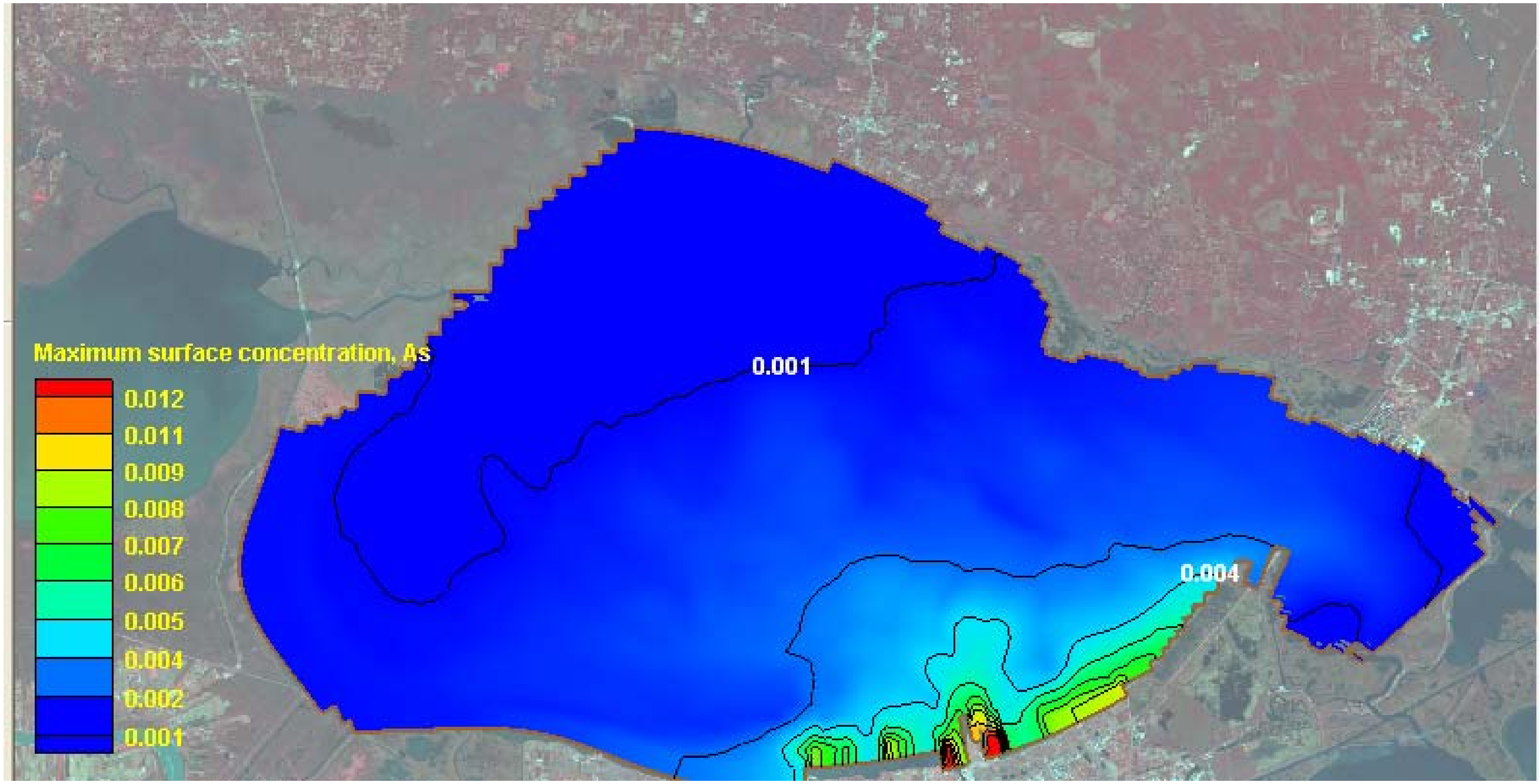

Figure A1. Incremental maximum As water surface concentrations (mg/L total) in Lake Pontchartrain for *actual* conditions, median loading concentrations.

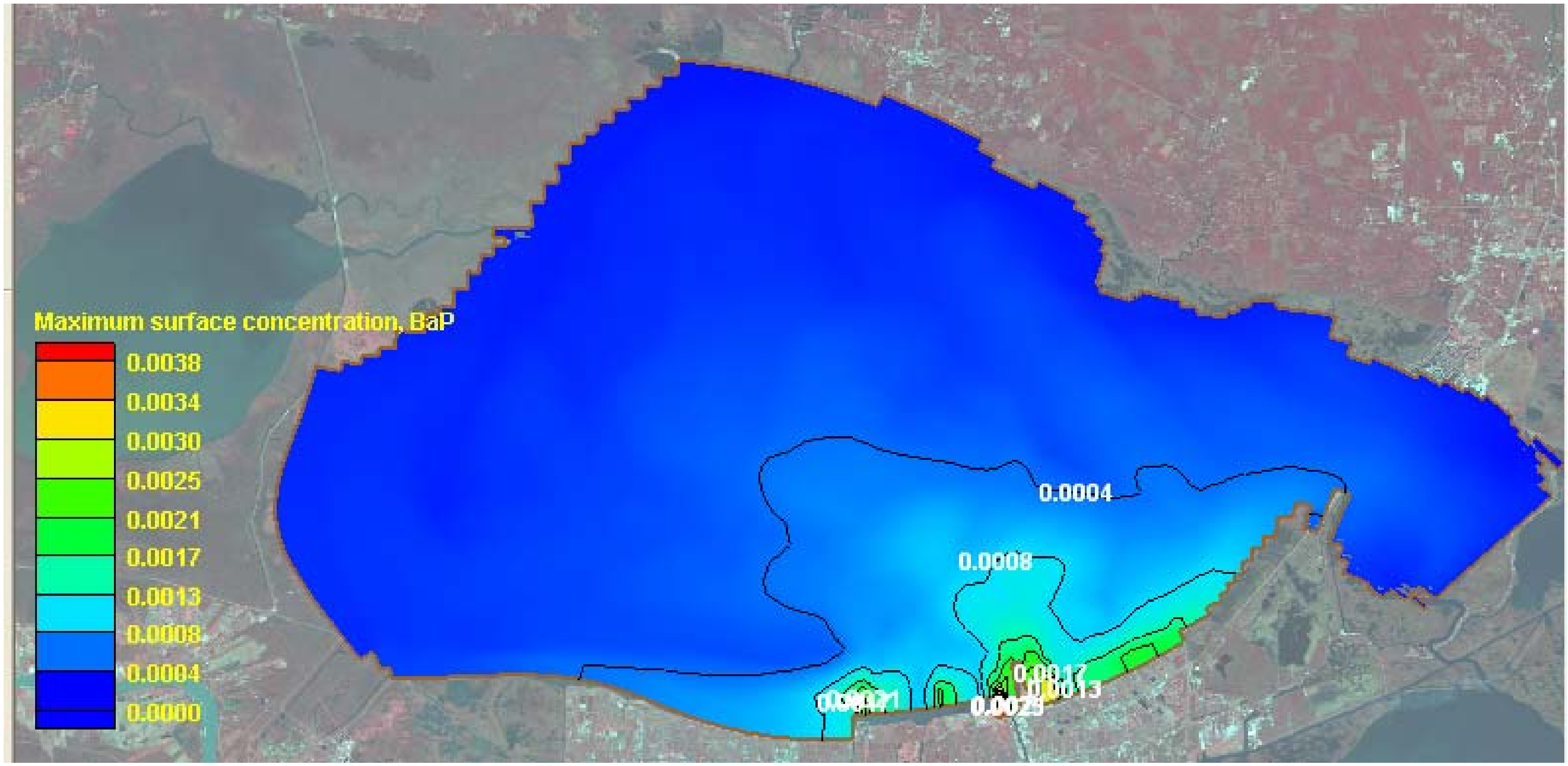

Figure A2. Incremental maximum BaP water surface concentrations (mg/L total) in Lake Pontchartrain for *actual* conditions, median loading concentrations.

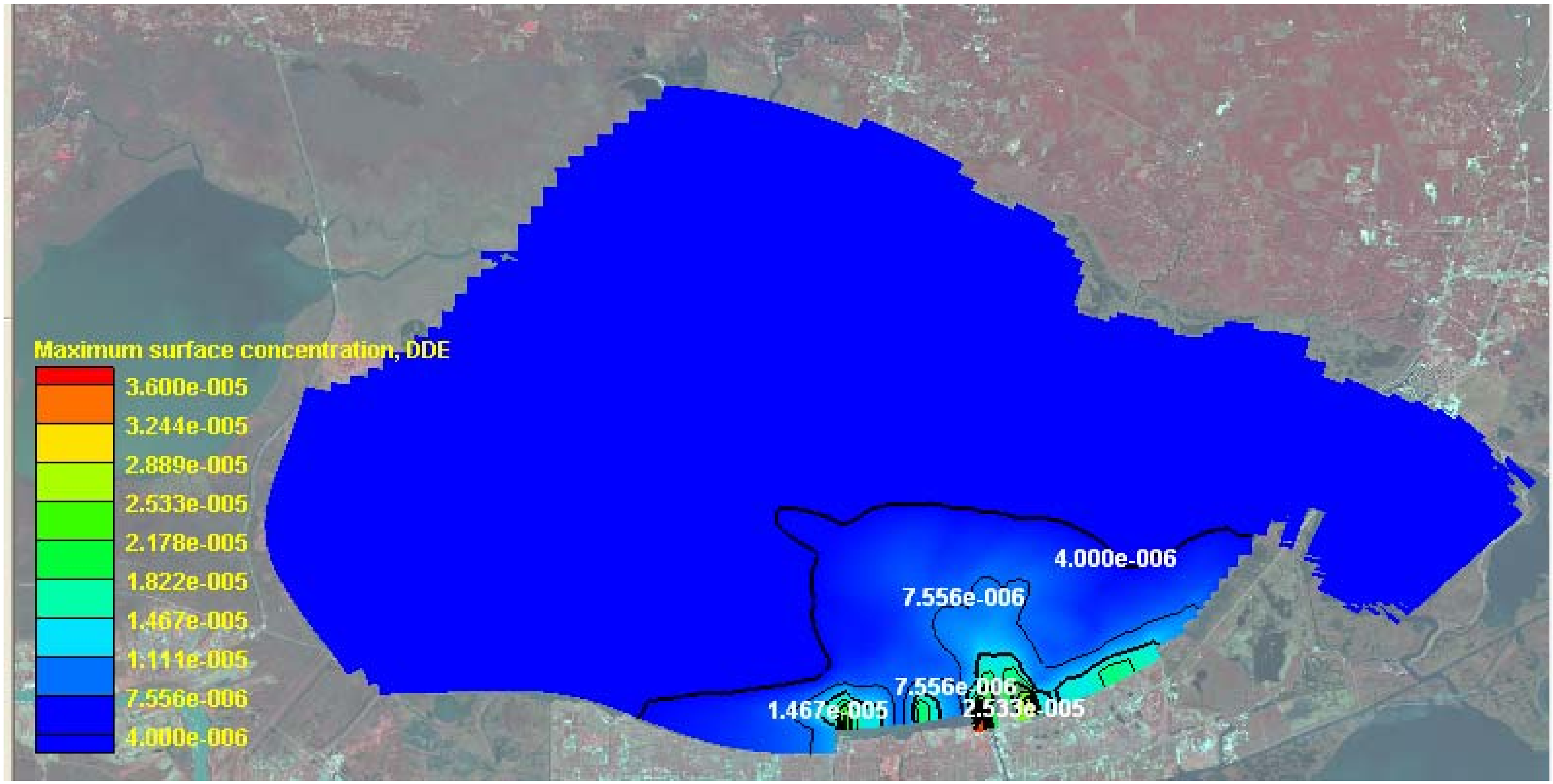

Figure A3. Incremental maximum DDE water surface concentrations (mg/L total) in Lake Pontchartrain for *actual* conditions, median loading concentrations.

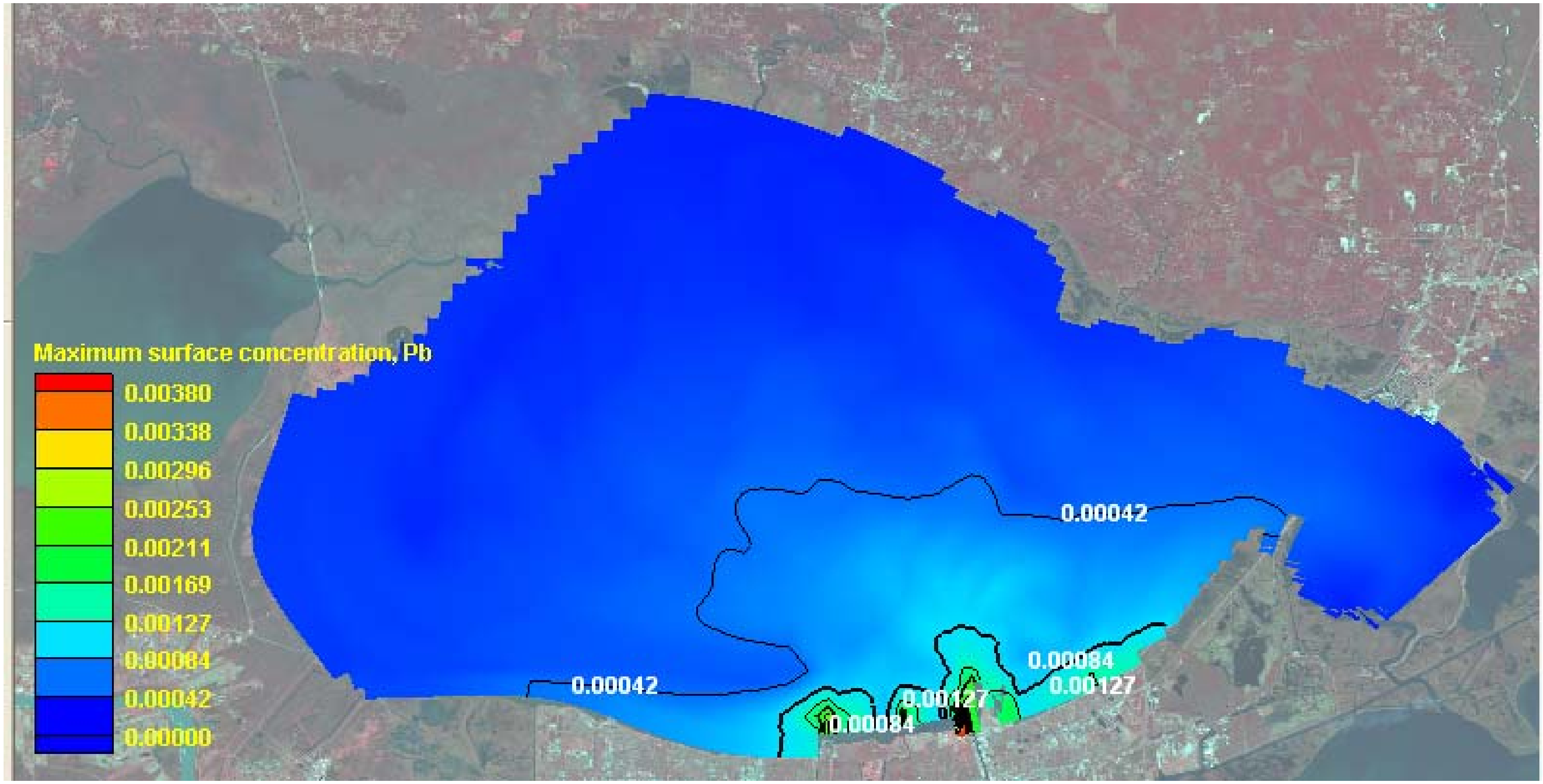

Figure A4. Incremental maximum Pb water surface concentrations (mg/L total) in Lake Pontchartrain for *actual* conditions, median loading concentrations.

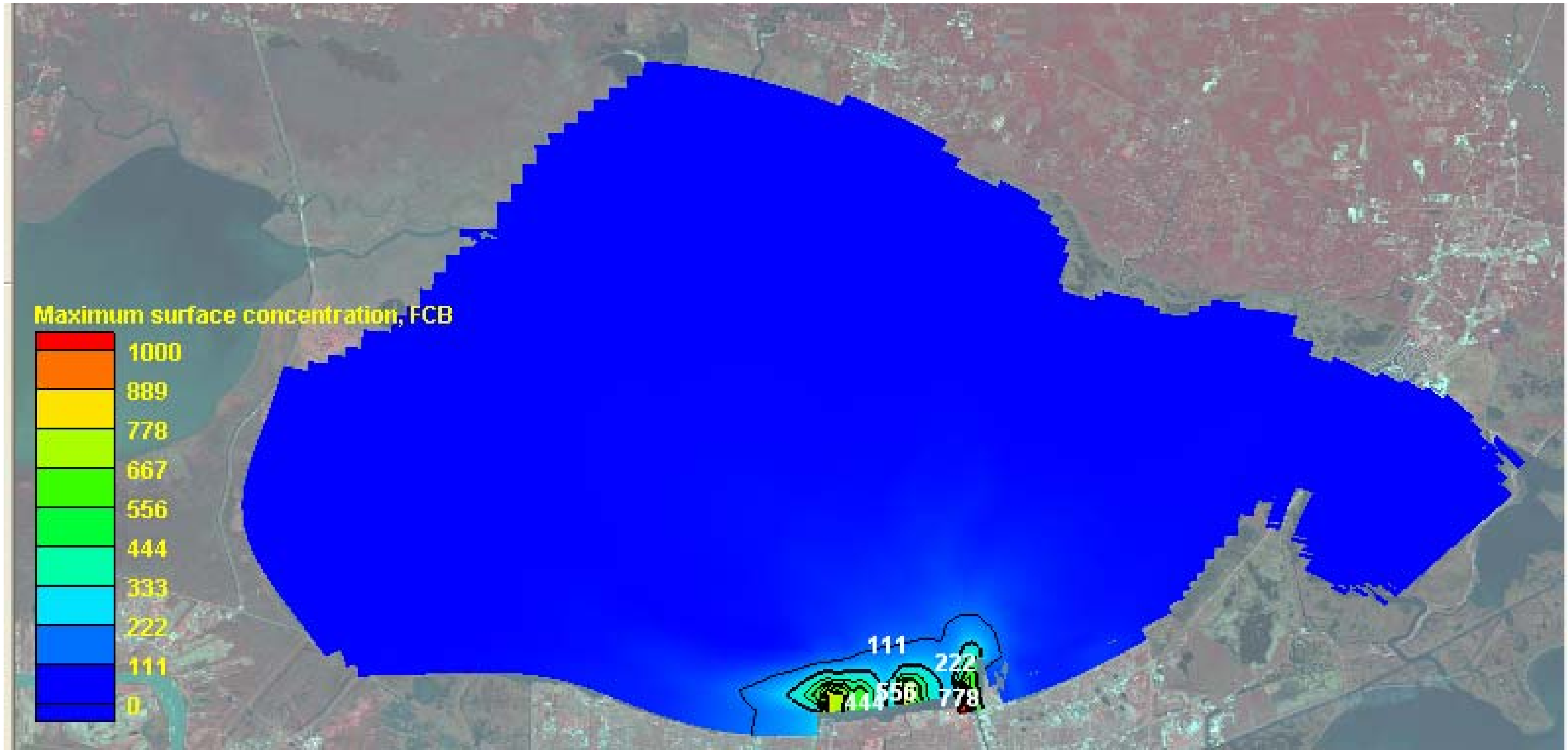

Figure A5. Incremental maximum FCB water surface concentrations (cfu/100ml) in Lake Pontchartrain for *actual* conditions, median loading concentrations.

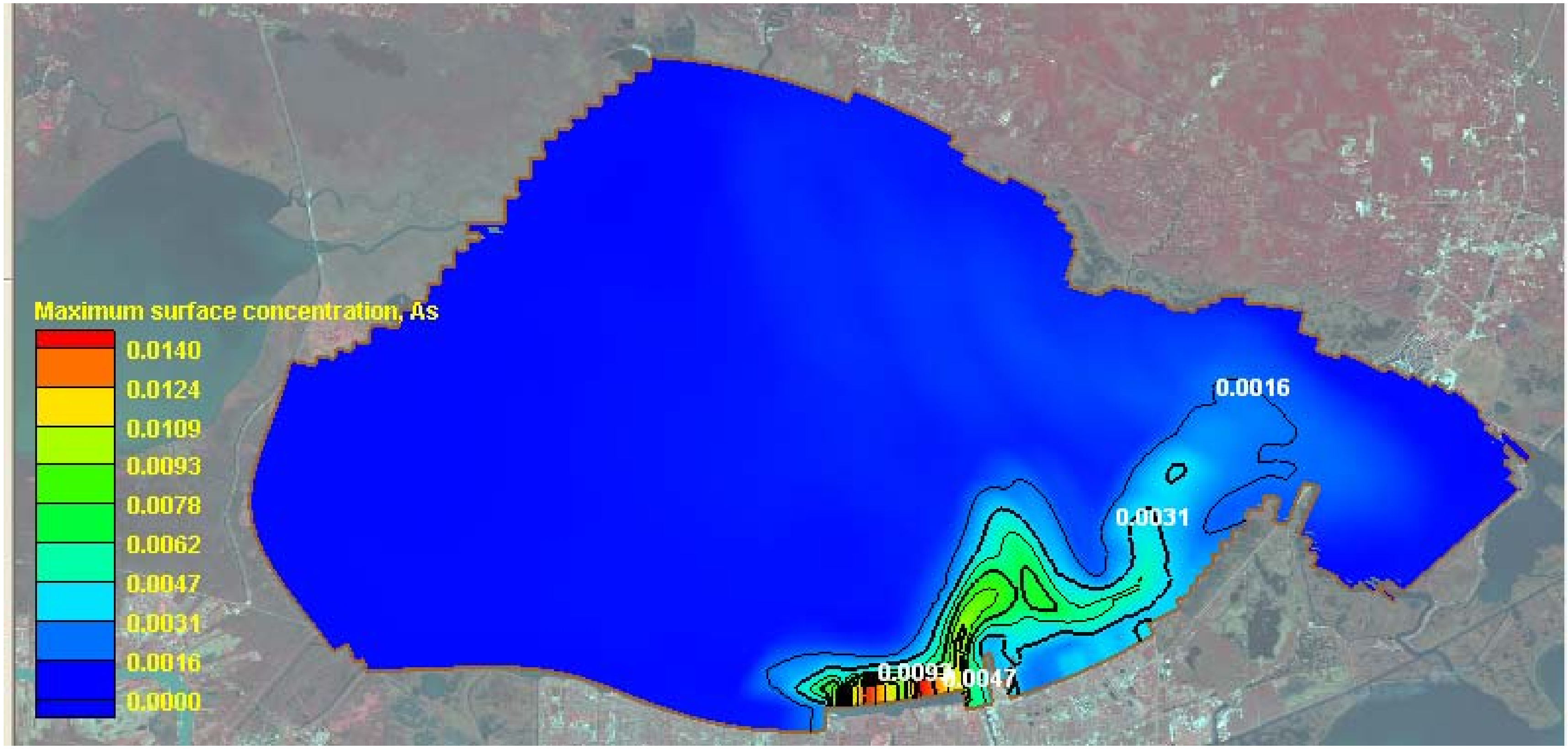

Figure A6. Incremental maximum As water surface concentrations (mg/L total) in Lake Pontchartrain for *baseline* conditions, median loading concentrations.

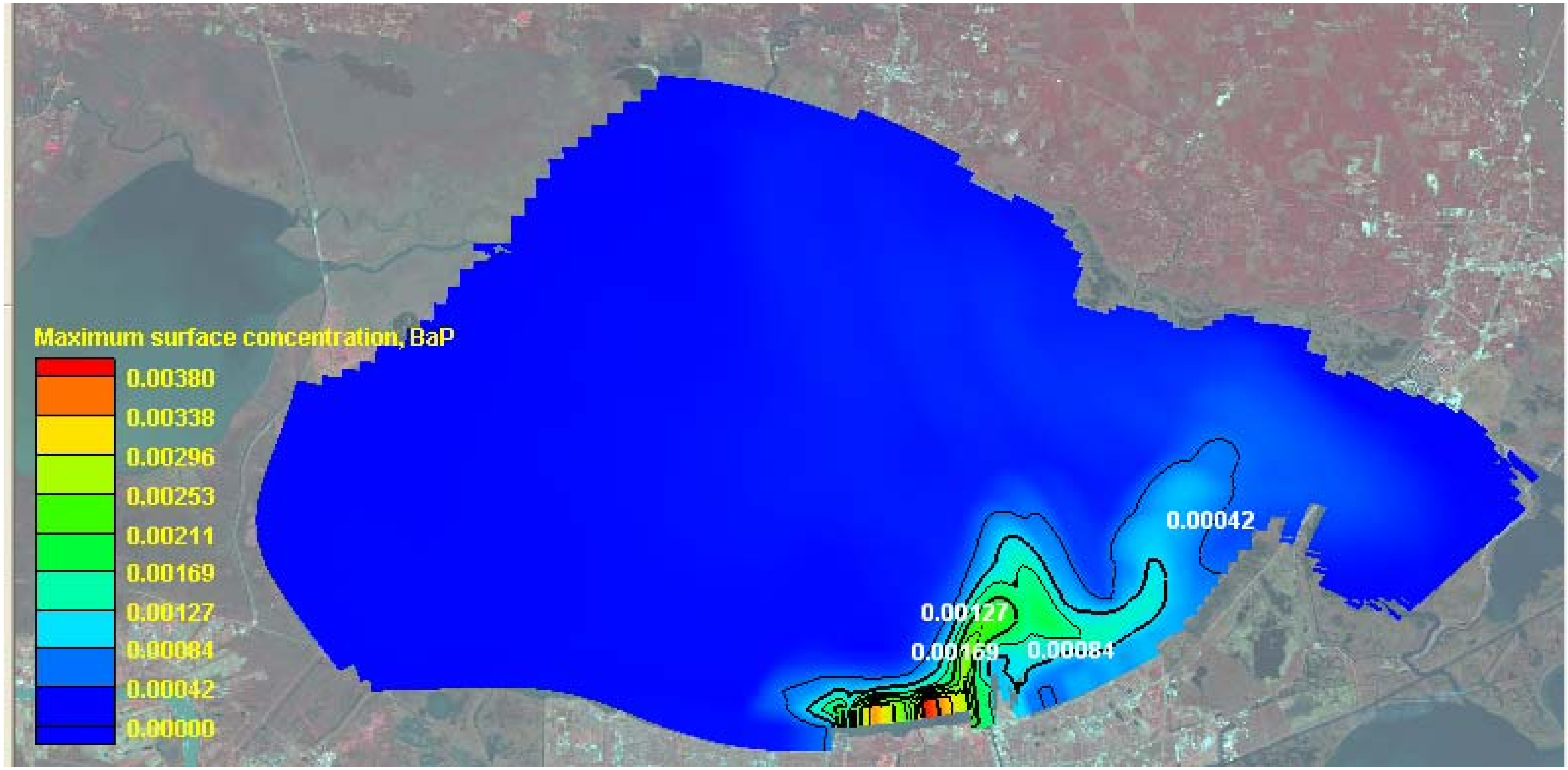

Figure A7. Incremental maximum BaP water surface concentrations (mg/L total) in Lake Pontchartrain for *baseline* conditions, median loading concentrations.

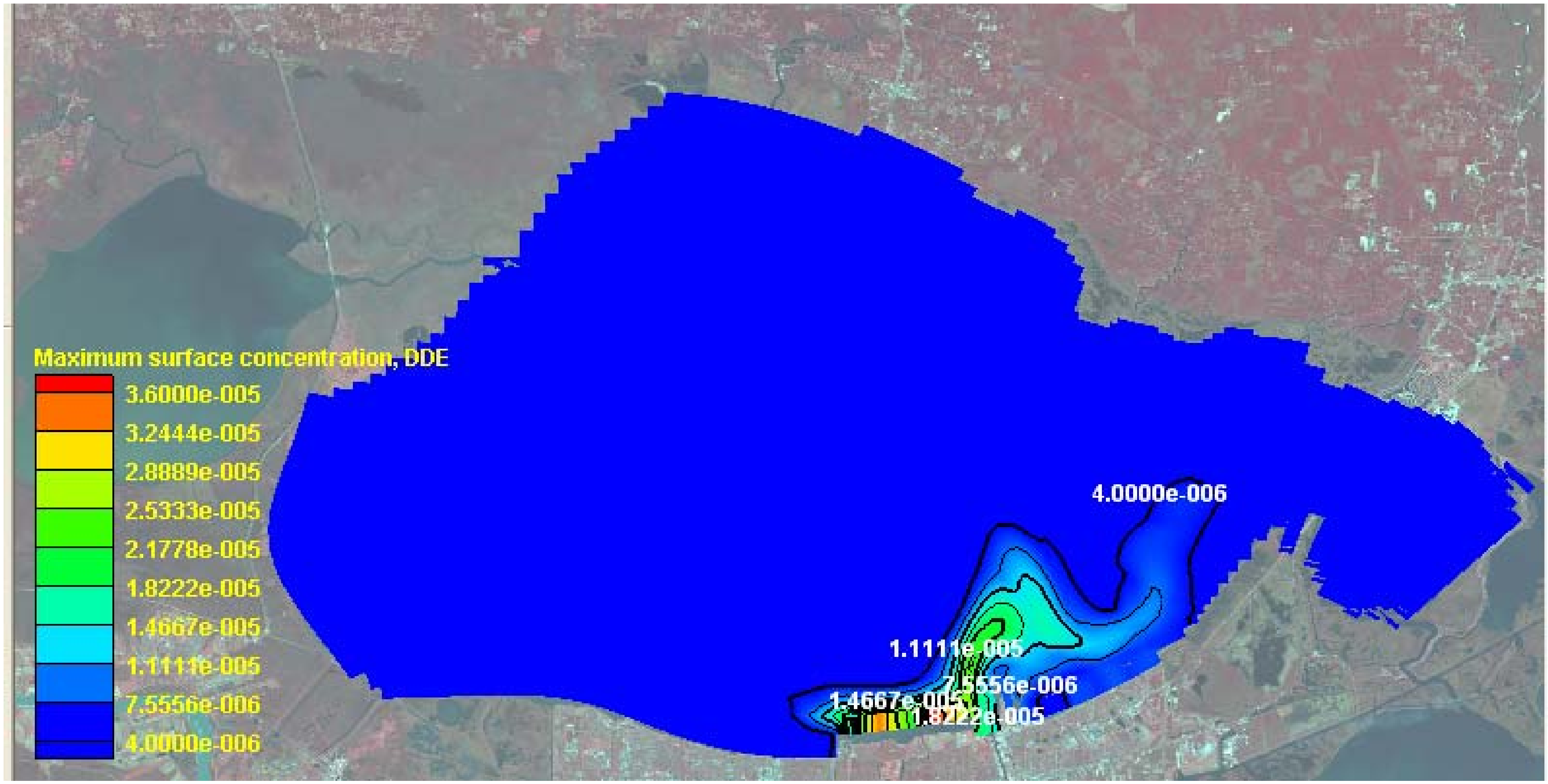

Figure A8. Incremental maximum DDE water surface concentrations (mg/L total) in Lake Pontchartrain for *baseline* conditions, median loading concentrations.

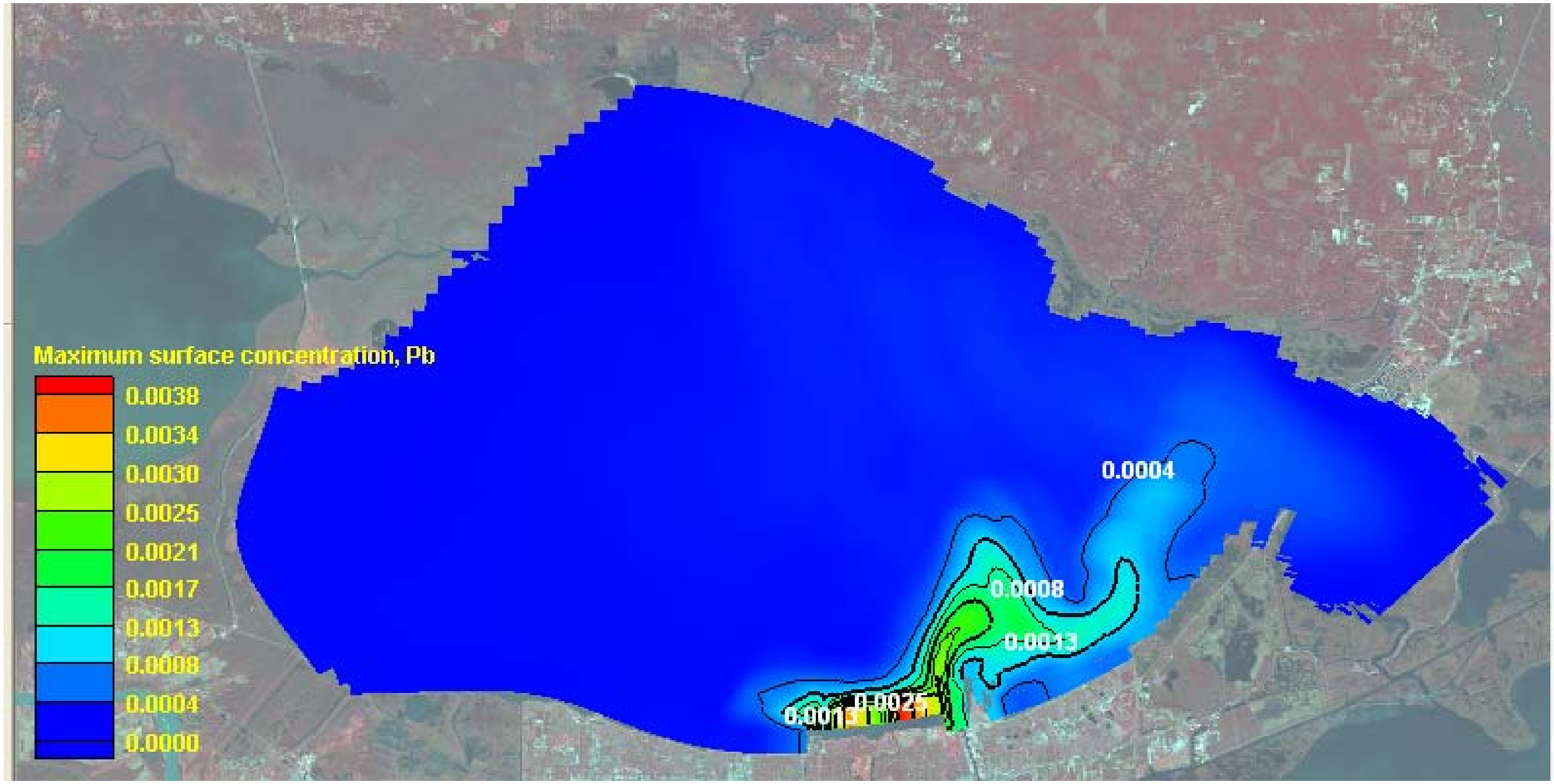

Figure A9. Incremental maximum Pb water surface concentrations (mg/L total) in Lake Pontchartrain for *baseline* conditions, median loading concentrations.

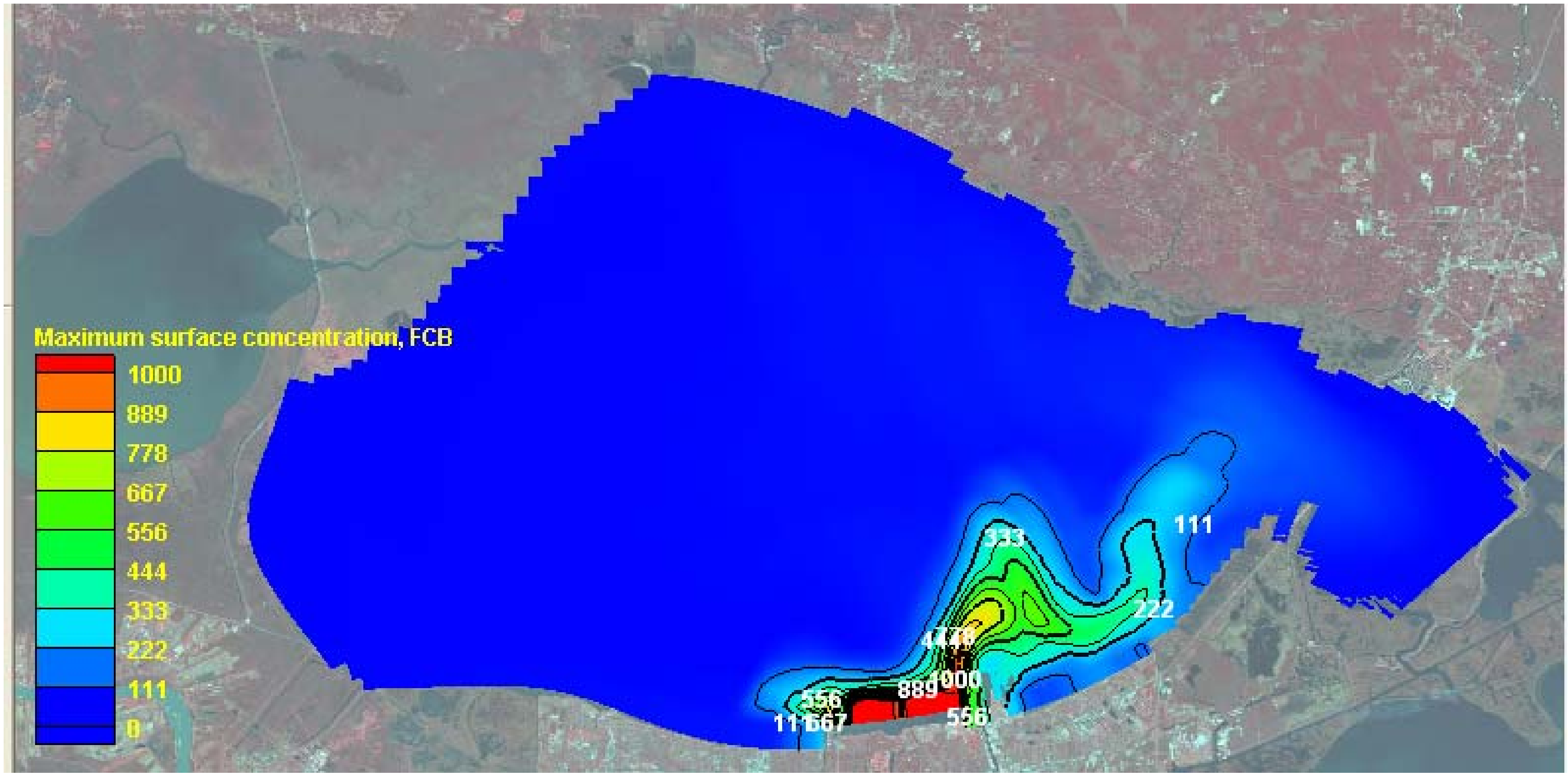

Figure A10. Incremental maximum FCB water surface concentrations (cfu/100ml) in Lake Pontchartrain for *baseline* conditions, median loading concentrations.

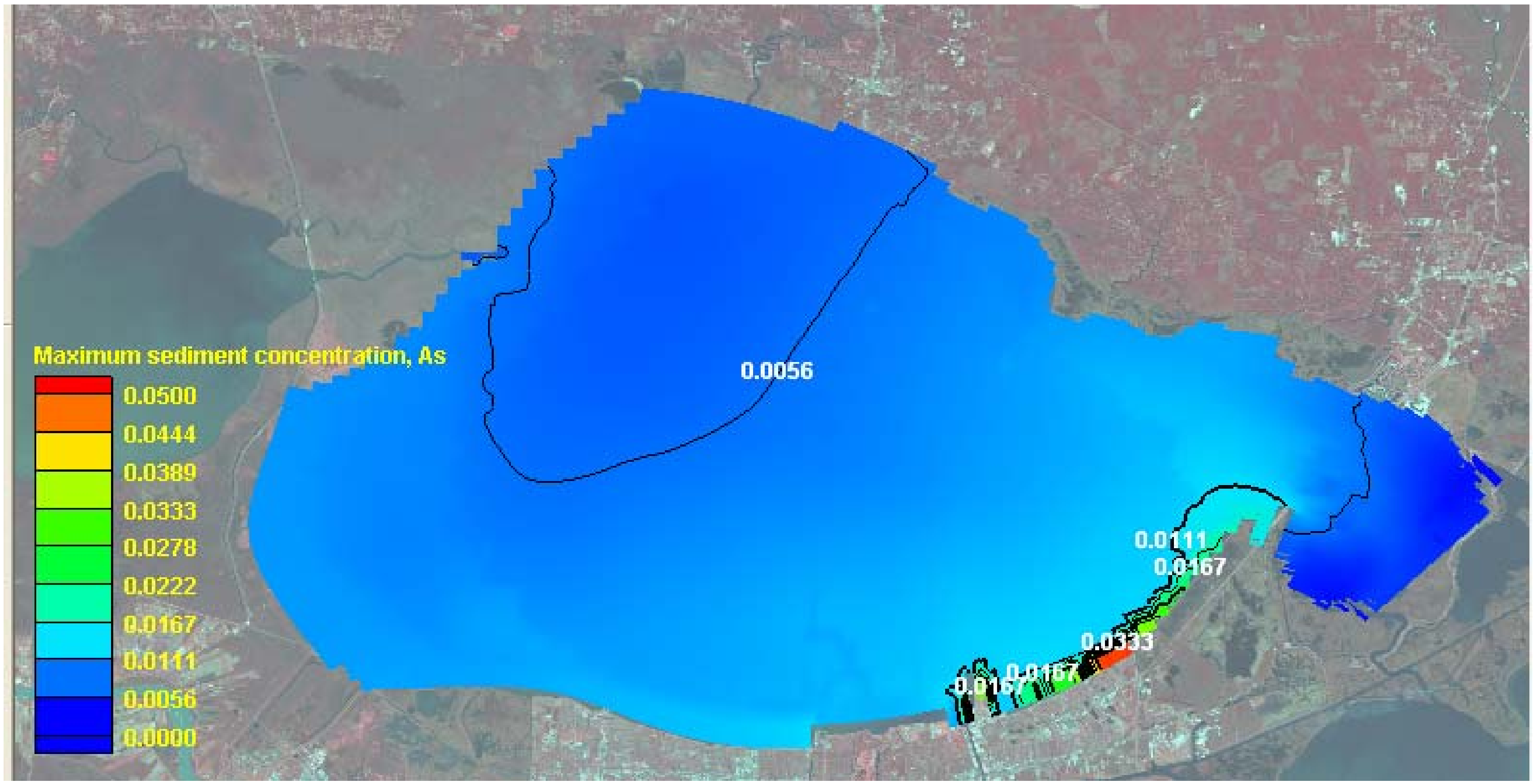

Figure A11. Incremental maximum As sediment concentrations (mg/kg total) in Lake Pontchartrain for *actual* conditions, median loading concentrations.

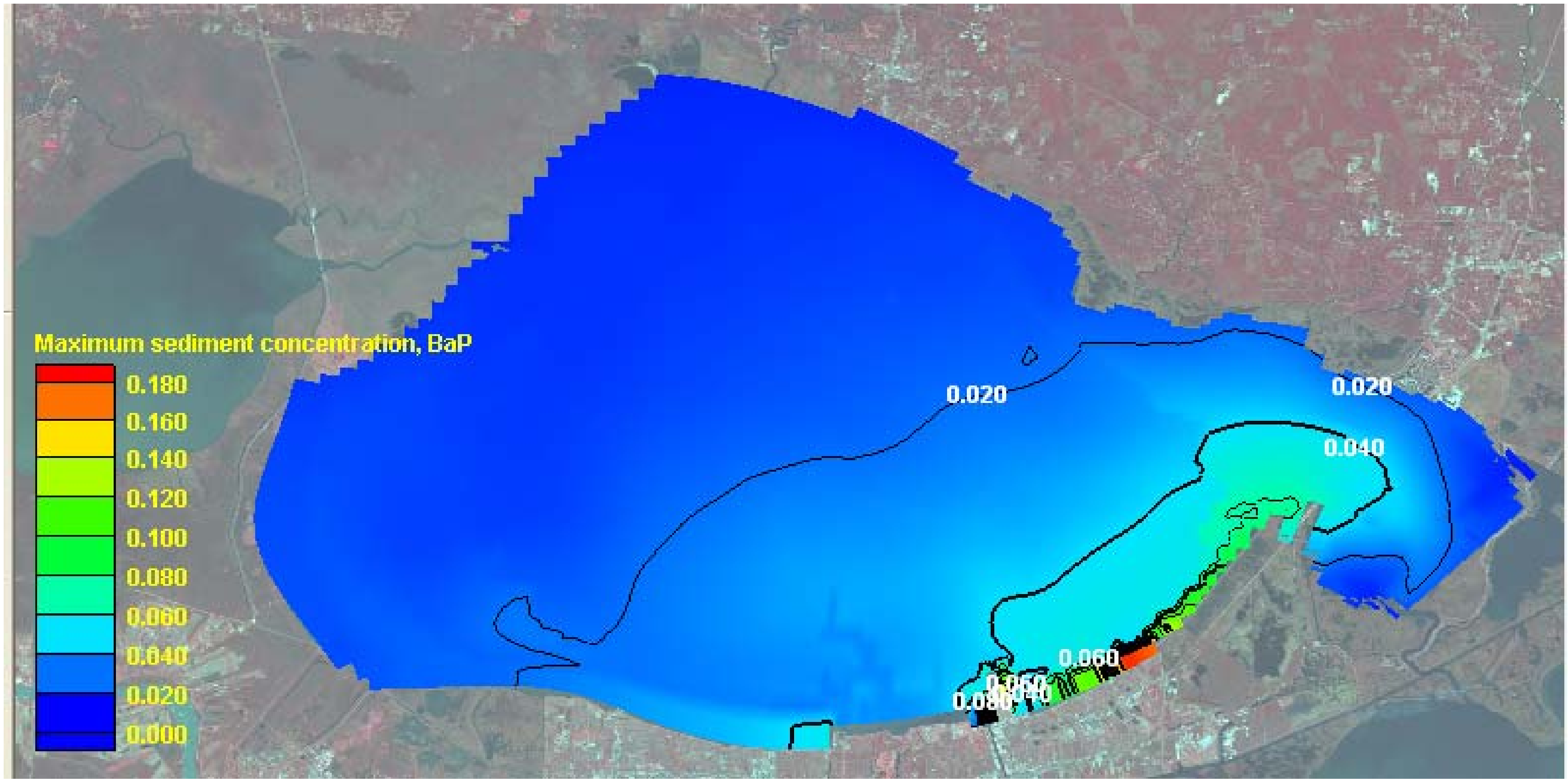

Figure A12. Incremental maximum BaP sediment concentrations (mg/kg total) in Lake Pontchartrain for *actual* conditions, median loading concentrations.

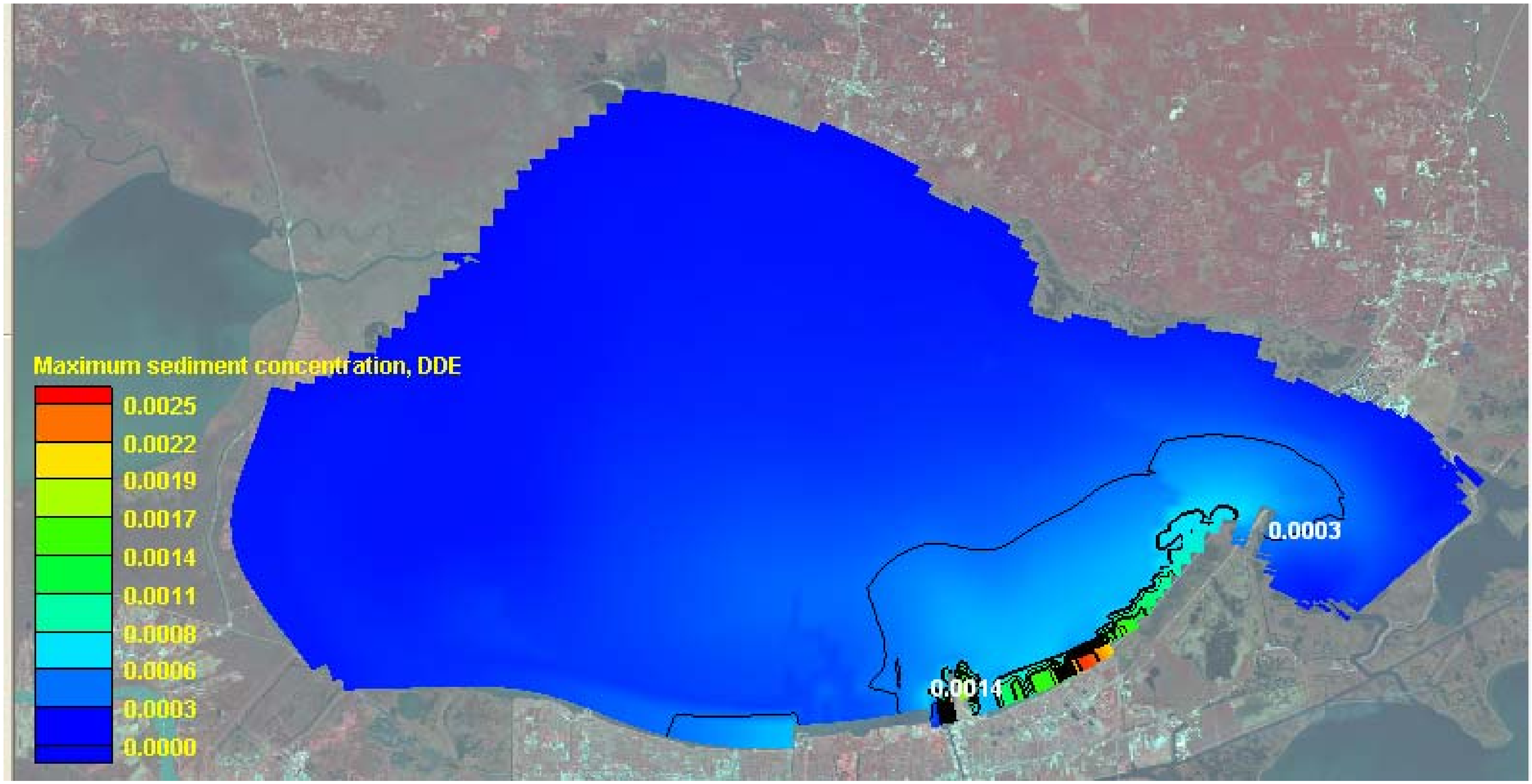

Figure A13. Incremental maximum DDE sediment concentrations (mg/kg total) in Lake Pontchartrain for *actual* conditions, median loading concentrations.

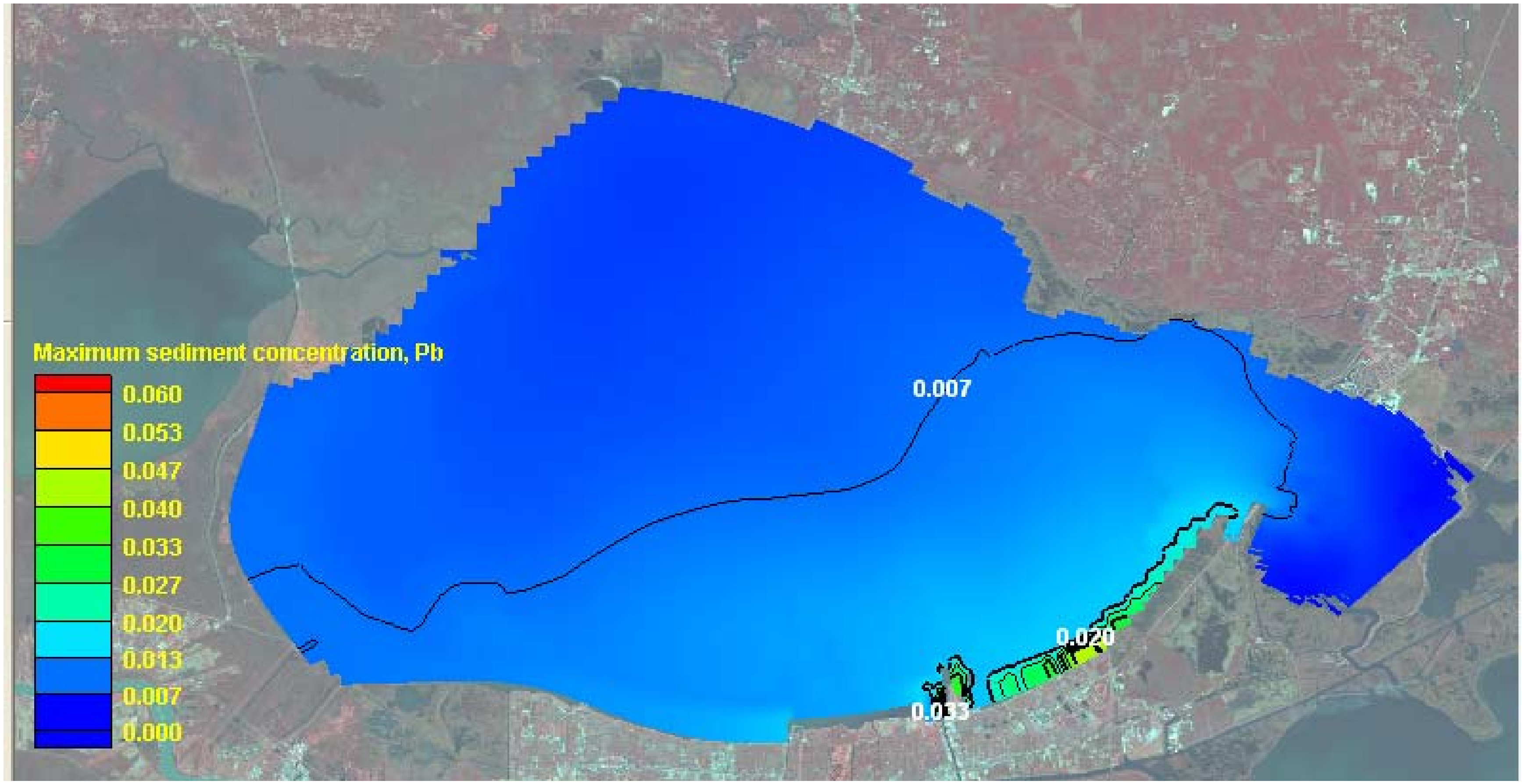

Figure A14. Incremental maximum Pb sediment concentrations (mg/kg total) in Lake Pontchartrain for *actual* conditions, median loading concentrations.

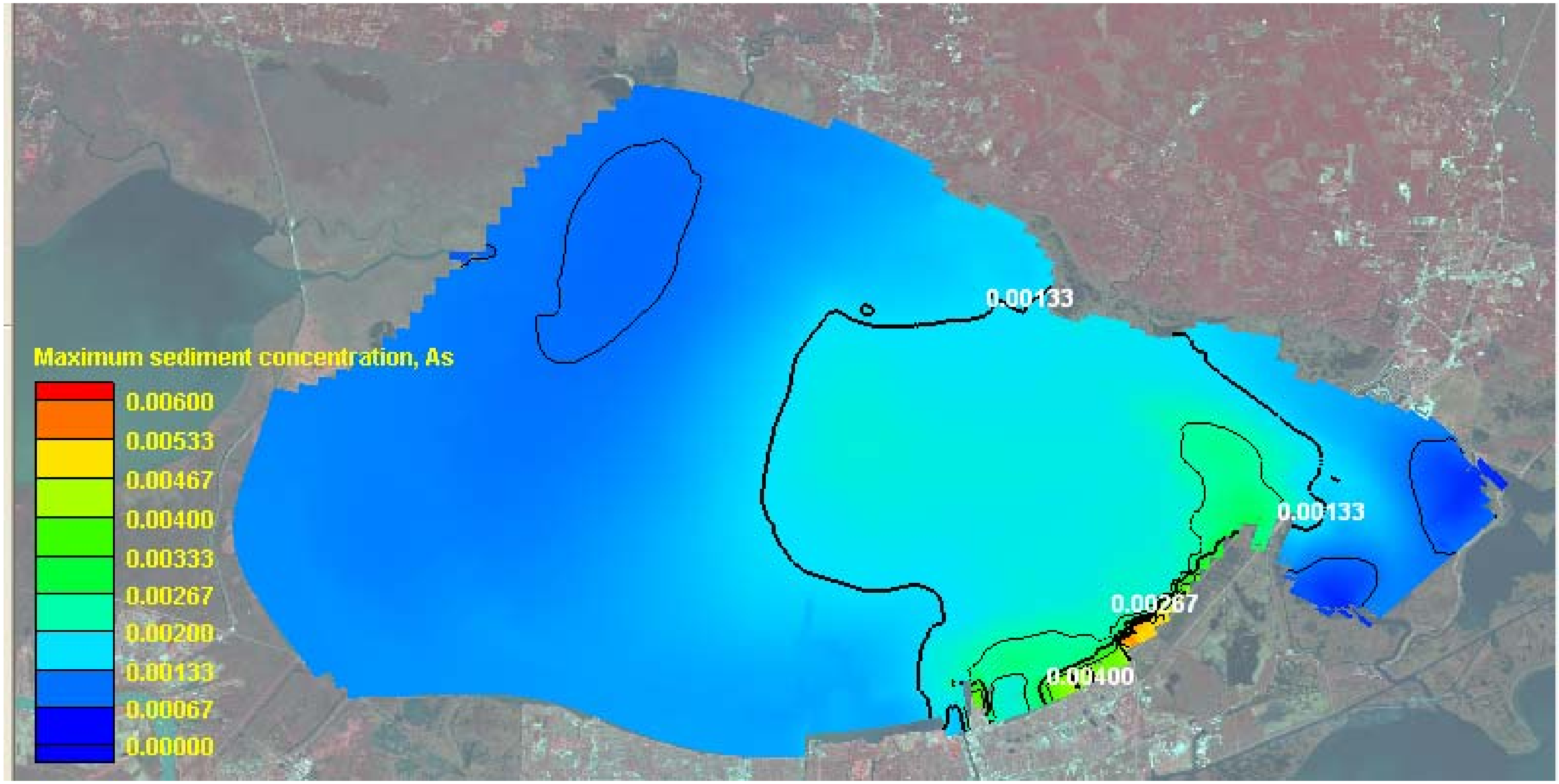

Figure A15. Incremental maximum As sediment concentrations (mg/kg total) in Lake Pontchartrain for *baseline* conditions, median loading concentrations.

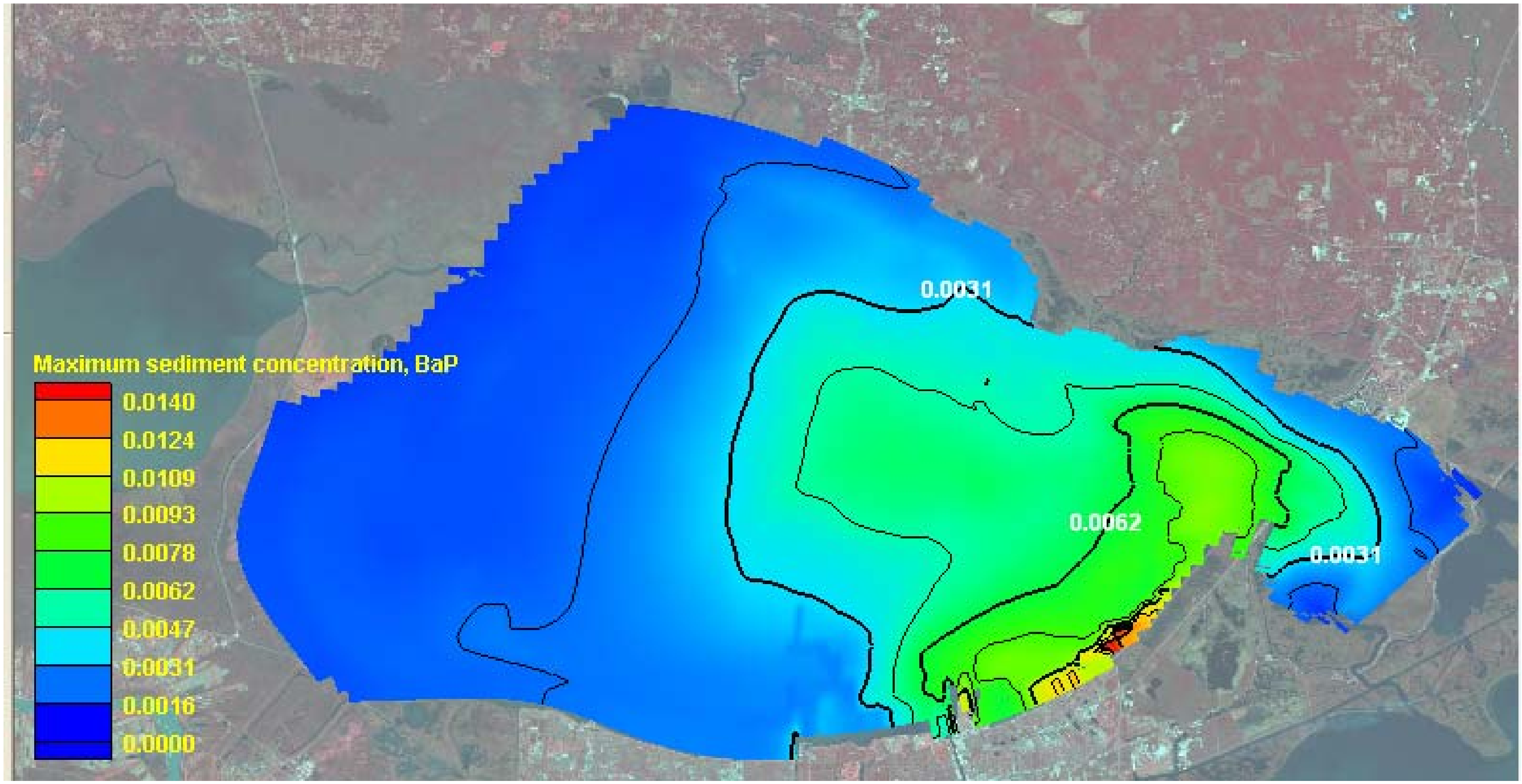

Figure A16. Incremental maximum BaP sediment concentrations (mg/kg total) in Lake Pontchartrain for *baseline* conditions, median loading concentrations.

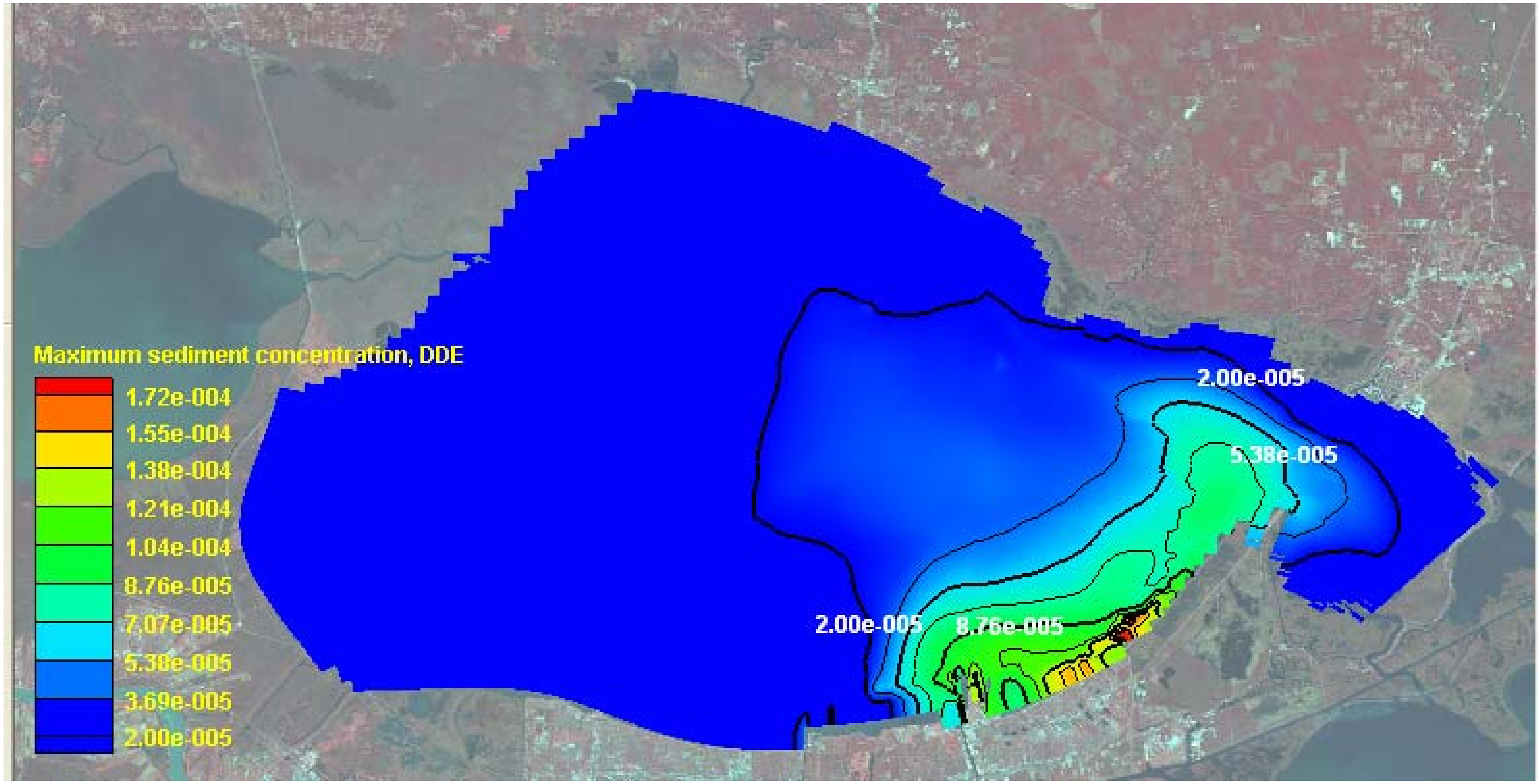

Figure A17. Incremental maximum DDE sediment concentrations (mg/kg total) in Lake Pontchartrain for *baseline* conditions, median loading concentrations.

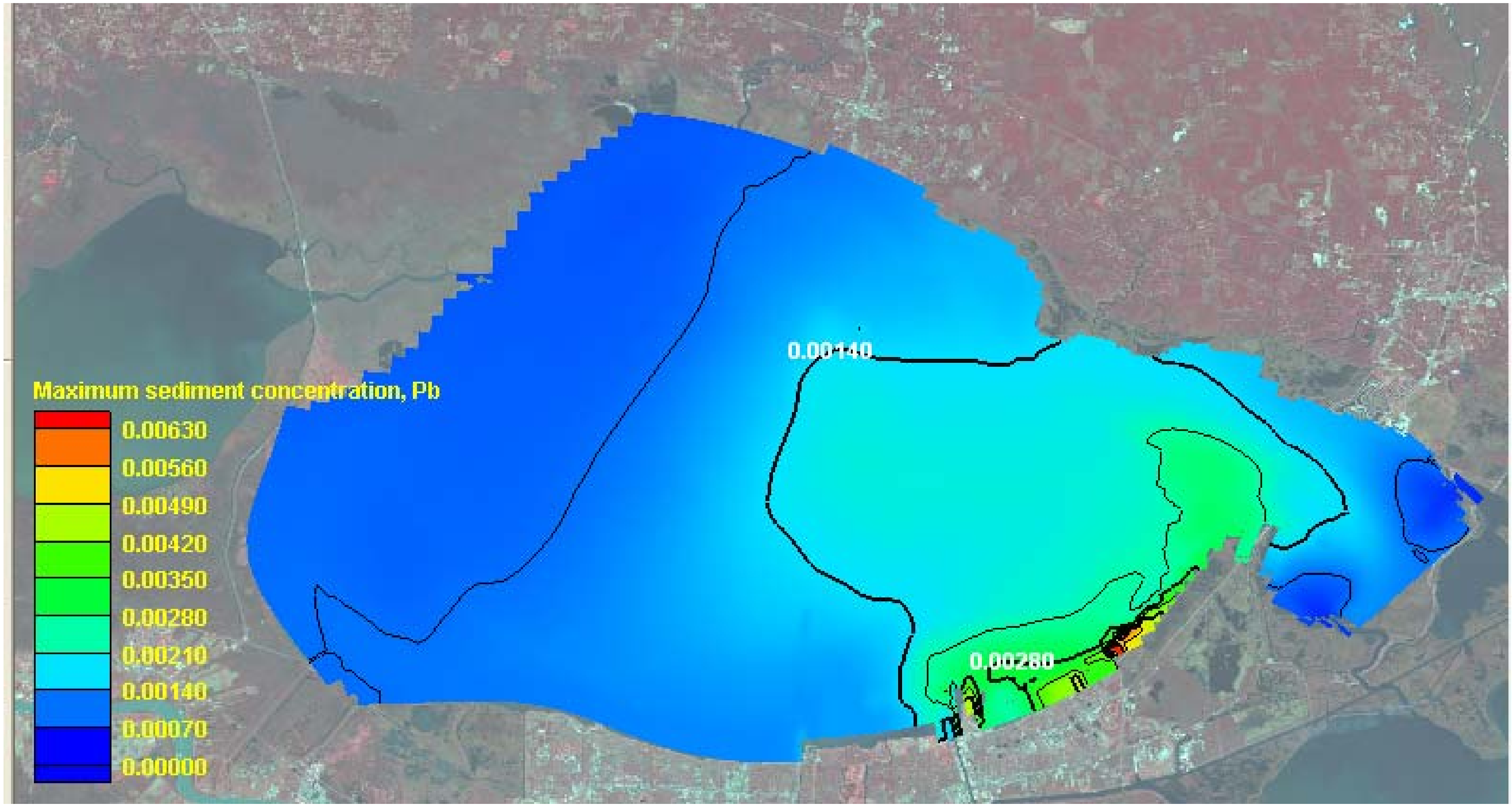

Figure A18. Incremental maximum Pb sediment concentrations (mg/kg total) in Lake Pontchartrain for *baseline* conditions, median loading concentrations.

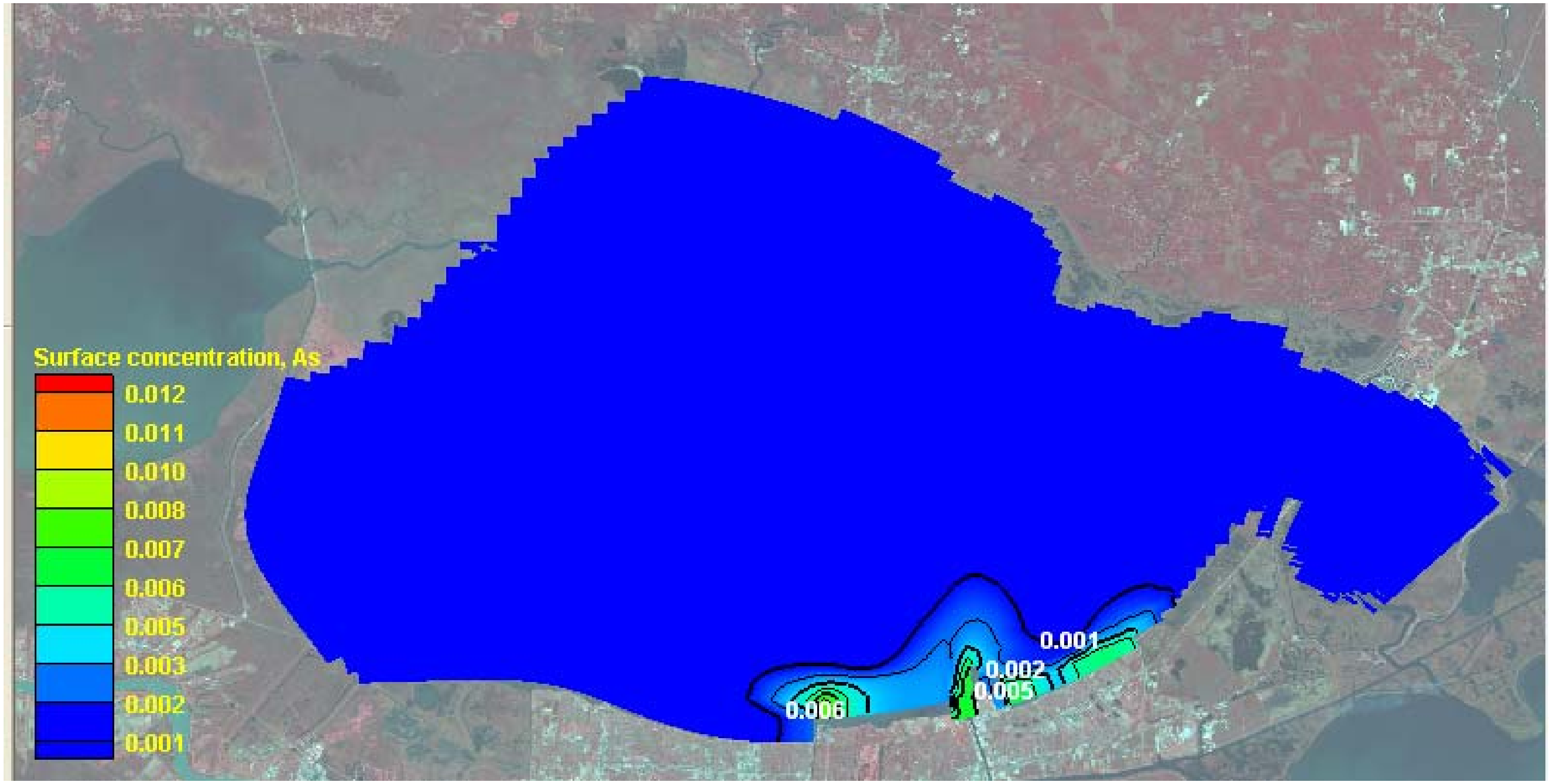

Figure A19. Water surface concentrations (mg/L total) for As in Lake Pontchartrain on September 12 for *actual* conditions, median loading concentrations.

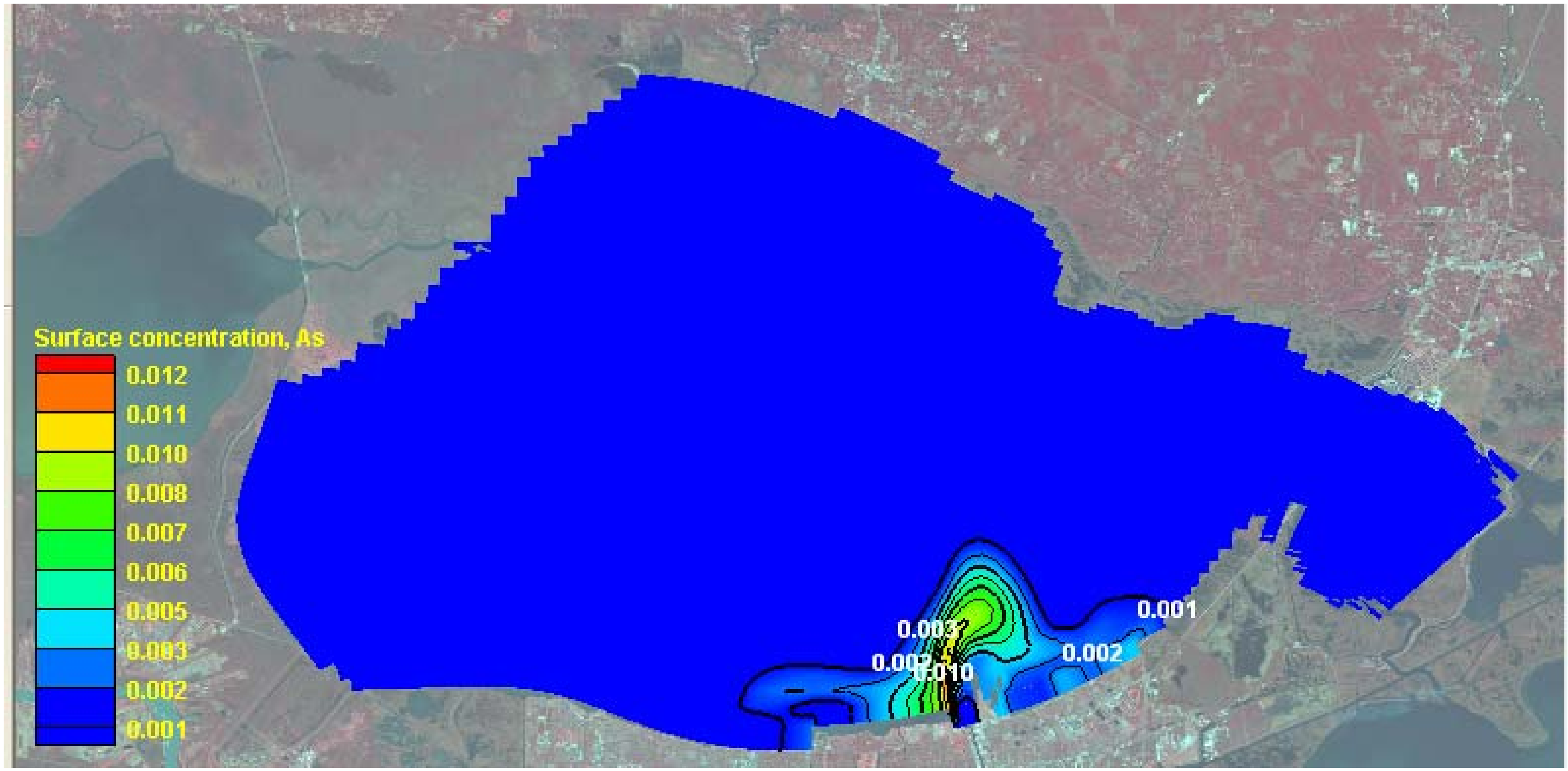

Figure A20. Water surface concentrations (mg/L total) for As in Lake Pontchartrain on September 12 for *baseline* conditions, median loading concentrations.

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14. ABSTRACT

This report documents mathematical model studies conducted to evaluate the environmental impacts of Hurricane Katrina. One of the primary environmental concerns associated with Hurricane Katrina was the impacts to ecological resources stemming from contaminants that were released into the floodwaters and subsequently pumped into surrounding water bodies outside the levee system. Contaminant concentrations within the water column and sediment bed were computed for two environmentally important water bodies, Lake Pontchartrain and Violet Marsh, both of which received pumped floodwater effluents following Hurricane Katrina. Two different mathematical models were used to simulate contaminant concentrations within the lake and marsh. Contaminant concentration information on these two systems was used to draw conclusions regarding the environmental consequences of contaminant releases. The models were used to determine the consequences of dewatering the floodwaters of Hurricane Katrina, with or without levee failure.

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