ONONDAGA LAKE CAPPING, DREDGING, HABITAT AND PROFUNDAL ZONE (SMU 8) FINAL DESIGN

APPENDIX M – MNR MODELING

Prepared for:

Honeywell

301 Plainfield Road, Suite 330
Syracuse, NY 13212

Prepared by:

ANCHOR QEA

290 Elwood Davis Road, Suite 230
Liverpool, New York 13088

Reviewed by:

PARSONS

301 Plainfield Road, Suite 350
Syracuse, NY 13212

and

Exponent

420 Lexington Avenue, Suite 1740
New York, NY 10170

FEBRUARY 2012

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

APPENDIX M

MNR MODELING FOR ONONDAGA LAKE

As discussed in Section 6.2 of the main text of this Draft Final Design Submittal for the profundal zone of Onondaga Lake, surface sediment mercury concentrations in SMU 8 have been declining naturally for many years and are approaching remediation goals set forth in the Record of Decision (ROD). The primary process resulting in natural recovery of SMU 8 sediment is burial of older sediment by newer, cleaner sediment that settles in the deep water zone of the lake over time. Consistent with U.S. Environmental Protection Agency (USEPA; 2005) sediment guidance and Department of Defense monitored natural recovery (MNR) evaluation recommendations (Magar et al. 2009), multiple lines of evidence including detailed evaluations of empirical data and computer modeling together define the role of natural processes in reducing risk over time. Evaluations of the considerable empirical MNR data available for SMU 8 are discussed in Section 6.2. Predicting future natural recovery rates typically requires site-specific numerical models, which quantify key fate and transport processes to estimate the time to recovery and to determine the likely future effectiveness of MNR. The site-specific MNR model employed for this draft final design evaluation is based on the peer-reviewed work of Boudreau (1997) as described in the following subsections.

M.1 DESCRIPTION OF MNR MODEL

A one-dimensional numerical model was used to quantify natural sediment recovery rates in SMU 8. The model is based on the extensive peer-reviewed models developed by Boudreau (1997) on diagenetic[1] processes in sediments. The one-dimensional Boudreau mass balance/process model was used to assess the long-term solid and dissolved contaminant fate and transport associated with natural sediment recovery by representing the effects of diffusion, bioturbation, groundwater mediated advection, settling, and burial in SMU 8. The model assesses fate and transport along the vertical axis of the sediment bed.

The governing equations for the model have been extensively peer-reviewed in the literature (Boudreau 1997). In addition, the model has been used and accepted for remedial design at other similar sediment Superfund sites, including the Middle Waterway in Tacoma, Washington, (Anchor Environmental and Foster Wheeler 2001) and Duwamish/Diagonal Combined Sewer Overflow in Seattle (Anchor Environmental 2002), among others.

This natural recovery model is based on Boudreau's Equations 3.80 and 3.83 (1997), which determine the integral conservation balances (i.e., conserves mass) of a species (e.g., a chemical of interest, which, in this case, is mercury) for dissolved and solid phases in a thoroughly mixed

[1] Diagenesis refers to the cumulative processes that bring about changes in a sediment or sedimentary rock subsequent to deposition in water.

layer of surface sediments in SMU 8. The governing equation for the natural recovery model, referred to here as the "standard model," is:

$$\frac{\partial M}{\partial t} = D_0\left[\frac{\varphi}{\theta^2}\frac{\partial C}{\partial x}\right]_L + \left[\varphi D_B\frac{\partial C}{\partial x}\right]_L + [\varphi uC]_L + [\varphi_s wB]_0 - D_0\left[\frac{\varphi}{\theta^2}\frac{\partial C}{\partial x}\right]_0 - \left[\varphi D_B\frac{\partial C}{\partial x}\right]_0 - [\varphi uC]_0 - [\varphi_s wB]_L - \sum\int_0^L R dx$$

where:

M = mass of chemical of interest (milligrams [mg])

T = time (years)

D_0 = molecular diffusion coefficient (square centimeters per year [cm^2/yr])

φ = porosity of sediments (unitless)

θ = tortuosity of sediments (unitless)

C = concentration of chemical in dissolved phase (milligrams per liter [mg/L])

x = spatial variable (along the depth of sediments) (cm)

L = where x = L; the bottom of the mixed layer (cm)

D_B = biodiffusion or mixing coefficient for sediments (cm^2/yr)

μ = velocity of porewater (centimeters per year [cm/yr])

φ_s = solid fraction volume (unitless)

w = burial velocity of solids (or settling rate) (cm/yr)

B = concentration of chemical in solid phase (milligram per kilogram [mg/kg])

0 = where x = 0; top of the mixed layer (sediment water interface) (cm)

R = reaction of chemical through depth interval (i.e. biodegradation loss) (mg)

The governing equation provides the change in chemical mass over the specified time interval. By assuming a unit volume of mixed layer sediment, this equation can be used directly to calculate concentrations of the chemical of interest in the mixed layer over the same time. The net change in mixed layer mass is determined by the sum of changes produced by diffusion, biodiffusion (diffusion driven by bioturbation of sediments), groundwater advection, sediment settling, burial, and biodegradation (for organic chemicals). The model does not incorporate conversion of mercury to methylmercury. To the extent that methylmercury can flux to the water column more readily than total mercury, this is a conservative assumption in terms of estimating sediment total mercury concentrations (i.e., the model will likely overestimate the total mercury concentrations in sediments over time).

Following numerous examples in Boudreau (1997), the partial differential equation noted above was converted to a series of ordinary differential equations. The resulting ordinary differential equations are solved numerically in the model using Euler's method. The model was executed in Microsoft Excel. The visual basic for applications (VBA) code in Microsoft Excel used to execute the model incorporated additional quality control measures. For each time-step, each model variable used in the model equations (both input and calculated) was compared to the model variables output from STELLA, an independent model platform used to execute the model. The model variables and final model results outputs from the two platforms match well.

The variable R represents the total mass change due to all chemical production/destruction reactions that occur in the mixed layer. The only such reaction typically considered is anaerobic biodegradation for organic chemicals. Biodegradation was not specifically employed for this modeling effort, because total mercury was modeled.

The model defines two sediment layers: a buried layer and a surface mixed layer. The model assumes that mixing of sediments within the surface layer is essentially instantaneous within each time step. Generally, mixing of surface sediments due to physical and biological activity (bioturbation) takes place during a sufficiently short time scale that this assumption is reasonable for the purpose of predicting natural recovery over a period of years (Boudreau 1997; Berner 1980). Currently, the assumed depth of the mixed layer (4 cm) cannot be varied within the model as coded. If necessary in the future, the model interface and code can be modified to have the flexibility to simulate such changes to the mixed layer depth. The applicability of the mixed layer assumptions to this system is discussed more below.

The governing equation includes processes for both dissolved-phase and solid-phase chemicals. Consequently, equilibrium-partitioning assumptions are used to quantify the mass of chemical present in each phase at any given time in the model.

The model was used to simulate mercury concentration over the period ending in 2027. Based on the schedule for remediating the littoral zone outlined in the Consent Decree, remediation is anticipated to be completed in 2017. ROD compliance requires the mercury probable effect concentration (PEC) and bioaccumulation-based sediment quality value (BSQV) remedial goals be met in SMU 8 by 10 years following remediation, which is anticipated to be the year 2027.

M.2 MODEL INPUTS

Model inputs were derived from extensive site sampling efforts, bench scale testing, and literature. Key parameters in the model are mixed layer depth, sedimentation rates, and mercury concentration in settling sediment. These parameters and others are summarized in Table M.1 and are discussed more in the following paragraphs.

M.2.1 Mixing Depth

The model input for sediment mixing depth is denoted as the mixed layer depth. Mixing of relatively clean sediments that settle from the water column with underlying sediments is one of the key processes involved in predicting natural recovery in SMU 8. Mixing of sediments can result from physical processes, such as currents driven by wind, and from movement of bottom-dwelling (benthic) organisms in the sediment, denoted as bioturbation. As discussed below for each process, movement of profundal zone waters due to wind is insufficient to cause noteworthy physical surface sediment mixing and little, if any, bioturbation due to the anoxic conditions of the profundal zone that persist typically for three months each summer. Based on these conditions and visual evidence of currently and historically undisturbed surface sediment described below, 4 cm was determined to be a conservative estimate of the mixed layer depth.

A factor that could potentially change the 4 cm mixed layer depth is future increases in dissolved oxygen (DO) within the hypolimnion, as DO increase could result in greater colonization of SMU 8 sediment by benthic organisms and, consequently, increased bioturbation and associated mixing. Analysis of the possibility of this scenario indicates it to be highly unlikely, even in the future. The analysis was based on historic measurements of oxygen depletion and comparisons with a suitable reference system (i.e., Otisco Lake). The rate of hypolimnetic oxygen depletion reflects the decomposition of settling and deposited particulate organic matter that is formed mostly through primary production in the overlying photic zone, and is a widely recognized indicator of trophic state in dimictic lakes (Wetzel 2001). In lakes with large legacy deposits of degradable organic matter in the sediments, the rate of oxygen depletion may reflect historic, as well as contemporary levels of primary production (Matthews and Effler 2006). The rate of loss of dissolved oxygen from the hypolimnion can be represented on an areal basis as the areal hypolimnetic oxygen deficit (AHOD; grams per square meter per day [$g/m^2/d$]), or on a volumetric basis as the volumetric hypolimnetic oxygen deficit (VHOD; grams per cubic meter per day [$g/m^3/d$]). The VHOD representation is generally preferred for comparisons amongst lakes (Burns 1995; Denkenberger et al. 2007). Water temperature and lake morphometry, particularly the dimensions of the hypolimnion, influence the rate of oxygen depletion. Lakes with warm, shallow hypolimnia generally have higher rates of volumetric oxygen depletion.

Dramatic decreases in the rate of hypolimnetic oxygen depletion were observed in Onondaga Lake from the 1980s through the early 2000s (Figure M.1). No systematic trend is evident in the later years of the record, as VHOD values have remained in the range 0.15 to 0.23 $g/m^3/d$ since 2000. The timing of the onset of complete hypolimnetic anoxia in Onondaga Lake was computed for specified values of VHOD (Figure M.2). This analysis indicates that VHOD would have to decrease below 0.10 $g/m^3/d$ in order for the hypolimnion to remain oxic through the summer. This would represent a decrease of approximately 50 percent from contemporary VHOD values and a rate of hypolimnetic oxygen depletion lower than observed for nearby, mesotrophic Otisco Lake (see Denkenberger et al. [2007] for a more thorough comparison of hypolimnetic oxygen depletion in these two lakes). The highly non-linear relationship between the timing of hypolimnetic anoxia and the rate of volumetric oxygen depletion is particularly noteworthy. Further decreases in VHOD would result in progressively larger delays in the onset of anoxia. Such decreases would require further major reductions in nutrient loading, beyond those accomplished to date at the Metropolitan Syracuse Wastewater Treatment Plant (Metro), and/or time (e.g., decades) for the sediments to come into a new steady state with contemporary levels of particulate organic matter deposition (Matthews and Effler 2006). At this time, a scenario whereby the hypolimnion of Onondaga Lake will remain oxic throughout the summer does not appear to be realistic. It should be noted that oxygenation of the lake is still being considered as a means to reduce methylmercury flux from profundal sediment. However, as discussed in Section 6.1 of this Draft Final Design, nitrate addition has been very successful at reducing methylmercury formation in SMU 8 sediment and supplemental nitrate addition to the hypolimnion is currently being evaluated as the preferred method for minimizing methylmercury flux with a 3-year nitrate addition pilot test commencing during the year 2011 (Parsons and Upstate Freshwater Institute [UFI] 2011). If the pilot test proves nitrate addition to be successful, full-scale implementation of nitrate addition will likely be implemented as needed in place of oxygenation.

In the unlikely event that the profundal zone remains oxic in the future during the summer months (through natural or engineered means) or factors change such that this condition is predicted to occur, the appropriateness of the sediment mixed layer depth of 4 cm would be reassessed as part of the ongoing MNR monitoring and contingency approach (see Section 6.2.3).

M.2.1.1 Evidence of Layering/Laminations

Visual evidence based on freezing and slicing shallow sediment cores collected during 2010 in SMU 8 shows that mixing of sediment is not taking place at depths below the top 1.5 cm (Figure M.3a and M.3b). In all but one core[2], laminations were first observed at a depth of 1.5 cm or less (some began at the surface). The presence of layers or laminations in the SMU 8 sediment is primary evidence that SMU 8 sediment is relatively undisturbed and not affected by bioturbation or resuspension of lakebed sediment. Layering of SMU 8 sediment was observed by Rowell (1992) and has been attributed to deposition of calcite, clays, and diatoms (silica) associated with erosion of the watershed, productivity cycles within the lake, and other annual events (Effler and Harnett 1996).

To update and confirm prior observations, Parsons collected and processed three shallow sediment cores from the North Basin and three shallow sediment cores from the South Basin in 2010. The cores were collected in an undisturbed manner, kept vertical, frozen with dry ice once onshore, and then sliced vertically while frozen to examine layering. Each of these 2010 cores showed thin layering (laminations) from the sediment surface downward. This evidence of layering (Figure M.3a and M.3b) demonstrates that the mixed layer depth at multiple locations in SMU 8 shallow sediment is less than 2 cm. Use of a 4-cm mixed layer depth in modeling is conservative in the sense that a thicker mixed layer depth slows down the calculated rate of natural recovery, as shown in the sensitivity analysis in Section M.3.4. Consequently, the assumption of a 4-cm mixed layer will show a slower rate of natural recovery as compared to the recovery of the thinner mixed layer that actually appears to exist in the profundal sediments. Additional frozen cores collected from the shallow portions of SMU 8 during the 2011 field season to assess mixing depths in areas of SMU 8 with water depths of 30 to 50 feet (ft) support use of a 4-cm mixed layer.

[2] In core OL-MB-100, collected during 2010, the first clearly defined varve is visible faintly at a sediment depth of 5 centimeter [cm]. This core, however, was not frozen completely and as a result there may have been distortion of the upper portion of the core during storage and/or handling.

M.2.1.2 Lack of Benthic Organisms

The profundal zone of Onondaga Lake typically lacks oxygen from mid-June until fall turnover in mid-October each year (Parsons, Exponent, and Anchor QEA 2010). While some benthic organisms can persist for relatively short periods in anoxic sediment, they require oxygen in overlying water to propagate. The annual anoxia in Onondaga Lake precludes long-term activity and colonization of benthic organisms in SMU 8 sediment. This position is supported in multiple studies of Onondaga Lake.

Benthic macroinvertebrates were collected in 2008 as part of baseline monitoring program (Parsons, Exponent, and Anchor QEA 2009). Of the 20 locations sampled during 2008, two SMU 8 locations adjacent to the littoral zone SMUs were sampled to assess community composition. Five replicates (i.e., petite ponar dredge samples) making up a sample were collected at each location in August, with a goal of collecting 100 macroinvertebrates from each replicate for a total of 500 individuals in each sample (Parsons et al. 2008). Very few macroinvertebrates were found in 2008 in the SMU 8 sediments; 32 and 34 macroinvertebrates per sample were collected from the two SMU 8 locations at 13.4 m and 14.3 m water depth, respectively, where overlying water is anoxic compared to more than 500 macroinvertebrates per sample collected at littoral-area SMU locations where overlying water contains oxygen.

Benthic macroinvertebrates were also sampled in 1998 along an east to west transect of Onondaga Lake under the direction of Dr. Nelson Hairston of Cornell University and Dr. Steven Effler of UFI. Samples were collected with an Ekman dredge at water depths of 2, 5, 10, 15, and 19 meters. Results indicate relatively few benthic macroinvertebrates in sediment at water depths of 10 meters and greater (Figure M.4).

M.2.1.3 Lack of Physical Mixing

Physical mixing of sediment can occur if water currents are strong enough to resuspend sediment particles; however, such mixing is not evident in sediment cores from Onondaga Lake (i.e., presence of laminations) and water currents in the hypolimnion are not strong enough to cause noteworthy resuspension of SMU 8 sediment. The depth of the profundal zone provides protection from wind that controls movement of shallower water in Onondaga Lake (Owens and Effler 1996), even under extreme events.

Cowen and Rusello (2008) of Cornell University measured water current velocities near the SMU 8 sediment surface during October 2008 and performed a preliminary assessment of turbulence in the bottom boundary layer of Onondaga Lake. Their findings concur with the conclusions of Owens and Effler (1996) that velocities near the sediment bed are weak. Wind data collected at the South Deep location by UFI show that the most frequent (10 percent of the time) wind direction is out of the west and can reach up to 10 meters per second (22 miles per hour). The highest wind speeds of greater than 10 m per second are measured from the south winds, which occur about 6 percent of the time (Cowen and Rusello 2008). Cowen and Rusello observed mostly weak turbulence levels and currents. Burst mean currents measured ranged from 0.2 to 9.6 cm per second, with a mean of 3.0 cm per second (0.07 miles per hour) at the Saddle and were double the measurements at South Deep (between 0.1 and 4.4 cm per second, with a mean of 1.4 cm per second). The bed shear stress due to skin friction derived from the maximum velocities measured at the Saddle and South Deep locations equal 0.0276 and 0.0058 Pascals (Pa) (1 Pa = 1.4508 x 10^{-4} pounds per square inch [psi]), respectively. Scour is unlikely to occur given these small shear stress values (Ziegler 2002).

Parsons confirmed similar velocities during two nitrate field trial applications in July 2009 by deploying an acoustic doppler velocimeter at South Deep and at North Deep at 1 m above the lake bottom. Water velocities measured at the two locations through 2009 peaked at approximately 4 cm per second (Parsons and UFI 2010). Given the typical particle size typical of SMU 8 sediments and the observed near-bed velocities, the Hjulstrom Diagram shows that water velocities observed at the two locations are in the range of suspended sediment transport (i.e., sediment already in suspension) (Figure M.5) but are not high enough to move the fine-grained sediment present in SMU 8 (i.e., bedded sediment erosion).

Fluorescent microbead markers have been placed at representative locations in SMU 8 in part to evaluate mixing of SMU 8 sediments over time. The fluorescent microbead markers were applied during mid-2009 on behalf of Honeywell to nine different 1,400-square-foot plots of Onondaga Lake profundal zone sediments. Two types of markers were applied in 2009: a sand tracer, which marks the mudline (i.e., top of sediment) as of mid-2009 when the microbead particles were applied, and a silt tracer, which mimics the sediment type present in the profundal zone. This silt tracer will, in the future, be another tool to evaluate potential mixing of SMU 8 sediment over time. Sampling of the sediments in the area of the microbead plots took place during late 2009 and 2010 and 2011 and every three years thereafter in accordance with an approved work plan (Parsons, Anchor Environmental, and Environmental Tracing Systems, 2008). The ability to slice SMU 8 surface sediment into 1- or 2-cm-thick intervals means that measureable newly settled sediment above the sand microbead marker should be evident by approximately 2011 or 2012, two or three years following microbead marker placement. Additional microbead marker sampling techniques were tested in 2010 (Parsons and Environmental Tracing Systems, 2010) and 2011 to assess the effect of sub-sampling on the position of the tracer material within a core.

M.2.2 SMU 8 Compliance Depth

Compliance depth is the depth of sediment that will be considered in assessing compliance with sediment criteria. This sediment depth will be monitored over the course of the 10-year MNR period following dredging and capping. The sediment goals for SMU 8 are the mercury PEC of 2.2 mg/kg on a point-by-point basis and the BSQV for mercury of 0.8 mg/kg on an area-wide basis.

M.2.2.1 SMU 8 Compliance Depth for Mercury PEC

The PEC remediation goal for mercury was developed in consideration of potential toxicity to benthic macroinvertebrates that are exposed directly to mercury in sediment. In order to have exposure, benthic macroinvertebrates must be present. The discussion of mixing depth above clearly indicates that SMU 8 sediments do not mix vertically and benthic macroinvertebrates are not present in significant numbers in SMU 8 sediment and are not expected to be present in significant numbers in the future. Therefore, the use of a 4-cm mixed layer depth to assess compliance with the mercury PEC has been identified as a conservative compliance depth for SMU 8 in the absence of oxygen year round in hypolimnion waters.

M.2.2.2 SMU 8 Compliance Depth for BSQV

The BSQV remediation goal for mercury was developed in consideration of potential bioaccumulation of methylmercury from sediment to fish. Unlike the PEC, the exposure pathway is indirect and multiple factors influence the relationship between mercury in sediment and methylmercury in fish. A key factor is methylmercury release from SMU 8 sediment to overlying water where it can eventually be bioaccumulated. This release occurs when oxygen and nitrate are depleted from overlying water. Another potential route of methylmercury bioaccumulation is from sediment to benthic macroinvertebrates to fish; however, such bioaccumulation is not relevant to SMU 8 sediment because benthic macroinvertebrates are not expected to be present in significant numbers.

Recent Onondaga Lake sediment incubation work by Michigan Technological University, UFI, and Syracuse University conducted on behalf of Honeywell evaluated the flux of methylmercury from SMU 8 sediment (Exponent et al. 2010). The researchers measured concentrations of total mercury, methylmercury, and key redox parameters in sediment cores and water overlying the sediment cores under three conditions: 1) oxic (DO and nitrate in overlying water); 2) anoxic (nitrate but no DO in overlying water); and 3) anaerobic (no DO and no nitrate in overlying water). Microelectrode probes and fine resolution slicing and analysis of cores showed that the main sulfate-reduction zone and maximum methylmercury concentration occurs at approximately the 2- to 3-cm depth in the sediment when DO is present in overlying water. However, the methylmercury produced within the 2- to 3-cm depth interval does not diffuse to overlying water due to the intervening sediment layers containing DO and/or nitrate that can sorb or demethylate methylmercury. Under anoxic and anaerobic conditions that mimicked the progress of stratification as DO and then nitrate are depleted from overlying water, the sulfate reduction (and mercury methylation) zone moved upward toward the sediment/water interface. Under anoxic conditions, where nitrate, but not DO, was present in overlying water, the maximum methylmercury concentration was found at the 1- to 2-cm depth (Figure M.6). When methylmercury release occurs under anaerobic conditions, the methylmercury production zone is likely within very near surface sediment (i.e., within 0- to 1-cm depth interval) or at the sediment surface. Total mercury concentrations deeper in the sediment are irrelevant to the release of methylmercury. Mercury partitioning to sediments is strong and, therefore, the movement of mercury from deeper sediments towards the surface sediment/water column interface, where the methylmercury production occurs, is limited.

According to the laboratory incubations described above, the release of methylmercury—the form of mercury that bioaccumulates in biota—is from very near surface sediment (i.e., within 0- to 1-cm depth interval) or at the sediment surface. Taking into account any uncertainty associated with applying these laboratory results to Onondaga Lake, a 4-cm compliance depth for BSQV modeling and mercury monitoring is a conservative basis for assessing mercury that could be contributing to methylmercury flux to the water column.

M.2.3 Sedimentation Rates

Sedimentation is the physical process by which particulate matter settles out of the water column and deposits on top of the existing sediment bed, such that the current surface sediments (and contaminants contained within those sediments) are buried over time beneath the new sediment surface.

M.2.3.1 Existing Sedimentation Rates

Sedimentation rates were estimated from historical Remedial Investigation (RI) and more recent data collected using two basic techniques: high resolution cores (including radioisotope analyses and use of mercury markers) and sediment traps.[3] "Recent" sedimentation rates from the 2008 high resolution core data average 0.26 grams per square centimeter per year (g/cm^2/yr), with a range of 0.13 to 0.35 g/cm^2/yr (Parsons 2010, Appendix F) across the various cores measured. These "recent" rates were derived from the high resolution core sections representing the most recently deposited sediment. Recent sedimentation rates are derived from the top two sections of these seven cores (0- to 2-and 2- to 4-cm intervals). Rates derived from the deeper core sections were not used for the purposes of quantifying recent sedimentation rates. Sedimentation rates from the most recent sediment trap data collected during 2009 and 2010 average 0.28 g/cm^2/yr and 0.34 g/cm^2/yr, respectively, with a range of 0.09 to 0.78 g/cm^2/yr across the seasons sampled (i.e., not including winter). It should be noted that the ranges provided throughout this section for cores are based on spatial variations in data collected, while the ranges provided from sediment traps represent temporal (monthly) variations at one location that were converted to g/cm^2/yr for consistent comparison to core sedimentation rates. A sedimentation rate of 0.25 g/cm^2/yr, consistent with the findings from the analyses described below, was used as a conservative input to the MNR model.

High Resolution Cores

Radioisotope cores were collected in the north and south portions of the profundal zone of Onondaga Lake during 1988 by Rowell (1992). As reported in the Remedial Investigation Report (TAMS Consultants 2002), six cores were sampled; two collected by Rowell (1992) and four collected by (PTI) for Honeywell. Five of the six cores sampled show a clear trend of cesium-137 (^{137}Cs) radioisotope deposition consistent with historical sources of this isotope, and subsequent preservation of the sediment column that maintained that historical record. Figures of

[3] As discussed more fully in Section M.2.1, the profundal zone of Onondaga Lake is quiescent and, given the low near-bed velocities, the likelihood of resuspension is low; therefore, for the purposes of this discussion, sedimentation rate refers to "net" sedimentation rate, rather than the "gross" sedimentation. Because of the relatively low resuspension rates in the profundal zone of Onondaga Lake, for most purposes, net and gross sedimentation rates can be assumed to be nearly the same.

these cores are shown in Figure N.6 of Appendix N of the Feasibility Study (FS; Parsons 2004). The demarcations of interest, such as the appearance of ^{137}Cs associated with nuclear testing that started in 1954 and the peak of that testing in 1963, have good resolution. This indicates little disturbance to the sediments since that time (i.e., the sediment column was stable and did not exhibit signs of significant erosion events or large-scale re-working).[4]

Effler (1996) and Hairston et al. (1999) present radioisotope results from three additional cores. The two cores presented by Effler (1996) were collected during 1988 by Rowell and were subjected to both ^{137}Cs and lead-210 (^{210}Pb) radioisotope analysis. The Hairston et al. (1999) core was collected in 1997 and was analyzed for ^{210}Pb. (Sharpe [2004] subsequently obtained archived samples of this core and analyzed them for mercury as well to evaluate mercury markers). All three of these cores show clear evidence of long-term undisturbed deposition (i.e., stability), consistent with the findings from Rowell's cores. Sedimentation rates from these cores are listed in Table M.2. In addition to the high resolution core data described above, Honeywell collected more recent high resolution cores during 2008 to evaluate sedimentation rates using both ^{137}Cs and ^{210}Pb analyses (Table M.2 and Parsons 2010).

Sedimentation results are typically reported in either g/cm^2/yr or in cm/yr. In order to review and compare sedimentation rates, data reported in cm/yr were converted to g/cm^2/yr based on a typical bulk density of 0.243 grams per cubic centimeter (g/cm^3) derived from a porosity of 0.91 and a specific gravity of 2.7 g/cm^3.

Historical core sedimentation rates presented in the FS (pre-2008) are shown in Table M.2. Mid-range sedimentation rates on a g/cm^2/yr basis range between 0.07 g/cm^2/yr and 0.30 g/cm^2/yr. The maximum of the range (0.30 g/cm^2/yr) is from evaluation of a core horizon dating to approximately 1984 from a core collected in 1997 core reported by Hairston et al. (1999). The low end value of 0.07 g/cm^2/yr is from TAMS Consultants (2002; a discussion of Rowell's cores from the early 1990s). Sedimentation rates from recent high resolution cores collected during 2008 range from 0.13 to 0.35 g/cm^2/yr (average of 0.26 g/cm^2/yr).

4 One core collected in the Ninemile Creek Outlet Area does not follow this pattern, although it shows a clear increase of ^{137}Cs activity with depth. This profile is probably related to deposition of ^{137}Cs from the creek itself, which could have occurred in more sporadic events associated with periodic watershed runoff and erosion that blurred the concentration profile. Dredging conducted during the 1960s may have also affected this profile.

Sediment Traps

Sediment trap data also provide a reasonable measure of net sedimentation rates[5]. Sediment trap data collected from the Onondaga Lake profundal zone between 1986 and 2009 were compiled for this assessment of sedimentation rates (UFI 2010). Table M.1 lists the sediment trap data. Sedimentation rates post-1986 are lower than 1986 rates (soda ash was being produced in Syracuse in earlier years and likely contributed to higher sedimentation rates). Thus, the 1986 rates are not used further in this analysis except for comparative purposes, as they are not representative of current conditions.

Historical trap sedimentation rates presented in the RI and FS were obtained from sediment traps collected mostly during summer months, and vary in a range of approximately 0.11 to 1.4 $g/cm^2/yr$ (average of mid-range values is 0.47 $g/cm^2/yr$) after 1986. The high value in this range represents the seasonal maximum result from sediment trap samples from Effler (1996) collected during 1988. The low-end value is based on the seasonal minimum value obtained from sediment traps deployed during 1996 for one month in the summer. Thus, it is unlikely that this low value is representative of overall annual deposition rates within the lake. Recent sediment trap samples collected by UFI during 2009 from April until fall turnover in October were collected in triplicate and subsequently averaged. Sedimentation rates from the averaged triplicates ranged from 0.1 to 0.78 $g/cm^2/yr$ (seasonal average of 0.28 $g/cm^2/yr$). Sediment trap samples collected by UFI during 2010 from April until fall turnover in October were collected in triplicate and subsequently averaged. Sedimentation rates from the averaged triplicates ranged from 0.09 to 0.75 $g/cm^2/yr$ (seasonal average of 0.34 $g/cm^2/yr$). Seasonal average deposition during 2011 was 0.29 $g/cm^2/yr$. It should be noted that while these sedimentation rates from the sediment traps are represented on an annual basis (per year), they do not consider sedimentation rates during the winter months, which may be lower; the rates assume summertime rates occur year-round. Also, as noted above, these temporal ranges should not be confused with the spatial annual rate ranges discussed for the cores.

Figure M.7 presents a summary of the sedimentation rates from Table M.2. Average sedimentation rates (mid-range values of pre-2008 data, and average of triplicate sediment trap data collected during 2009) were summarized statistically. As FigureM.7 shows, the recent 2009 and 2010 sediment trap minimum, mean, and median seasonal sedimentation rates are less than minimum, mean, and median rates from the 1988 to 2000 seasonal sediment trap data. For core sedimentation rates, this pattern is reversed with the recent 2008 core minimum, mean, median,

5 Typically, sediment traps capture all sediment regardless of whether it might normally resuspend at some later time, and therefore provide a measure of the “gross” sedimentation rate. Sediment traps also intercept sediments higher in the water column before solids have settled to the sediment bed. Consequently, with sediment traps there is also an assumption that the particles intercepted by the traps will eventually settle to the sediment bed. Also, sediment traps are only able to be deployed and measured during months when ice is not present on the lake surface. Finally, individual trap measurements may cover periods of less than a month and should not be assumed to represent annual overall deposition rates. However, despite these limitations, sediment traps provide a reasonable indication of “net” deposition rates, because very little of the sediments are expected to be resuspended and most of the sediments intercepted by the traps would be expected to eventually deposit on the sediment surface under normal quiescent conditions in the profundal zone.

and maximum rates being higher than the core-based sedimentation rates collected prior to 2008. The recent median and mean 2009 and 2010 seasonal trap rates and 2008 core rates are all 0.25 g/cm^2/yr or greater. Weekly sediment trap data have been collected by UFI between the months of April and October since 1980. Focusing on data post-closure of the soda ash facility (1987 through 2010), the data show similar sedimentation rates as discussed above (Figure M.8). While the mean seasonal downward flux of suspended solids varies year to year, the average annual downward flux of suspended solids ranged from 0.22 to 0.51 g/cm^2/yr, with a mean of 0.38 g/cm^2/yr. Note that the bars shown in Figure M.8 represent ranges of the temporal measurements, not an estimate of error of confidence in each individual measurement. As discussed in Section M.3, the model has been calibrated to date using a sedimentation rate of 0.25 g/cm^2/yr, which is also similar to, but lower than the average sedimentation rate of 0.28 g/cm^2/yr for 2009 sediment trap data, 0.34 g/cm^2/yr for 2010 sediment trap data, and the average rate of 0.26 g/cm^2/yr for the 2008 high resolution cores noted above. Although, for the reasons stated above, rates in cores and sediment and the variability of the measurements are not exactly analogous, taken together this information suggests that an overall rate of 0.25 g/cm^2/yr is a reasonable estimate.

The microbead markers applied during mid-2009 on behalf of Honeywell to nine different 1,400-square-foot plots of Onondaga Lake profundal zone sediments may also be used to establish sedimentation rates. The sand tracer marks the top of sediment as of mid-2009 when the microbead particles were applied. As discussed in Section M.2.1, the depth of the newly settled sediment above the sand marker should be measurable by approximately 2011, based on the ability to slice the cores in 1-cm or 2-cm vertical intervals. The results of this ongoing study may be used to reassess the appropriateness of the sedimentation rate of 0.25 g/cm^2/yr as part of the ongoing MNR monitoring and contingency plan.

M.2.3.2 Anticipated Future Sedimentation Rate

Sedimentation rates are influenced by internally generated sources of solids and external upland/watershed sources of solids that enter the lake. As described in Section M.2.3.1, the current average sedimentation rate based on the 2008 high resolution core data is 0.26 g/cm^2/yr and the 2009 and 2010 sediment trap data is 0.28 g/cm^2/yr and 0.34 g/cm^2/yr, respectively. Based on considerations presented in the following paragraphs, it is possible that future sedimentation rates could be lower than the current 0.26 g/cm^2/yr, 0.28 g/cm^2/yr, and 0.34 g/cm^2/yr averages noted above; however, these reductions are difficult to predict, in part because the current contributions to overall net deposition in the profundal zone from external and internal sediment loads are difficult to quantify. Thus, sedimentation rates used in the modeling described in Section M.3 are kept constant throughout the projection period. The appropriateness of this assumption will be reassessed as new data are available as part of the ongoing MNR monitoring and contingency plan. The first scheduled reassessment of model parameters will occur prior to the start of the MNR period and should provide an early warning to any sedimentation rate changes should they occur. Long-term monitoring and contingency actions, including additional assessment of modeling approaches and appropriate input parameter values, are discussed in Section 6.2 of this Draft Final Design.

Appendix N of the Feasibility Study (FS) discusses the potential for external sources of suspended solids from tributaries to decrease over time. Some researchers have hypothesized that reductions in future sedimentation rates are possible due to mechanisms such as phosphorus reductions due to wastewater system upgrades, changes in internal production of calcium carbonate, and influence of *Daphnia* sp, grazing (Hurteau et al. 2010). However, data collected as part of the Onondaga Lake Ambient Monitoring Program suggest that no decreases in tributary suspended solids inputs have occurred (Figure M.9). The temporal pattern observed in these data shows year-to-year variations, but overall the pattern appears steady over time. Overall, current evidence for decreasing sedimentation rates is limited, and hypotheses of future potential conditions are difficult to predict and quantify in terms of appropriate variations in the model sedimentation rate. Therefore, the rate of 0.25 g/cm^2/yr derived from recent data are used in the model described in Section M.3, and this rate is kept constant throughout the model projection period, subject to future adaptive management as appropriate.

M.2.4 Mercury Concentration in Settling Sediment

Current settling sediment mercury concentrations are estimated to be between 1.0 mg/kg and 1.9 mg/kg based on mercury concentration in surface sediment (0 to 2 cm) data collected from SMU 8 during 2007 and 2008. Shallow surface sediment data are a good indication of mercury concentrations in recently settled sediment, given that the lake bottom acts as a natural "sediment trap," and as noted above little or no mixing occurs in these bottom surface sediments. Average settling sediment mercury concentrations are assigned to three sub-areas of SMU 8 based on variability observed in the 0 to 2 cm recent surface sediment data. As such, the model may be over-predicting the settling sediment concentration at some stations and under-predicting in others. Sediment concentrations in the North Basin range from 0.7 mg/kg to 1.3 mg/kg (one outlier at 3.1 mg/kg was removed), with a mean of 1.1 mg/kg. To improve model calibration in the North Basin calibration stations (which were selected for calibration based on their longer available record of surface sediment concentrations), a slightly lower value of 1.0 mg/kg was used. Concentrations in recent surface sediment data in Ninemile Creek Outlet Area, the Saddle area, and the South Basin range from 1.0 mg/kg to 1.8 mg/kg, with an average concentration of 1.5 mg/kg. A value of 1.4 mg/kg was used for calibration stations in these areas to provide a better calibration at those points. Concentrations in the South Corner tend to be higher than in other areas of the lake. Mercury concentrations measured on sediments collected from the 2009 sediment traps deployed at South Deep range from 0.18 mg/kg to 3.5 mg/kg, with an average of 1.66 mg/kg. Concentrations from surface sediment data range from 1.5 mg/kg to 2.3 mg/kg, with an average of 1.9 mg/kg. This latter value was used for calibration in South Corner stations.

After remediation of the littoral zone and upland sources are complete in the year 2017, the incoming mercury sediment concentration is expected to decrease significantly starting in the year 2018. The future post-remediation incoming mercury concentration of 0.4 mg/kg was estimated based on evaluations of potential future mercury sources, including tributary influences and resuspended sediments from the littoral zone, as discussed below. Loads from Metro were not included in the assessment of future mercury concentrations because Onondaga County has apparently not yet started to measure low-level mercury in the wastewater treated at Metro and discharged to Onondaga Lake. However, based on extensive sediment mercury monitoring in the lake, inputs of mercury to the lake from Metro are not expected to significantly affect sediment mercury concentrations entering the lake at this time or in the future.

Future mercury concentrations in settling sediment were estimated based in part on tributary sediment mercury concentrations. Sediment in tributaries or portions of tributaries outside the area being remediated on behalf of Honeywell can be assessed to quantify sediment mercury concentration settling in the lake profundal zone in the future after Honeywell sites are remediated. Tributaries outside the areas being remediated by Honeywell have an average surface sediment concentration of 0.4 mg/kg. Average surface sediment mercury concentrations in four different tributaries are similar. Surface sediment mercury concentrations in lower Onondaga Creek are available from samples collected and analyzed at nine locations on behalf of Honeywell during 2009; the arithmetic average of those concentrations is 0.4 mg/kg (Parsons, Exponent, and Anchor QEA 2010). Surface sediment mercury concentrations in upper Geddes Brook are available from samples collected and analyzed at approximately ten locations over many years; those concentrations ranged from 0.03 to 0.18 mg/kg with the exception of two samples at one location that contained 1.3 and 1.6 mg/kg of mercury (TAMS Consultants 2003; Parsons 2005). Surface sediment mercury concentrations in upper Ninemile Creek are available from over 40 samples collected and analyzed from various locations over many years; the arithmetic average of those concentrations is also 0.4 mg/kg. Finally, for lower Ley Creek, surface sediment mercury concentrations are available from samples collected and analyzed at six locations on behalf of Honeywell during 2009 (Parsons, Exponent, and Anchor QEA 2010) and from many locations collected and analyzed the same year for USEPA. Sediment mercury results from sediment samples collected in lower Ley Creek on behalf of Honeywell ranged from 0.04 to 0.56 mg/kg. Sediment mercury results from sediment samples collected in lower Ley Creek on behalf of USEPA ranged from 0.028 to 0.8 mg/kg with the exception of 9 of the 120 results that had a maximum sediment mercury concentration of 2.1 mg/kg. As part of the baseline monitoring program conducted on behalf of Honeywell, mercury concentrations on solid particles were measured at Spencer Street in Onondaga Creek. The results of these two samples are 0.2 mg/kg and 0.6 mg/kg for mercury on solid particles based on total and filtered mercury and total suspended solids (TSS) results from the Book 3 baseline monitoring work. As part of a snowmelt and storm event sampling conducted by Syracuse University in April, June, and August of 2009, particulate mercury concentrations were measured in Onondaga Creek (Driscoll 2010). The concentrations in Onondaga Creek at the Spencer Street site were highly variable with a mean particulate mercury concentration of 0.28 mg/kg (standard deviation of 0.81 mg/kg). Particulate mercury concentrations measured in Onondaga Creek at the Dorwin site are much lower and more uniform with a mean particulate mercury concentration of 0.083 mg/kg (standard deviation of 0.059 mg/kg). The average particulate mercury concentration for the two Onondaga Creek sites is 0.17 mg/kg (standard deviation of 0.56 mg/kg).

Sediment mercury concentrations in the littoral zone will change substantially as a result of active sediment remediation, and this will reduce the overall average littoral sediment mercury concentration available for resuspension and possible deposition in the profundal zone. The future littoral sediment concentration following remediation was estimated to be 0.4 mg/kg based on average mercury concentrations in areas of the littoral zone that will not be remediated and based on a mercury concentration in the cap material of 0.1 mg/kg for areas that will be remediated (Appendix N). The average mercury concentration in areas of the littoral zone that will not be remediated is also based on surface sediment mercury data collected in the littoral zone since 1992. The cap material mercury concentration of 0.1 mg/kg is the same concentration as fill material applied as part of the remediation at Linden Chemical and Plastics (LCP; Parsons 2009).

Based on this tributary and post-remediation littoral area information, a conservative mercury concentration of 0.4 mg/kg on settling sediment was used for MNR modeling of the 10-year MNR period that will begin following completion of lake remediation efforts.

The value of 0.4 mg/kg is higher and more conservative than the estimate of 0.28 mg/kg provided in the Onondaga Lake FS, which was estimated based on a 70.5 percent reduction in mercury load due to the following remediation scheduled to be completed by 2017:

- Remediation of Harbor Brook, LCP, and Ninemile Creek
- Metro upgrades completed in 2004 and 2005
- Elimination of groundwater inputs to the lake from Willis Avenue, Semet Ponds, and Harbor Brook

M.2.5 Upwelling Velocity of Porewater

Upwelling velocities of porewater were calculated from RI data and range from 0 to 4.6 cm/yr (Andrews 2008; Parsons 2004). The mean and median upwelling velocities are 0.54 cm/yr and 0.2 cm/yr, respectively. The observed upwelling velocities in the profundal zone are below 1 cm/yr, with the exception of two boring locations: Location P39 having an upwelling velocity of 2.5 cm/yr and location P65 having an upwelling velocity of 4.6 cm/yr. Twelve of the 30 boring locations have a calculated upwelling velocity of 0 cm/yr. A conservative estimated average upwelling velocity of 1 cm/yr was therefore used in the modeling presented here. In a depositional environment such as the profundal zone of Onondaga Lake, sediment deposition provides a substantially greater flux of mass than the upwelling velocity, given the high partition coefficient; therefore, predicted mercury concentrations in the mixed layer are relatively insensitive to changes in upwelling velocity.

M.2.6 Mercury Partition Coefficient (Kd)

During the Preliminary Design Investigation (PDI), paired porewater and sediment samples were collected from SMUs 1, 3, 4, 6, and 7 and analyzed for mercury. Calculations were performed on these data to develop site-specific mercury partitioning coefficients (Kd). Samples from SMU 4 stations were used to calculate a representative Kd, as sampling in SMU 8 for this purpose was not conducted. SMU 4 was selected for its lack of Solvay waste material. The site-specific log Kd calculated from the Phase IV pre-design data was 5.6 L/kg (Kd of approximately 400,000 liters per kilogram [L/kg]; Parsons 2010). The model has been updated to these more accurate values (Table M.1), which do not appreciably impact the model calibration as shown in the model calibration discussed in Section M.3.3).

M.2.7 Initial Buried Layer Sediment Mercury Concentration

Generally, buried total mercury concentrations (deeper than 10 cm) have higher concentrations in the profundal sediments than more shallow sediment, consistent with recent natural recovery. A range of potential values for the buried layer sediment mercury concentration is shown in Table M.1. To be conservative for this submittal, an upper value of 20 mg/kg was used for the buried layer. To advance our modeling in future efforts, the concentrations measured from the deeper sediment at each station may be used, though this is not expected to have a great impact on the results.

M.2.8 Molecular Diffusion Coefficient

This value was obtained using the following equation using the molecular weight (MW) of elemental mercury (DiToro et al. 1981).

$$D_o = 6935 \times MW^{-2/3}$$

M.2.9 Mixed Layer Porosity

The porosity value of 0.91 was used based on an evaluation of density data provided by TAMS Consultants/New York State Department of Environmental Conservation (NYSDEC) during preparation of the FS. That evaluation used percent moisture data from Hairston et al.'s 1997 core (provided by TAMS Consultants to Honeywell/Anchor Environmental via email on July 16, 2004) in the top 0 to 4 cm and an assumed specific gravity (noted below).

M.2.10 Buried Layer Porosity

The porosity value of 0.86 was used based on the density evaluation provided by TAMS Consultants/NYSDEC during preparation of the FS using the same evaluation noted for the mixed-layer porosity and slightly deeper layers in those cores.

M.2.11 Biodiffusion (or Mixing) Coefficient

Boudreau presents a relationship between this parameter and burial velocity based on empirical data (1997).

$$D_b = 15.7 * s^{0.69}$$

A settling sediment flux of g/cm^2/yr (w) was converted to a burial velocity in cm/yr based on porosity (j) and particle specific gravity (SG) of the sediment using the following equation:

$$S = \frac{w}{(1 - j) * SG}$$

M.2.12 Specific Gravity of Dry Sediment

This value is known as particle density, and values observed from 2007 PDI cores range from 2.5 to 2.8 g/cm^3. A typical value of 2.7 g/cm^3 is used for this model. Specific gravity is used to determine the *in situ* density of the mixed layer using the porosity (derived from water content as noted above) and relationship noted for biodiffusion (e.g., (1-j)*SG).

M.2.13 Initial Mixed Layer Sediment Mercury Concentrations

To assess the rate of natural recovery of sediments relative to the mercury PEC and BSQV goals, sediment mercury concentrations in the top 4 cm were applied as model input for the initial mercury concentration in the mixed layer. To assess whether natural recovery of sediments are on track to meet the mercury PEC, sediment mercury concentrations in the top 4 cm were applied as model input for the initial mercury concentration in the mixed layer. Table M.4 lists the mercury concentrations in the 0 to 4 cm depth interval used for the initial mercury concentrations in SMU 8 sediment. In cases where multiple sections made up the 0 to 4 cm interval, a weighted averaging approach was used as follows:

$$\frac{\sum M_{ci}}{\sum M_{si}} = C_{\sum i}$$

Where:

$M_{ci} = \rho_{bi} \cdot V_i \cdot C_i$

$M_{si} = \rho_{bi} \cdot V_i$

$\rho_{bi} = (WC \cdot SG_w) + \big((1 - WC) \cdot SG_s\big)$

C = concentration of chemical over combined intervals (i) in mg/g (note: this can be converted to mg/kg by multiplying by 1,000 [g/kg])

M_{ci} = mass of chemical in interval i (mg)

M_{si} = mass of sediment in interval i (g)

ρ_{bi} = In-situ density of sediment in interval i (g/cm^3)

V_i = volume of interval i, calculated assuming a constant surface area of 1 cm^2 times the interval depth in cm (cm^3); this assumption is valid based on the fact that the diameter (and, hence, area) of the core tubes used for sampling would be equal between samples at a given location

C_i = concentration of chemical measured in interval i (mg/g)

WC = water content measured in sediment (proportion)

SG_w = specific gravity of water (assumed to be 1.0) (g/cm^3)

SG_s = measured specific gravity of dry sediment (g/cm^3)

Concentrations of mercury in the top 4 cm of the core locations modeled, as shown in Figures 6.3a and 6.3b, range from 0.84 to 5.39 mg/kg.

M.3 MODEL CALIBRATION AND SENSITIVITY ANALYSES

M.3.1 Boundary Conditions

The one-dimensional sediment mixed layer mass or concentration, which is the primary focus of the model, is bounded by surface water at the top (x=0) and buried layer at the bottom (x=L).

The concentration of mercury in lake surface water is assumed to be zero. Generally, surface water concentrations are well below porewater chemical concentrations (particularly for contaminated sediments), so the use of a zero value for the surface water boundary condition does not significantly affect predictions of natural recovery. That is, the model is insensitive to small changes in surface water concentration. The primary input from the surface water is the flux of suspended sediment (and associated chemicals) settling on the mixed-layer bed. The chemical concentration in the buried layer boundary was held constant at 20 mg/kg, which is generally representative of the higher range of buried mercury concentrations in SMU 8. As shown in Section M.3.4, the model is also insensitive to changes in buried sediment mercury concentrations. The model is also relatively insensitive to changes in the dissolved advection/diffusion over the subsurface boundary. Consequently, the general assumption of 20 mg/kg of mercury in buried sediment was applied for each modeled location.

M.3.2 Sediment Locations Modeled

Locations sampled during the PDI (2007 to 2010) were considered as model projection locations, because they represent the most recent surface sediment data set. Figure M.10 shows these SMU 8 locations by sample year for each portion of the lake's profundal zone. The one-dimensional model assesses the fate and transport of mercury along the vertical axis of the sediment bed and, therefore, each location shown in Figure M.10 was modeled separately. Modeled locations provide good coverage of the various sections of the lake: North Basin, Saddle, Ninemile Creek Outlet Area, South Basin, and South Corner.

M.3.3 Model Calibration

The model was calibrated during the FS effort (Parsons 2004) based on empirical time series sediment mercury data available at that time. More recent surface sediment mercury data are available as noted in the previous subsection, so more recent model calibration work was performed as part of this Draft Final Design effort. This model calibration accounts for new data collected by Honeywell during the PDI (2005 through 2010). Results show the model calibrates well to the pre-design data from the top 4 cm using settling sediment mercury concentrations noted in Section M.2.4 (Figure M.11). In general, the model is conservatively calibrated, meaning that it is within the range of data or typically over-predicts the observed mercury concentrations in sediment. At location S24 the model under-predicts the mercury concentrations in sediment, which may be due to its closer proximity to the remediation areas in the littoral zone as compared to any other location modeled (Figure M.12). The model calibration is shown at various sedimentation rates and mixing depths.

M.3.4 Model Sensitivity

The model was evaluated for sensitivity to various input parameters. Because all model parameters were varied, including the site-specific parameters such as initial sediment mercury concentration and settling mercury concentrations, it was necessary to select only one model location for this analysis: OL-STA-80070, with an initial mercury concentration of 5.39 mg/kg. The model was evaluated one parameter at a time; while one parameter was varied, the other model input parameters were set to the calibrated values, as described in Section M.2. The exception is for the initial settling sediment mercury concentrations, which was kept constant throughout the model period. Results of this sensitivity analysis are provided in Table M.3.

The model is sensitive to variations in sedimentation rate, settling sediment mercury concentrations (initial and future concentrations after 2017), mixed-layer depth, buried layer partition coefficient, and mixed layer porosity inputs. The mixed-layer depth sensitivity is expected, because sediment mixing depth defines the size of the “reservoir” that is impacted by transport processes. A larger reservoir will show less responsiveness to variations in flux to and from the mixed layer over time. Porosity is sensitive for the same reason; it is the primary factor determining the in situ density of sediments present in the mixed layer. The model is sensitive to the sediment settling mercury concentrations because the settling concentration largely defines the sediment concentration that will eventually make up the mixed layer (sediment settling mercury concentrations). As particles from the water column deposit on the sediment bed, the settling particles become part of the mixed layer. The sedimentation rate defines the speed at which the newly deposited particles build up to the mixed layer depth. The model is sensitive to the buried layer partition coefficient at high buried layer concentrations. Advection of mercury in the dissolved phase is calculated from the buried layer concentration and the partitioning from the solid phase concentration input into the model. The less mercury partitions to the sediment, the more dissolved mercury is released via advection to the mixed layer from the buried layer. At buried layer concentrations less than 10 mg/kg mercury, the change in buried layer partition coefficients has little impact on the mixed layer mercury concentrations.

The model is relatively insensitive to changes in mixed layer mercury partition coefficient, buried layer mercury concentration, and upwelling velocity. Additional runs of the model indicated that the mixed layer mercury partition coefficient would have to be considerably lower (in the 1,000 to 10,000 L/kg) range before any of these parameters would have a substantial effect on the model. Thus, because mercury appears to be strongly associated with the sediment particles, processes involving particule movement dominate over dissolved-phase transport processes like porewater advection. This means that stable layers of new sediment will effectively isolate older layers of even highly contaminated sediment. This finding is consistent with the distinct variations with depth in the mercury concentration core profiles, indicating that dissolved phase transport has not “smeared” these profiles over time. This finding also indicates that the particulate phase processes of sedimentation and incoming concentrations of settling sediments have the greatest impact on the model results. As noted before, there is a considerable amount of information to support the values used in the modeling for these two inputs (Table M.1). Therefore, the uncertainty associated with use of the mid-range value is low.

M.4 MODEL RESULTS

The one-dimensional numerical model was applied to predict the mercury concentrations in sediment at multiple locations 10 years following dredging and capping. The sediment mercury concentration at that time, which is assumed to be the year 2027, was compared to the PEC of 2.2 mg/kg for mercury (for the top 4 cm) and the BSQV of 0.8 mg/kg mercury (also for the top 4 cm). As discussed in Section M.3.2, locations modeled were those where 0 to 4 cm samples were taken (Figure M.10).

M.4.1 Comparison to Mercury PEC

Based on model results, all SMU 8 sediments are predicted to achieve the mercury PEC remediation goal within the 10-year MNR period (Table M.4). Results from the modeling predict that sediment mercury concentrations in the top 4 cm will be 0.48 to 0.58 mg/kg at the end of the 10-year MNR period (i.e., the year 2027). The PEC of 2.2 mg/kg for mercury is predicted to be achieved at all modeled locations by the year 2018, which is Year 2 of the 10-year MNR monitoring period. Figure M.13 shows the temporal trend of the mercury concentration in the top 4 cm at each modeling location for the projection period. As described in Section M.2.4, the average settling sediment concentration decreases starting in the year 2018, after remediation of the littoral zone and upland sources are complete. The temporal trend changes at year 2018 due to the decreased settling sediment concentrations and the mixed layer begins to equilibrate with the newly settled concentrations. At some locations (e.g., OL-VC-80210 located in the South Basin), it appears that concentrations in the surface sediment are increasing prior to the year 2018 which is an artifact of the use of an average surface settling concentration for a subarea of SMU 8 that is higher than the model location-specific initial mercury concentration in the mixed layer. Other factors are likely contributing to the settling sediment mercury concentrations than can be accounted for on a small spatial scale.

To effectively model numerous lake sediment locations, timeframes represented in the model runs begin on January 1 of each year. Samples collected are assigned a model start date of January 1 in the following year. For example, Sample OL-STA-80067 was collected during November 2007, so the model start date for assessing natural recovery at that location is January 1, 2008. On this basis, the year the mercury concentration falls below the mercury PEC at location OL-STA-80067 is 2008.

M.4.2 Comparison to BSQV

Based on model results, all SMU 8 sediments are also predicted to achieve the mercury BSQV remediation goal within the 10-year MNR period (Table M.4 and Figure M.13). Unlike the PEC, the BSQV is meant to be applied on an area-wide basis; therefore, the BSQV of 0.8 mg/kg was compared to sediment mercury concentrations on an area-wide basis, as discussed in Appendix N. Areas of influence (based on Thiessen polygons) for each modeled profundal zone location are presented in Figure M.14. The area-weighted surface sediment mercury concentration in the littoral zone is projected to be 0.4 mg/kg following remediation of the littoral zone based on cap cover material containing 0.1 mg/kg and based on parceling the littoral zone into Thiessen polygons and applying surface sediment mercury concentrations measured in the littoral zone outside the remediation areas since 1992. On an area-weighted basis following the calculation method described in Appendix N, the mercury concentrations in each area of the lake are well below the BSQV of 0.8 mg/kg as shown in the table below.

Onondaga Lake Sub-area (Littoral and Profundal Zones_	**Area-weighted Average Mercury Concentration Predicted for the Year 2027 (mg/kg)**	**Year BSQV (0.8 mg/kg) is Met in Surface Sediment**
North Basin	0.61	2019
Ninemile Creek	0.39	2017
Saddle	0.49	2019
South Basin	0.49	2020
South Corner	0.34	2018

The area-weighted surface sediment mercury concentration is predicted to fall below the BSQV of 0.8 mg/kg in each sub-area of the lake by the year 2020, which is the third year of the 10-year MNR monitoring period.

M.5 APPENDIX A REFERENCES

Anchor Environmental, L.L.C., 2002. *Final Cleanup Study Report Duwamish/Diagonal CSO/SD*. Prepared for King County, Washington Department of Natural Resources, Seattle, Washington.

Anchor Environmental, L.L.C. and Foster Wheeler, 2001. *Final Design Submittal for Middle Waterway Problem Area*. Prepared for Middle Waterway Action Committee, Seattle, Washington.

Andrews, C., 2008. Memorandum to Onondaga lake Groundwater Upwelling Technical Working Group, December 4.

Berner, R.A., 1980. Early Diagenesis: A Theoretical Approach. Princeton University Press. Princeton, New Jersey.

Boudreau, B., 1997. Diagenetic Models and Their Implementation: Modeling Transport Reactions in Aquatic Sediments. New York: Springer.

Burns, N. M., 1995. Using Hypolimnetic Dissolved Oxygen Depletion Rates for Monitoring Lakes. N. Z. J. Mar. Freshw. Res. 29:1–11.

Cowen, E.A. and P.J. Rusello, 2008. Memorandum on Fall 2007 Preliminary Field Observations of Hypolimnetic Turbulence Levels in Onondaga Lake. DeFrees Hydraulics Laboratory, School of Civil and Environmental Engineering, Cornell University

Denkenberger, J. S., C. T. Driscoll, S. W. Effler, D. M. O'Donnell and D. A. Matthews, 2007. Comparison of an Urban Lake Targeted for Rehabilitation and a Reference Lake Based on Robotic Monitoring. Lake and Reserv. Manage. 23:11-26.

DiToro, D.M., P.J. O'Conner, R.V. Thomann, and J.P. St. John, 1981. *Analysis of Fate of Chemicals in Receiving Waters, Phase I*. Prepared for Chemical Manufacturer Association, Washington, D.C.

Driscoll, C.T. Jr., 2010. Regarding: Particulate Hg OC. Email to: Betsy Henry. November 15, 2010

Effler, S.W., M.T. Auer, N. Johnson, M. Penn, and H.C. Rowell, 1996. Sediments. In S.W. Effler, ed. *Limnological and Engineering Analysis of a Polluted Urban Lake: Prelude to Environmental Management of Onondaga Lake, New York*. 600-666. New York: Springer-Verlag.

Effler, S.W. and G. Harnett, 1996. Background. In S.W. Effler, ed. Limnological and *Engineering Analysis of a Polluted Urban Lake: Prelude to Environmental Management of Onondaga Lake*, New York. 1-31. New York: Springer-Verlag.

Exponent, Michigan Technological University, Upstate Freshwater Institute, and Syracuse University, 2010. *Data Report: Sediment Incubations and Supporting Studies for Onondaga Lake Sediment Management Unit (SMU) 8*. Prepared for Honeywell. June.

Hairston, N. and S. Effler, 2007. Figures documenting macroinvertebrate extent and depth of penetration in Onondaga Lake. Unpublished. Provided to Parsons by N. Hairston. June 2007.

Hairston, N.G. Jr., L.J. Perry, A.J. Bohonak, M.Q. Fellow, and C.M. Kearns, 1999. Population Biology of a Failed Invasion: Paleolimnology of *Daphnia exilis* in upstate New York. Limnology and Oceanography 44.3: 477-486.

Hurteau C.A., D.A. Matthews, and S.W. Effler, 2010. A Retrospective Analysis of Suspended Solids Deposition in Onondaga Lake, New York: Composition, Temporal Patterns, and Drivers. Lake and Reservoir Management. 26:43-53.

Magar, V.S., D.B. Chadwick, T.S Bridges, P.C. Fuchsman, J.M. Conder, T.J. Dekker, J.A. Stevens, K.E. Gustavson, and M.A. Mills, 2009. Monitored Natural Recovery at Contaminated Sediment Sites. Technical Guide, Environmental Security Technology Certification Program (ESTCP) Project ER-0622. May 2009.

Matthews, D. A. and S. W. Effler, 2006. Long-Term Changes in the Areal Hypolimnetic Oxygen Deficit (AHOD) of Onondaga Lake: Evidence of Sediment Feedback. Limnol. Oceanogr. 51:702–714.

Owens, E.M. and S.W. Effler, 1996. *Hydrodynamics and Transport. In Limnological and Engineering Analysis of a Polluted Urban Lake: Prelude to Environmental Management of Onondaga Lake, New York.* edited by S.W. Effler. New York.: Springer-Verlag, 200-201.

Parsons, 2004. *Onondaga Lake Feasibility Study Report.* Onondaga County, NY. Three Volumes. Prepared for Honeywell. Draft Final (final version). November. Appendix N: Monitored Natural Recovery prepared by Anchor Environmental, Exponent, and Papadopulos and Associates.

Parsons, 2005. *Geddes Brook / Ninemile Creek Feasibility Study Report*. Prepared for Honeywell. Draft Final. May.

Parsons, 2009. *Final Remedial Action Report for the Soil Washing, Soil and Sediment Consolidation, Sewers, Slurry Wall, Groundwater Containment / Pretreatment and Interim Soil Cover at the LCP Bridge Street Site (OU-1).* Prepared for Honeywell. November.

Parsons, 2010. *Onondaga Lake Pre-Design Investigation Phase IV Data Summary Report*. Prepared for Honeywell. December.

Parsons, Anchor Environmental, and Environmental Tracing Systems, 2008. *Microbead Marker Work Plan for Monitoring Natural Recovery in SMU 8.* Prepared for Honeywell. September.

Parsons and Environmental Tracing Systems, 2010. *Microbead Marker Placement Report*. Prepared for Honeywell. May. Draft.

Parsons, Exponent, and QEA, 2008. *Onondaga Lake Baseline Monitoring. Book 2 Work Plan. Fish, Invertebrate, and Littoral Water Monitoring for 2008*. Prepared for Honeywell. September.

Parsons, Exponent, and Anchor QEA, 2009. *Onondaga Lake Baseline Monitoring Report for 2008*. Prepared for Honeywell. June. Draft.

Parsons, Exponent, and Anchor QEA, 2010. *Onondaga Lake Baseline Monitoring Report for 2009*. Prepared for Honeywell. July. Draft.

Parsons and Upstate Freshwater Institute (UFI), 2010. *Report for the Nitrate Application Field Trial in the Hypolimnion of Onondaga Lake (Sediment Management Unit 8*). Prepared for Honeywell. March.

Parsons and Upstate Freshwater Institute (UFI), 2011. *Work Plan for Pilot Test to Add Nitrate to the Hypolimnion of Onondaga Lake* . Prepared for Honeywell. March.

Rowell, H.C., 1992. Paleolimnology, Sediment Stratigraphy, and Water Quality History of Onondaga Lake, Syracuse, NY. Dissertation. State University of New York, College of Environmental Science and Forestry, Syracuse, New York.

Sharpe, C., 2004. Mercury Dynamics of Onondaga Lake and Adjacent Wetlands. *Masters Thesis: Syracuse University*. 1996.

TAMS Consultants, Inc., 2002. *Onondaga Lake Remedial Investigation Report*. Prepared with YEC, Inc. for NYSDEC, Division of Environmental Remediation, Albany, New York.

TAMS Consultants, Inc., 2003. *Geddes Brook / Ninemile Creek Remedial Investigation Report*. Prepared for NYSDEC. July.

United States Environmental Protection Agency, 2005. *Contaminated Sediment Remediation Guidance for Hazardous Waste Sites*. OSWER 9355.0-85, EPA-540-R-05-012. Office of Solid Waste and Emergency Response, Washington, D.C. December.

Upstate Freshwater Institute (UFI), 2010. *A Retrospective Analysis of Suspended Solids Deposition in Onondaga Lake, NY.* Prepared for Honeywell. March.

Wetzel R. G., 2001. *Limnology: Lake and River Ecosystems. 3rd ed.* San Diego (CA): Academic Press.

Ziegler, C.K., 2002. *Evaluating Sediment Stability at Sites with Historic Contamination. Environmental Management*. Vol. 29, No. 3, pp.409 – 427. Springer-Verlag New York.

TABLES

Table M.1 Input Parameters and Source Information for Natural Recovery Modeling of Total Mercury

Parameter	Symbol	Units	Range of Input Values			Information Sources	MNR Model Input Parameters
			Low	Mid	High		
Initial Mixed Layer Concentration	B_{LT}	mg/kg	0.64	1.75	3.9	Range of values found in surface layer profundal sediments throughout lake (PDI).	The input varies for each model location. The initial mixed layer concentration is set to the mercury concentration from latest sampling year at that model location, using the 0 to 4 cm interval (see Table A.4)
Initial Buried Layer Concentration	B_{BT}	mg/kg	1.5	4	20	Range of values found in profundal and deep littoral sediments throughout lake (TAMS, 2002).	20 as a conservative assumption[a]
Existing Settling Sediment Concentration	B_0	mg/kg	0.7	1.6	3.1	Concentrations observed in the 0-2cm depth interval from 2007/2008 PDI cores. Mid-range is the mean of the samples, and low and high values represent the range of concentrations.	Until upland remediation is complete in 2017: 1.4 mg/kg for Saddle, Nine Mile Creek, and South basin portions of SMU 8 1.0 mg/kg for North Basin 1.9 mg/kg for South Corner
Future Settling Sediment Concentration	B_0	mg/kg	--	0.4	--	Considerations include tributary influences and resuspended sediments from the littoral zone	0.4 after upland remediation is complete in 2017
Partition Coefficient, Mixed Layer	K_d	L/kg	145,332	398,107	1,161,971	SMU 4 paired sediment and porewater data (PDI Phase IV).	398,107[a,b]
Partition Coefficient, Buried Layer	K_d	L/kg	145,332	398,107	1,161,971	SMU 4 paired sediment and porewater data (PDI Phase IV).	398,107[a,b]
Molecular Diffusion Coefficient	D_o	cm^2/yr	--	202	--	Calculated per DiToro *et al*., 1981 ($D_o = 6935*MW^{-2/3}$)[c]	202
Mixed Layer Porosity	*j*	unitless	--	0.91	--	Calculated from estimates of bulk density provided by NYSDEC (0.25 g/cm^3, which is the NYSDEC recommended value - Based on Hairston 1997 core, percent moisture and assumed SG value (2.7) for 0 - 4 cm and consistent with 2007 Honeywell analyses).	0.91
Buried Layer Porosity	*j*	unitless	--	0.86	--	Calculated from estimates of bulk density provided by NYSDEC (0.39 g/cm^3, which is the NYSDEC recommended value - Based on Hairston 1997 core, percent moisture and assumed SG value (2.7) for 4-10 cm and consistent with 2007 Honeywell analyses).	0.86
Biodiffusion Coefficient	D_b	cm^2/yr	--	16.01	--	Boudreau, 1997, Equation 4.148. $D_b = 15.7*s^{0.69}$ and s = w / ((1-j)*SG)[d]	16.01
Velocity of Porewater	*u*	cm/yr	0		4.6	Andrews, 2008 and Parsons 2004. All but two upwelling velocities from SMU 8 boring locations sampled during the RI are less than 1 cm/yr.	1[a]
Settling Sediment Flux	*w*	$g/cm^2/yr$	0.080	0.280	0.780	Post-1986 sediment trap and core data (See Table A.2).	0.25
Specific Gravity of Dry Sediment	SG	g/cm^3	--	2.7	--	Typical value.	2.7

Notes:

[a] Model simulations are not significantly sensitive to this parameter (see Section N.3.4). Adjusting these values across a wide range will not significantly affect model results.

[b] Partition coefficient consistent with more recent SMU 4 partition coefficient data based on paired sediment and porewater cores collected from SMU 4 during Phase IV of the PDI.

[c] D_o is the molecular diffusion coefficient, MW is the molecular weight of mercury.

[d] D_b is the biodiffusion coefficient; s is the burial velocity (cm/yr), w is the settling sediment flux ($g/cm^2/yr$), j is porosity, SG is specific gravity.

Table M.2 Sedimentation rates from core and sediment trap data.

Type	Source	Evaluation Period		Minimum	Mid-Range	Maximum	Units	Sedimentation Rates
		Start Year	End Year					($g/cm^2/yr$)[a, b,c]
Core Data	Direct evaluation of RI 1996 Core Data	1953	1963	0.625	0.75	0.875	cm/yr	0.18
Core Data	Direct evaluation of RI 1996 Core Data	1964	1970	0.536	0.714	0.893	cm/yr	0.17
Core Data	Direct evaluation of RI 1996 Core Data	1971	1996	0.577	0.721	0.769	cm/yr	0.18
Core Data	Effler, 1996 (Cs 137 Cores pp. 648, 655)	1954	1963	0.444	0.722	1	cm/yr	0.18
Core Data	Effler, 1996 (Cs 137 Cores pp. 648, 655)	1964	1988	0.574	0.595	0.616	cm/yr	0.14
Core Data	Effler, 1996 (mercury cor p. 634)	1946	1970	-	0.42	-	cm/yr	0.10
Core Data	Effler, 1996 (Pb 210 cores p. 649)	1955	1988	0.909	1.212	1.515	cm/yr	0.29
Core Data	Hairston et al. 1999	1981	1981	0.667	0.756	0.874	cm/yr	0.18
Core Data	Hairston et al. 1999	1984	1984	1.052	1.244	1.481	cm/yr	0.30
Core Data	Hairston et al. 1999	1987	1987	0.504	0.563	0.622	cm/yr	0.14
Core Data	Hairston et al. 1999	1993	1993	0.341	0.37	0.385	cm/yr	0.09
Core Data	Hairston et al. 1999	1997	1997	0.293	0.326	0.341	cm/yr	0.08
Core Data	TAMS, 2002 (discussion of RI 1992 deep cores)	1954	1964	-	1.1	-	cm/yr	0.27
Core Data	TAMS, 2002 (discussion of Rowell 1992 cores)	1954	1964	-	0.28	-	cm/yr	0.07
Core Data	TAMS, 2002 (discussion of Rowell 1992 cores)	1964	1988	-	0.83	-	cm/yr	0.20
Core Data	TAMS, 2002 Fig. 6-28 (RI 1992 cores)	1954	1963	0.9	1.1	1.5	cm/yr	0.27
Core Data	TAMS, 2002 Fig. 6-28 (RI 1992 cores)	1963	1992	0.828	0.897	1.034	cm/yr	0.22
Core Data	TAMS, 2002 Fig. 6-29 (RI 1996 Cores)	1954	1963	0.7	0.8	1	cm/yr	0.19
Core Data	TAMS, 2002 Fig. 6-29 (RI 1996 Cores)	1964	1996	0.697	0.827	0.788	cm/yr	0.20
Core Data	TAMS, 2002 Fig. 6-30 (Rowell 1992)	1954	1963	0.333	0.556	0.778	cm/yr	0.14
Core Data	TAMS, 2002 Fig. 6-30 (Rowell 1992)	1964	1988	0.833	0.875	0.917	cm/yr	0.21
Core Data	2008 High Resolution Core Data (OL-STA-80068)	2008	2008	0.13	0.13	0.14	$g/cm^2/yr$	0.13
Core Data	2008 High Resolution Core Data (OL-STA-80073)	2008	2008	0.34	0.35	0.35	$g/cm^2/yr$	0.35
Core Data	2008 High Resolution Core Data (OL-STA-80076)	2008	2008	0.22	0.25	0.27	$g/cm^2/yr$	0.25
Core Data	2008 High Resolution Core Data (OL-STA-80089)	2008	2008	0.26	0.26	0.27	$g/cm^2/yr$	0.26
Core Data	2008 High Resolution Core Data (OL-STA-80103)	2008	2008	0.26	0.28	0.31	$g/cm^2/yr$	0.28
Core Data	2008 High Resolution Core Data (ST-51)	2008	2008	0.25	0.25	0.26	$g/cm^2/yr$	0.25
Core Data	2008 High Resolution Core Data (ST-51A)	2008	2008	0.25	0.27	0.30	$g/cm^2/yr$	0.27
Sediment Trap Data	TAMS, 2002 1992 Sediment Traps (Table 6-19)	1992	1992	0.27	0.487	0.762	$g/cm^2/yr$	0.49
Sediment Trap Data	TAMS, 2002 1992 Sediment Traps (Table 6-19)	1992	1992	0.27	0.448	0.654	$g/cm^2/yr$	0.45
Sediment Trap Data	Direct evaluation of RI 1996 Sediment Trap Data	1996	1996	0.106	0.48	1.153	$g/cm^2/yr$	0.48
Sediment Trap Data	Effler, 1996 Sediment Traps 1986 (pp. 606-607)	1986	1986	0.806	2.049	3.558	$g/cm^2/yr$	2.05
Sediment Trap Data	Effler, 1996 Sediment Traps 1986 (pp. 606-607)	1988	1988	0.162	0.622	1.373	$g/cm^2/yr$	0.62
Sediment Trap Data	Sharpe, 2003 Sediment Traps 2000	2000	2000	0.138	0.317	0.53	$g/cm^2/yr$	0.32
Sediment Trap Data	2009 Sediment Trap Data	2009	2009	0.10	0.28	0.78	$g/cm^2/yr$	0.28
Sediment Trap Data	2010 Sediment Trap Data	2010	2010	0.09	0.34	0.75	$g/cm^2/yr$	0.34

Notes:

(a) Mid-Range values used for sedimentation rates. Conversion from cm/yr to $g/cm^2/yr$ based on dry bulk density values.

(b) Dry density values for high resolution cores from Appendix N converted to $g/cm^2/yr$ assuming a dry bulk density equal to 0.243 g/cc based on a porosity of 0.91 and a specific gravity of 2.7 g/cc. 2008 High resolution core data statistics are reported for the top two sections of the core.

(c) Although shown as annual averages, the sediment trap data do not consider sedimentation rates during the winter months, which may be lower.

Table M.3. Sensitivity analysis of model inputs.

Input Parameter	Input Value	Mercury Concentration (mg/kg) at the end of the MNR Period (2027)	Sensitivity Ratio
Settling Sediment Flux (g/cm2/yr)			
Minimum	0.02	3.93	
Maximum	6.22	0.40	
Relative Percent Difference	-199%	163%	0.82
Settling Sediment Mercury Concentration (mg/kg)[1]			
Minimum	0.65	0.72	
Maximum	1.9	1.96	
Relative Percent Difference	-98%	-93%	0.94
Reduced Settling Sediment Mercury Concentration (mg/kg)[2]			
Minimum	0.1	0.23	
Maximum	0.8	0.88	
Relative Percent Difference	-156%	-117%	0.75
Partition Coefficient in Mixed Layer (L/kg)			
Minimum	158,489.3	0.51	
Maximum	1,258,925.4	0.51	
Relative Percent Difference	-155%	0%	0.00
Partition Coefficient in Buried Layer (L/kg)			
Minimum	158,489.3	0.76	
Maximum	1,258,925.4	0.63	
Relative Percent Difference	-155%	19%	0.12
Buried Layer Mercury Concentration			
Minimum	1	0.47	
Maximum	20	0.51	
Relative Percent Difference	-181%	-8%	0.05
Porewater Velocity (cm/yr)			
Minimum	0	0.51	
Maximum	4.6	0.52	
Relative Percent Difference	-200%	-2%	0.01
Mixed Layer Depth (cm)			
Minimum	1	0.44	
Maximum	10	1.20	
Relative Percent Difference	-164%	-93%	0.57
Porosity Mixed Layer (unitless)			
Minimum	0.5	2.54	
Maximum	0.99	0.44	
Relative Percent Difference	-66%	141%	2.14

Notes:

[1] Unlike predictive modeling performed, the parameter held constant throughout the model period for the purposes of this sensitivity analysis.

[2] Reduced settling sediment mercury concentration begins after upland remediation is complete in the year 2017. Concentration from start of model to 2017 is 1.0 mg/kg at this North Basin Location.

Table M.4. Initial and final mercury sediment concentrations from model locations.

Area of Profundal Zone	Location ID	Initial Mercury Sediment Concentration (mg/kg)[1]	Final (2027) Mercury Sediment Concentration (mg/kg)	Year Mercury Sediment Concentration Is Below PEC (2.2 mg/kg)[2]	Year Mercury Sediment Concentration Is Below BSQV (0.8 mg/kg)[2]
North Basin	OL-STA-80067	1.19	0.49	2008	2020
North Basin	OL-STA-80068	0.84	0.48	2009	2020
North Basin	OL-STA-80069	1.25	0.49	2008	2020
North Basin	OL-STA-80070	5.39	0.51	2013	2022
North Basin	OL-STA-80071	1.64	0.49	2008	2020
North Basin	OL-STA-80072	1.29	0.49	2008	2020
North Basin	OL-VC-80157	1.30	0.49	2011	2020
North Basin	OL-VC-80158	1.20	0.49	2011	2020
North Basin	OL-VC-80159	1.30	0.49	2011	2020
North Basin	OL-VC-80198	1.50	0.49	2011	2020
North Basin	OL-VC-80199	0.87	0.48	2011	2020
North Basin	OL-VC-80200	1.00	0.48	2011	2020
North Basin	OL-VC-80201	0.85	0.48	2011	2020
North Basin	OL-VC-80202	0.86	0.48	2011	2020
Ninemile Creek	OL-STA-80073	1.25	0.51	2009	2022
Ninemile Creek	OL-STA-80074	1.70	0.52	2008	2022
Ninemile Creek	OL-STA-80091	1.80	0.52	2008	2022
Ninemile Creek	OL-VC-80160	2.40	0.53	2012	2023
Ninemile Creek	OL-VC-80161	2.00	0.52	2011	2022
Ninemile Creek	OL-VC-80162	2.60	0.53	2013	2023
Ninemile Creek	OL-VC-80163	1.70	0.52	2011	2022
Ninemile Creek	OL-VC-80164	1.20	0.51	2011	2022
Ninemile Creek	OL-VC-80203	1.10	0.51	2011	2022
Ninemile Creek	OL-VC-80204	1.00	0.51	2011	2022
Ninemile Creek	OL-VC-80205	1.30	0.51	2011	2022
Saddle	OL-STA-80075	1.75	0.52	2008	2022
Saddle	OL-STA-80103	1.35	0.51	2009	2022
Saddle	OL-VC-80206	1.80	0.52	2011	2022
South Basin	OL-STA-80076	1.45	0.51	2009	2022
South Basin	OL-STA-80077	1.50	0.52	2008	2022
South Basin	OL-STA-80078	1.65	0.52	2008	2022
South Basin	OL-STA-80079	1.80	0.52	2008	2022
South Basin	OL-STA-80080	1.50	0.52	2008	2022
South Basin	OL-STA-80081	1.84	0.52	2008	2022
South Basin	OL-STA-80082	1.70	0.52	2008	2022
South Basin	OL-STA-80083	1.81	0.52	2008	2022
South Basin	OL-STA-80084	1.94	0.52	2008	2022
South Basin	OL-VC-80165	1.90	0.52	2011	2022
South Basin	OL-VC-80166	2.00	0.52	2011	2022
South Basin	OL-VC-80167	1.50	0.52	2011	2022
South Basin	OL-VC-80168	2.30	0.53	2011	2022
South Basin	OL-VC-80169	1.70	0.52	2011	2022
South Basin	OL-VC-80207	1.20	0.51	2011	2022
South Basin	OL-VC-80208	1.30	0.51	2011	2022
South Basin	OL-VC-80209	1.20	0.51	2011	2022
South Basin	OL-VC-80210	1.20	0.51	2011	2022
South Basin	ST51	1.15	0.51	2009	2022
South Basin	ST-51a	1.35	0.51	2009	2022
South Corner	OL-STA-80085	1.90	0.55	2008	2024
South Corner	OL-STA-80086	1.89	0.55	2008	2024
South Corner	OL-STA-80087	1.94	0.55	2008	2024
South Corner	OL-STA-80088	2.30	0.55	2009	2024
South Corner	OL-STA-80089	2.06	0.55	2009	2024
South Corner	OL-STA-80090	2.49	0.56	2011	2024
South Corner	OL-STA-80092	2.25	0.55	2009	2024
South Corner	OL-VC-80170	1.90	0.55	2011	2024
South Corner	OL-VC-80171	1.70	0.55	2011	2023
South Corner	OL-VC-80172	1.40	0.55	2011	2023
South Corner	OL-VC-80173	1.40	0.55	2011	2023
South Corner	OL-VC-80174	1.30	0.54	2011	2023
South Corner	OL-VC-80175	1.70	0.55	2011	2023
South Corner	OL-VC-80176	1.90	0.55	2011	2024
South Corner	OL-VC-80177	1.60	0.55	2011	2023
South Corner	OL-VC-80178	1.60	0.55	2011	2023
South Corner	OL-VC-80179	1.40	0.55	2011	2023
South Corner	OL-VC-80180	1.60	0.55	2011	2023
South Corner	OL-VC-80181	2.00	0.55	2011	2024
South Corner	OL-VC-80182	1.80	0.55	2011	2023
South Corner	OL-VC-80183	2.50	0.56	2014	2024
South Corner	OL-VC-80184	2.30	0.56	2012	2024
South Corner	OL-VC-80185	2.20	0.56	2011	2024
South Corner	OL-VC-80186	2.10	0.55	2011	2024
South Corner	OL-VC-80187	1.70	0.55	2011	2023

Area of Profundal Zone	Location ID	Initial Mercury Sediment Concentration (mg/kg)[1]	Final (2027) Mercury Sediment Concentration (mg/kg)	Year Mercury Sediment Concentration Is Below PEC (2.2 mg/kg)[2]	Year Mercury Sediment Concentration Is Below BSQV (0.8 mg/kg)[2]
South Corner	OL-VC-80188	2.80	0.56	2016	2024
South Corner	OL-VC-80189	1.80	0.55	2011	2023
South Corner	OL-VC-80190	1.80	0.55	2011	2023
South Corner	OL-VC-80191	3.20	0.57	2017	2024
South Corner	OL-VC-80192	2.40	0.56	2013	2024
South Corner	OL-VC-80193	3.60	0.57	2018	2024
South Corner	OL-VC-80194	1.80	0.55	2011	2023
South Corner	OL-VC-80195	3.20	0.57	2017	2024
South Corner	OL-VC-80196	2.80	0.56	2016	2024
South Corner	OL-VC-80197	3.40	0.57	2018	2024
South Corner	OL-VC-80211	1.70	0.55	2011	2023
South Corner	OL-VC-80212	1.90	0.55	2011	2024
South Corner	OL-VC-80213	1.80	0.55	2011	2023
South Corner	OL-VC-80214	1.50	0.55	2011	2023
South Corner	OL-VC-80215	2.20	0.56	2011	2024
South Corner	OL-VC-80216	2.10	0.55	2011	2024
South Corner	OL-VC-80217	1.40	0.55	2011	2023
South Corner	OL-VC-80218	1.90	0.55	2011	2024
South Corner	OL-VC-80219	2.10	0.55	2011	2024
South Corner	OL-VC-80220	4.50	0.58	2018	2024
South Corner	OL-VC-80221	2.00	0.55	2011	2024
South Corner	OL-VC-80222	3.90	0.58	2018	2024
South Corner	OL-VC-80223	1.50	0.55	2011	2023

Notes:

[1] Initial mercury sediment concentration from 0 - 4 cm PDI data (includes LWA concentrations from 0 - 2 cm and 2 - 4 cm). Model mixed depth is 4 cm for all locations.

[2] Year model predicted concentrations reach the PEC or BSQV are rounded to the nearest whole year.

FIGURES

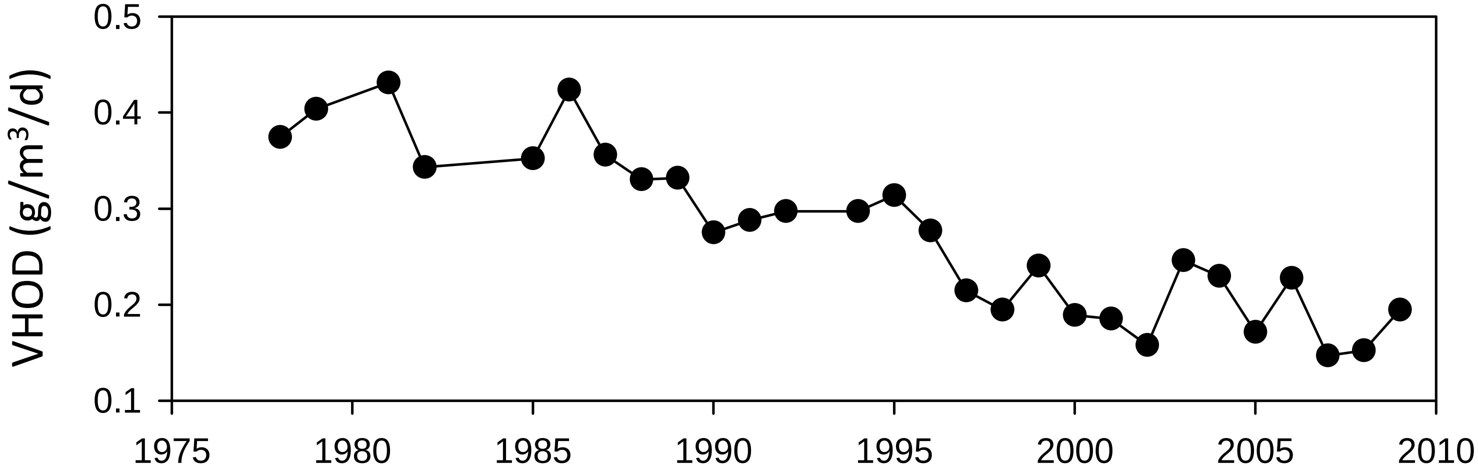


Figure M.1 Time series of the volumetric hypolimnetic oxygen deficit (VHOD) for Onondaga Lake over the 1978 – 2009 interval.

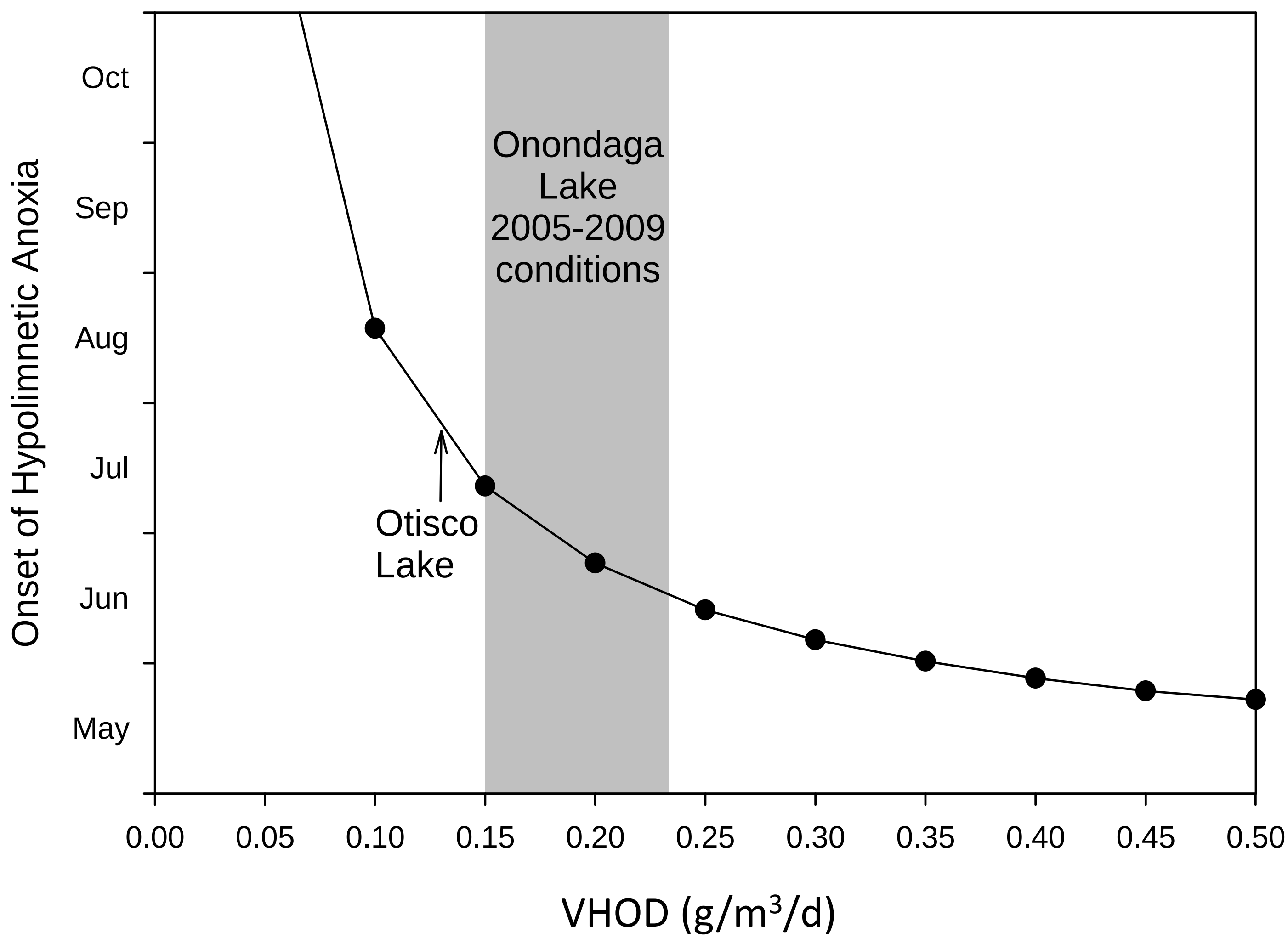


Figure M.2 Evaluation of the relationship between the timing of the onset of complete hypolimnetic anoxia and the volumetric hypolimnetic oxygen deficit (VHOD) for Onondaga Lake. Average VHOD conditions observed in Otisco Lake during 2002 – 2004 (Denkenberger et al. 2007) are presented for reference.

- No mixing in sediment from deep water areas based on tight layering observed during 2010 in North and South Basin cores (see horizontal lines in photos)

North Basin (S90 and QL-STA-80068)

Ninemile Creek Outlet Area
(OL-MB-80096)

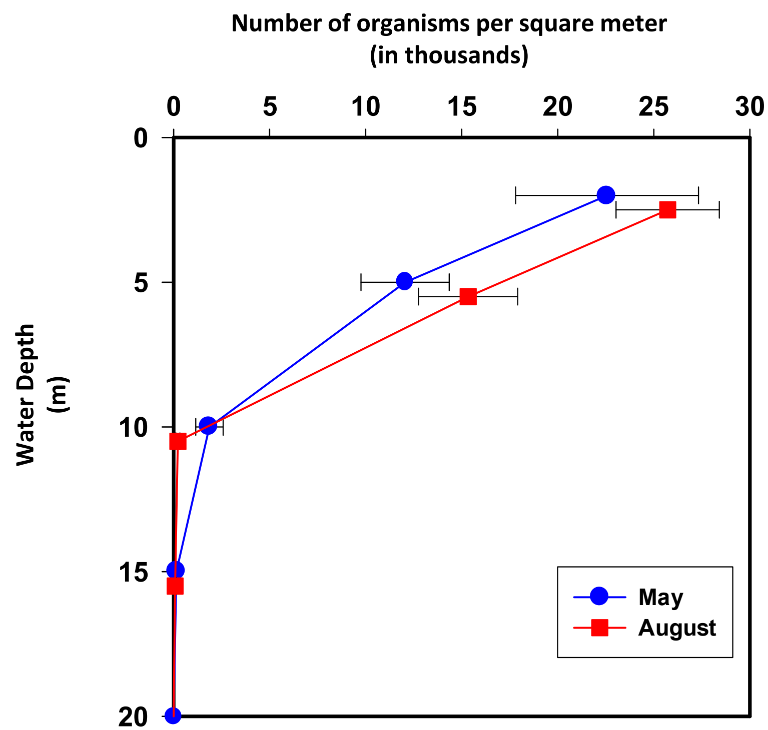
- Sediment layering also noted in top 18 cm of S90 core as "black, occasional subtle 1 cm bands" (from Rowell and Effler, 1996)

Figure M.3a Layering/Laminations within SMU 8 Sediment Cores

South Basin (OL-VC-80168)

South Basin (S51)

Figure M.3b Layering/Laminations within SMU 8 Sediment Cores



Source : Hairston and Effler, 1998.

Figure M.4 Macroinvertebrates Observed in Onondaga Lake as a Function of Water Depth (1998)

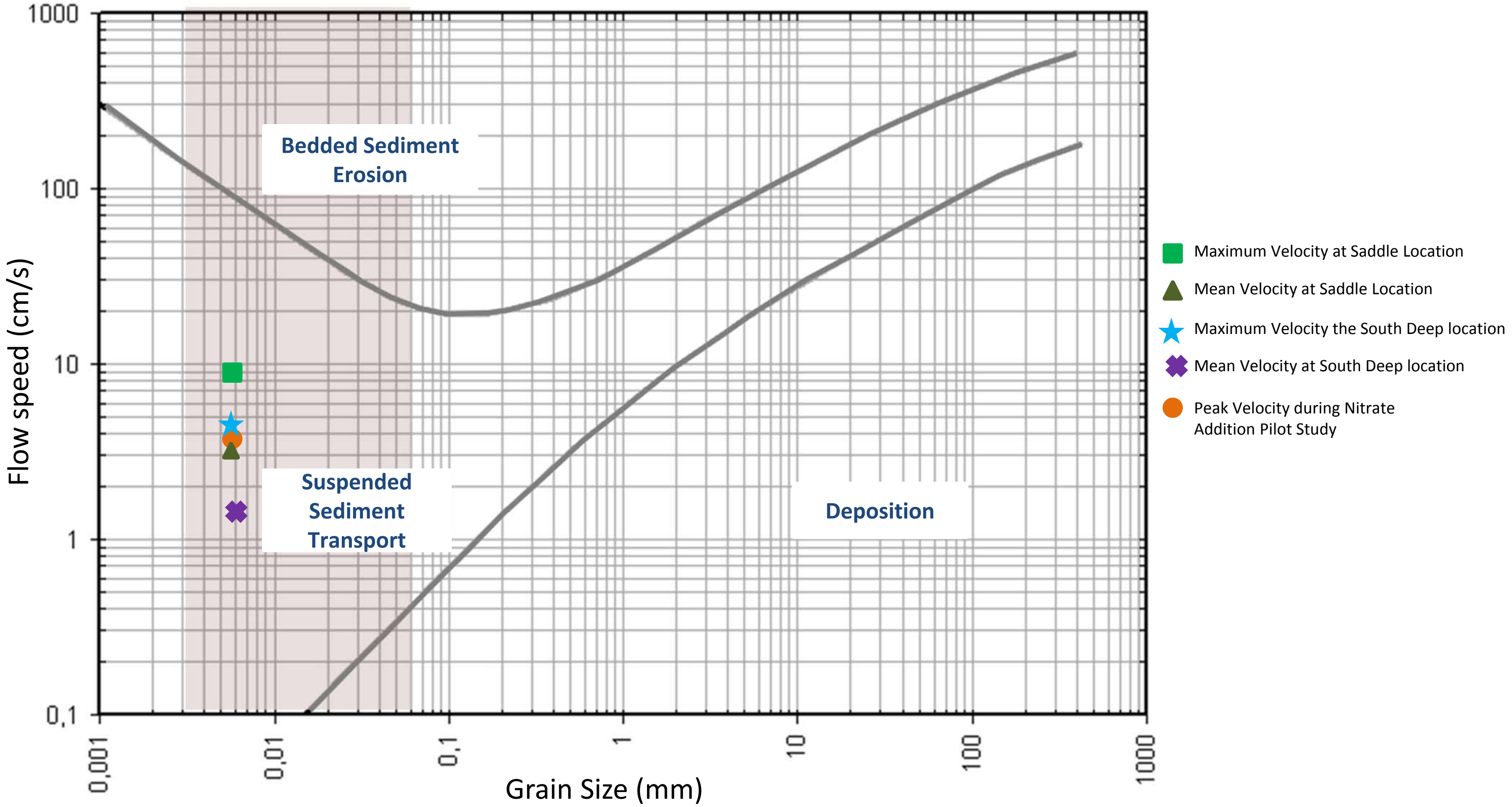
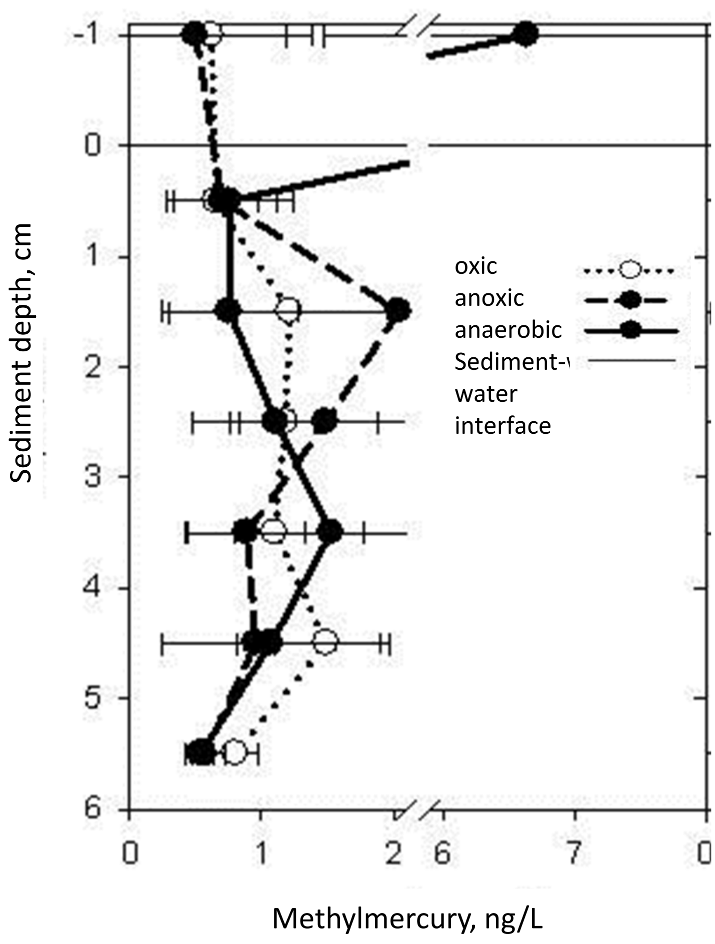


Figure M.5 Hjulstrom Diagram

Area shaded pink is the typical size range for silt.



Depth profiles for methylmercury under oxic, anoxic, and anaerobic conditions. *(From Figure 3.6, Draft Sediment Incubation Data Report, Exponent et al 2009)*

Figure M.6 Basis for 4 cm BSQV Compliance Depth in SMU 8

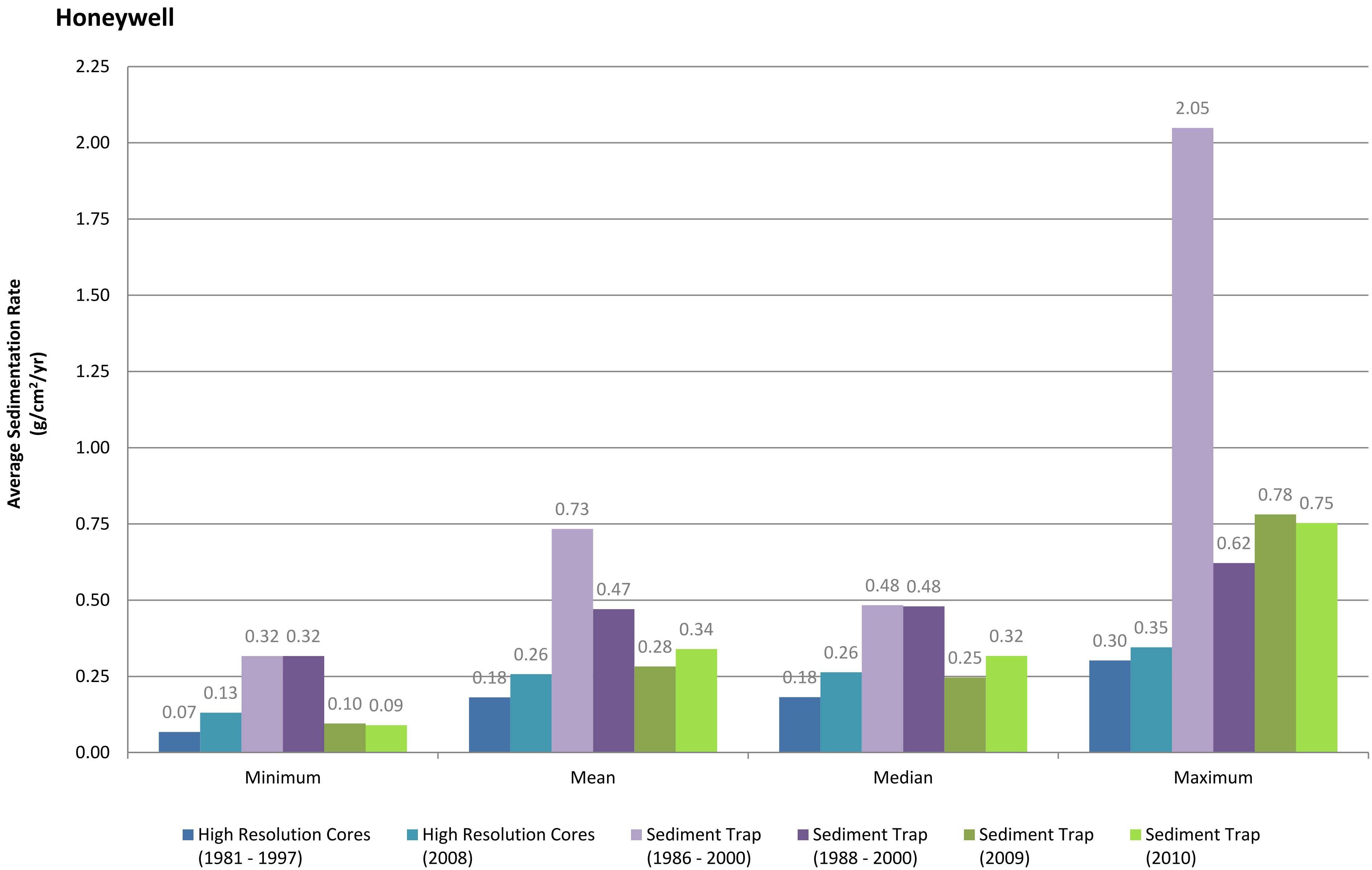


Figure M.7 Comparison of average sedimentation rates from various collection methods and years.

High resolution cores (1981-1997) converted to g/cm^2/yr assuming a dry bulk density equal to 0.243 g/cc based on a porosity of 0.91 and a specific gravity of 2.7 g/cc. High resolution cores (2008) include surface intervals represented by the top 0-4 cm. Sediment trap data have been extrapolated to annual rates; they do not consider sedimentation rates during the winter months, which may be lower.

Honeywell

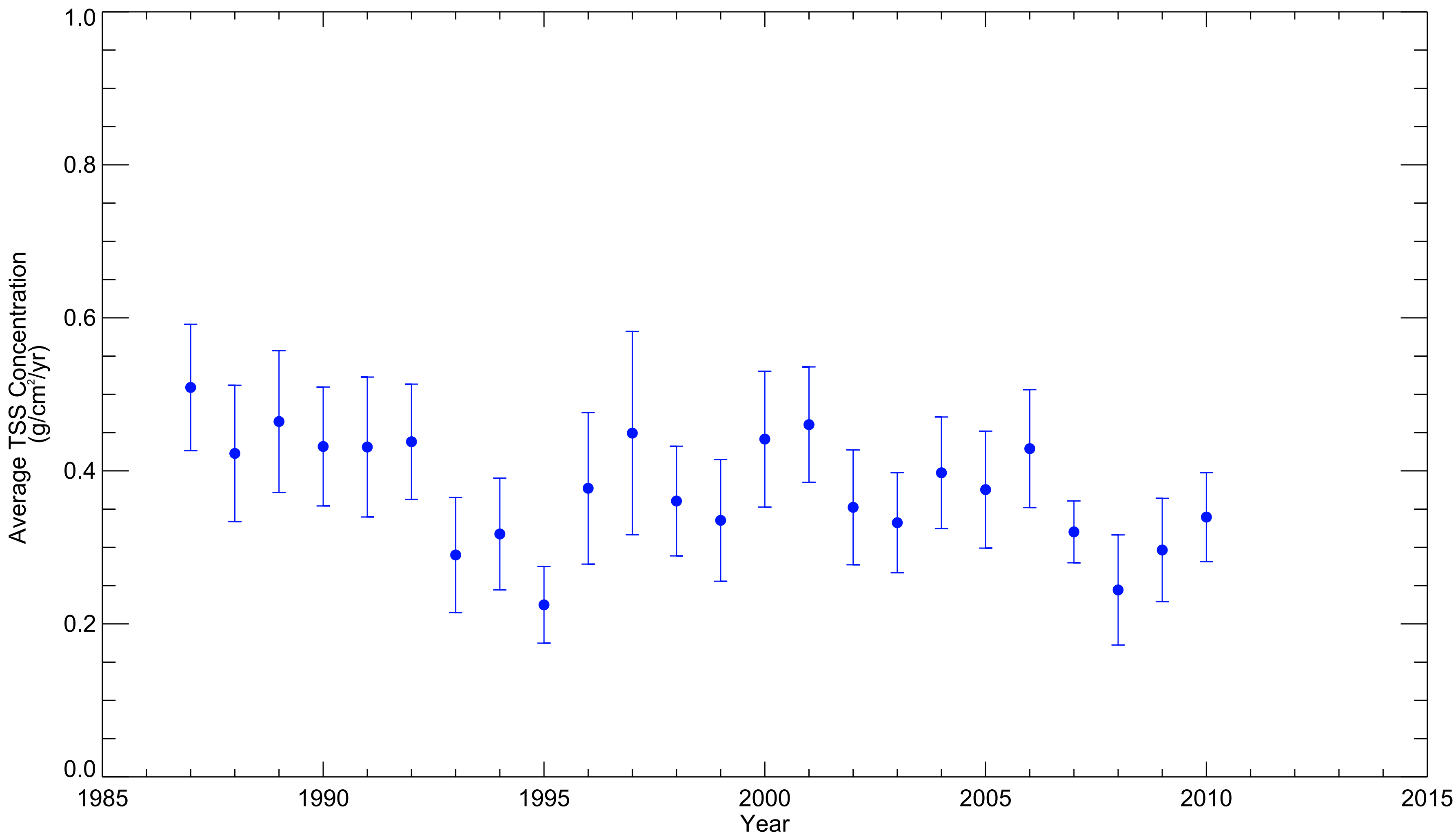


Figure M.8 Average Annual Sedimentation Rates From Sediment Trap Data Collected Between 1987 and 2010

Note: These plots are based on weekly sediment trap data collected by UFI from April through October of most years (1993 collection from May through September). Plots show the average +/- 2 standard error (SE) of the mean, which is one way of representing the variability in the weekly values obtained for the year noted. Although shown as annual averages, the sediment traps do not consider sedimentation rates during the winter months, which may be lower.

Honeywell

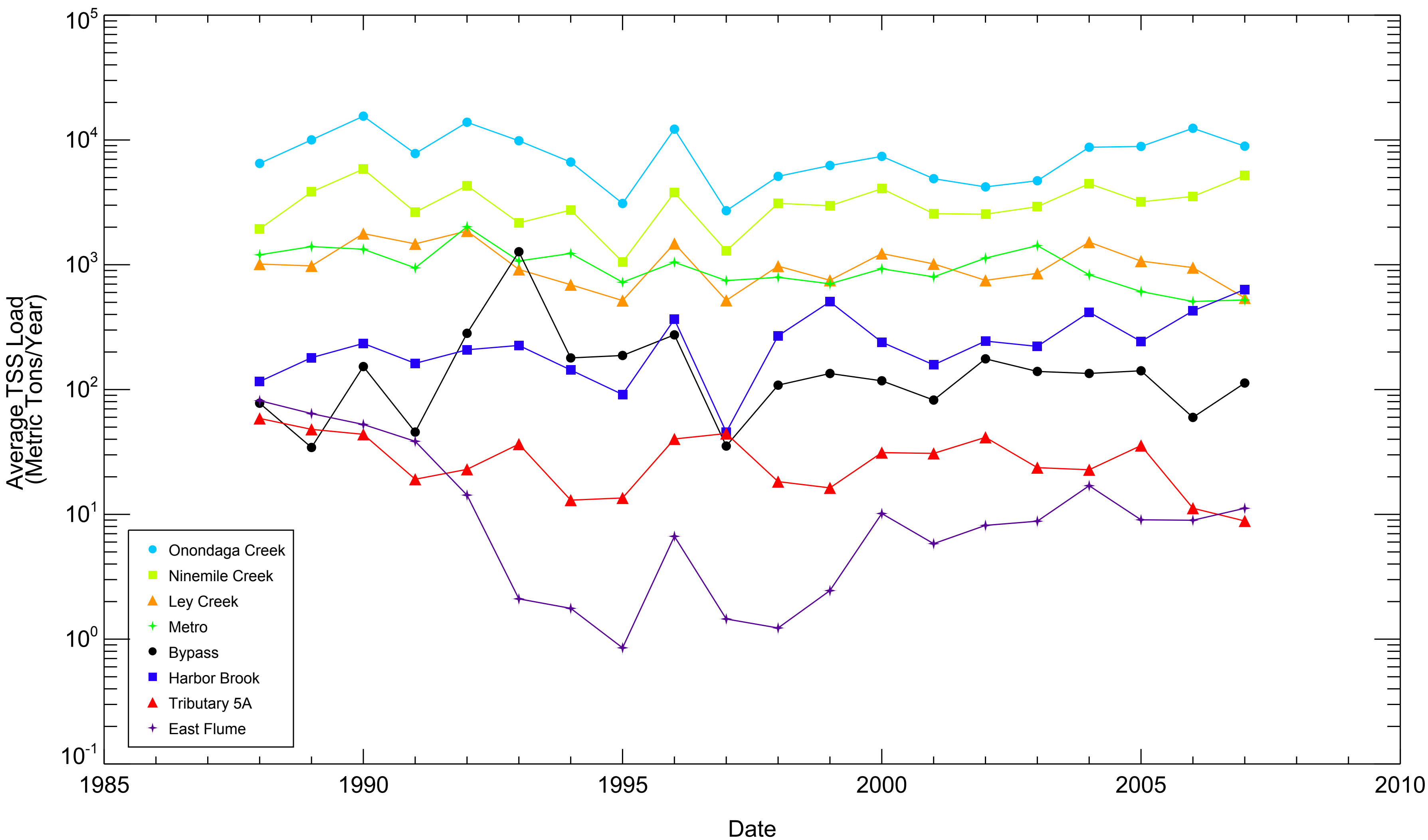


Figure M.9 Temporal Trends in TSS Loads to Onondaga Lake
Source: Onondaga Lake Ambient Monitoring Program (AMP)

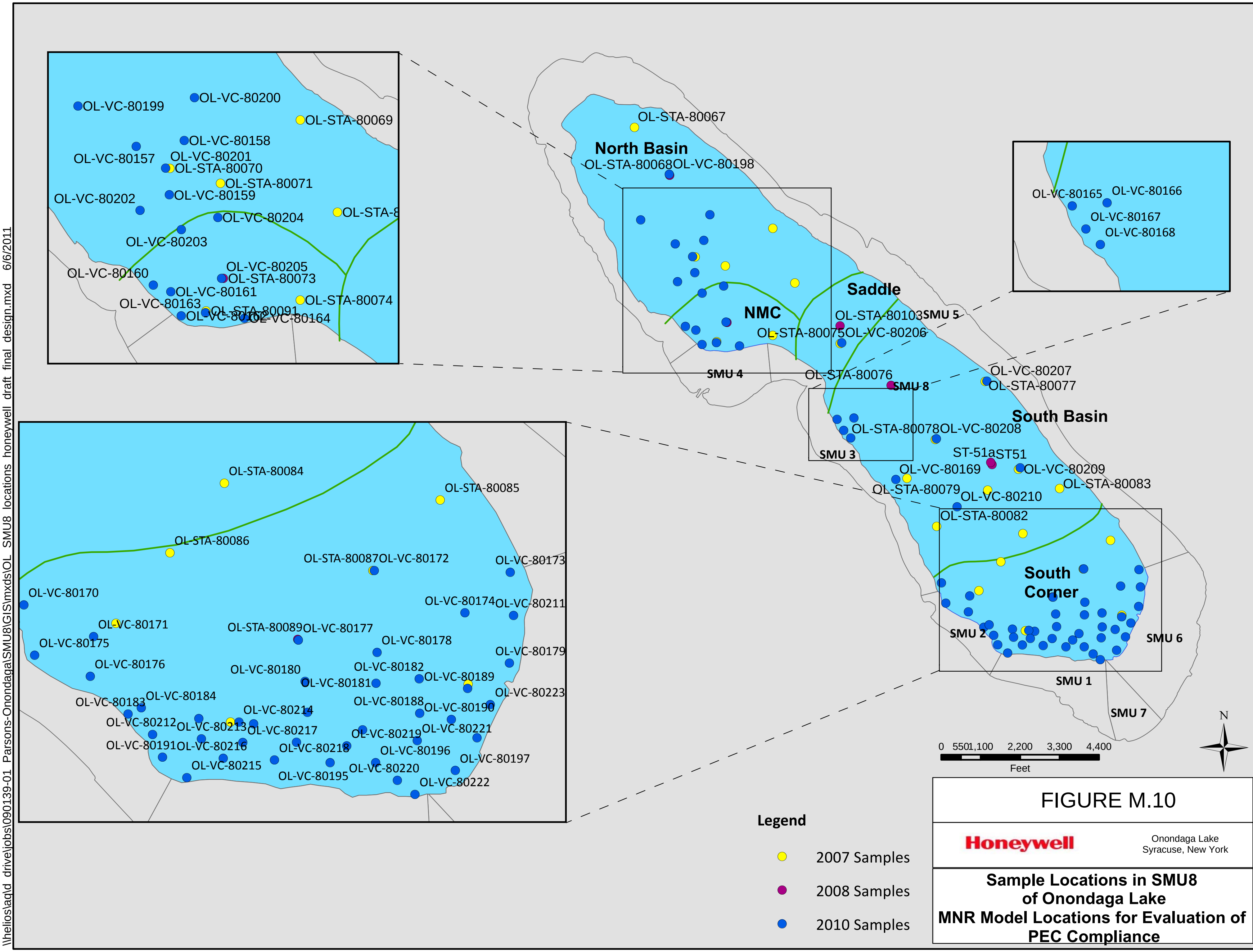


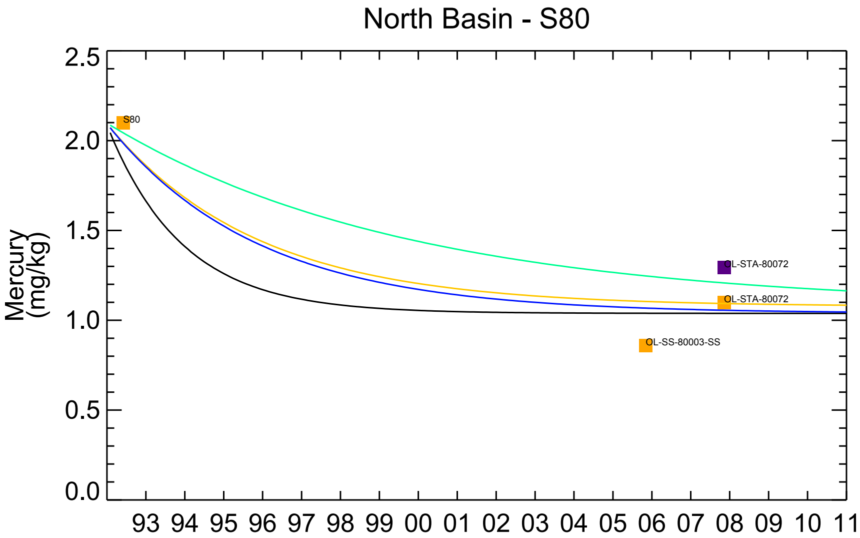
FIGURE M.10

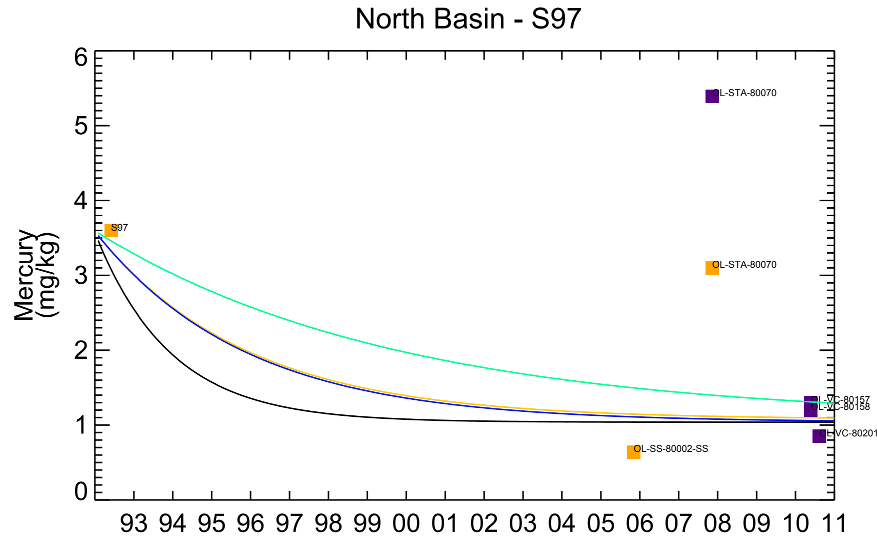
Honeywell

Onondaga Lake
Syracuse, New York

Sample Locations in SMU8 of Onondaga Lake MNR Model Locations for Evaluation of PEC Compliance

Honeywell





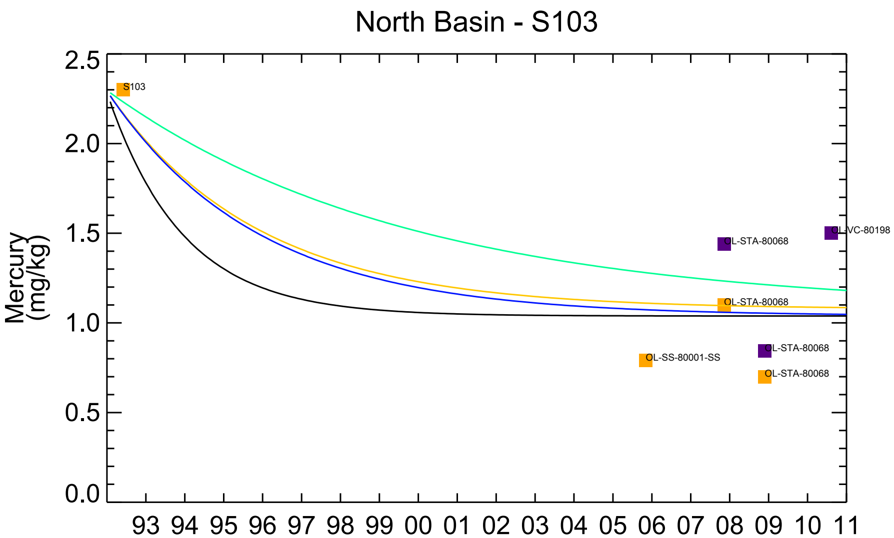


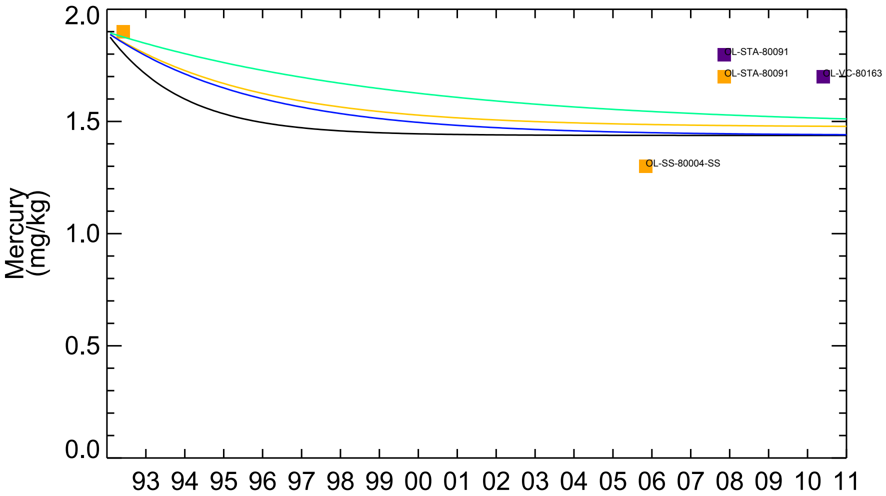
Figure M.11. MNR Model Sensitivity: Sedimentation Rate and Mixing Depth Sensitivity

Data: 2 cm
Data: 4 cm

Settling_Rate = 0.25 g/cm²/yr (2 cm mixing) g/cm²/yr
Settling_Rate = 0.125 g/cm²/yr (2 cm mixing) g/cm²/yr
Settling_Rate = 0.25 g/cm²/yr (4 cm mixing) g/cm²/yr
Settling_Rate = 0.125 g/cm²/yr (4 cm mixing) g/cm²/yr

Honeywell

Nine Mile Creek - S85



Nine Mile Creek - S78

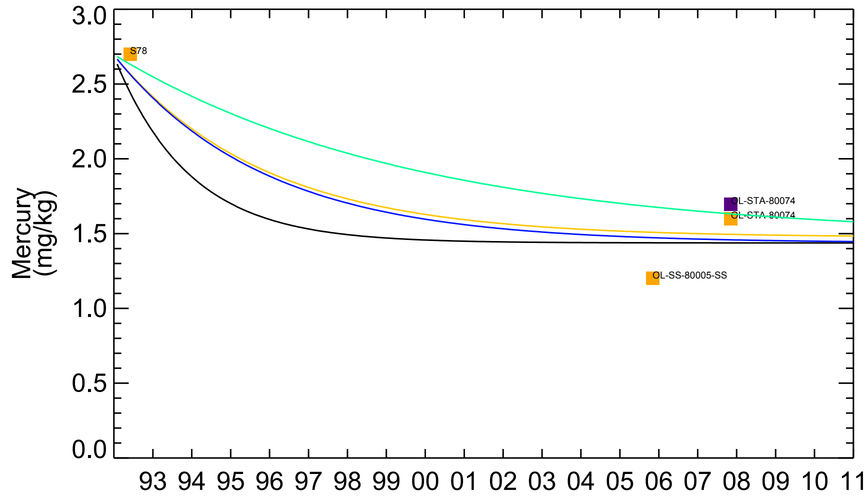


Figure M.11. MNR Model Sensitivity: Sedimentation Rate and Mixing Depth Sensitivity

Data: 2 cm
Data: 4 cm

Settling_Rate = 0.25 g/cm^2/yr (2 cm mixing) g/cm^2/yr
Settling_Rate = 0.125 g/cm^2/yr (2 cm mixing) g/cm^2/yr
Settling_Rate = 0.25 g/cm^2/yr (4 cm mixing) g/cm^2/yr
Settling_Rate = 0.125 g/cm^2/yr (4 cm mixing) g/cm^2/yr

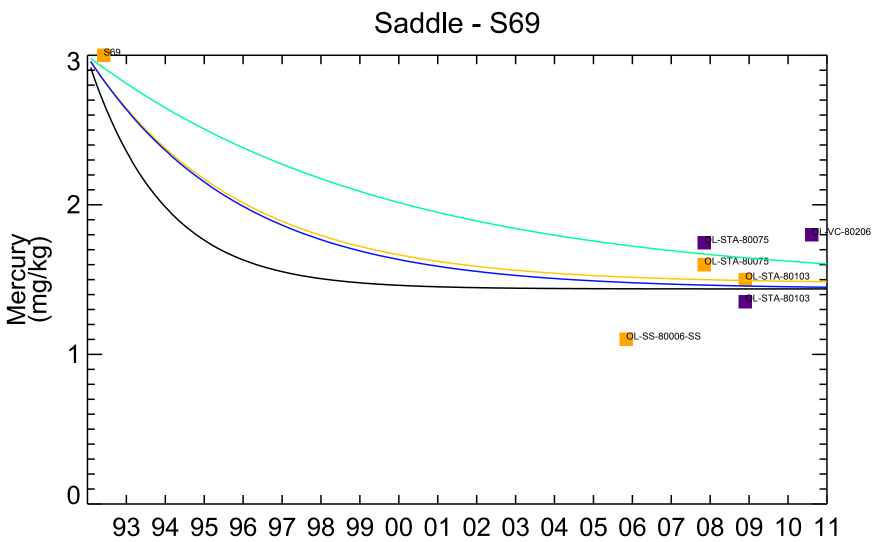


Figure M.11. MNR Model Sensitivity: Sedimentation Rate and Mixing Depth Sensitivity

Data: 2 cm
Data: 4 cm

Settling_Rate = 0.25 g/cm²/yr (2 cm mixing) g/cm²/yr
Settling_Rate = 0.125 g/cm²/yr (2 cm mixing) g/cm²/yr
Settling_Rate = 0.25 g/cm²/yr (4 cm mixing) g/cm²/yr
Settling_Rate = 0.125 g/cm²/yr (4 cm mixing) g/cm²/yr

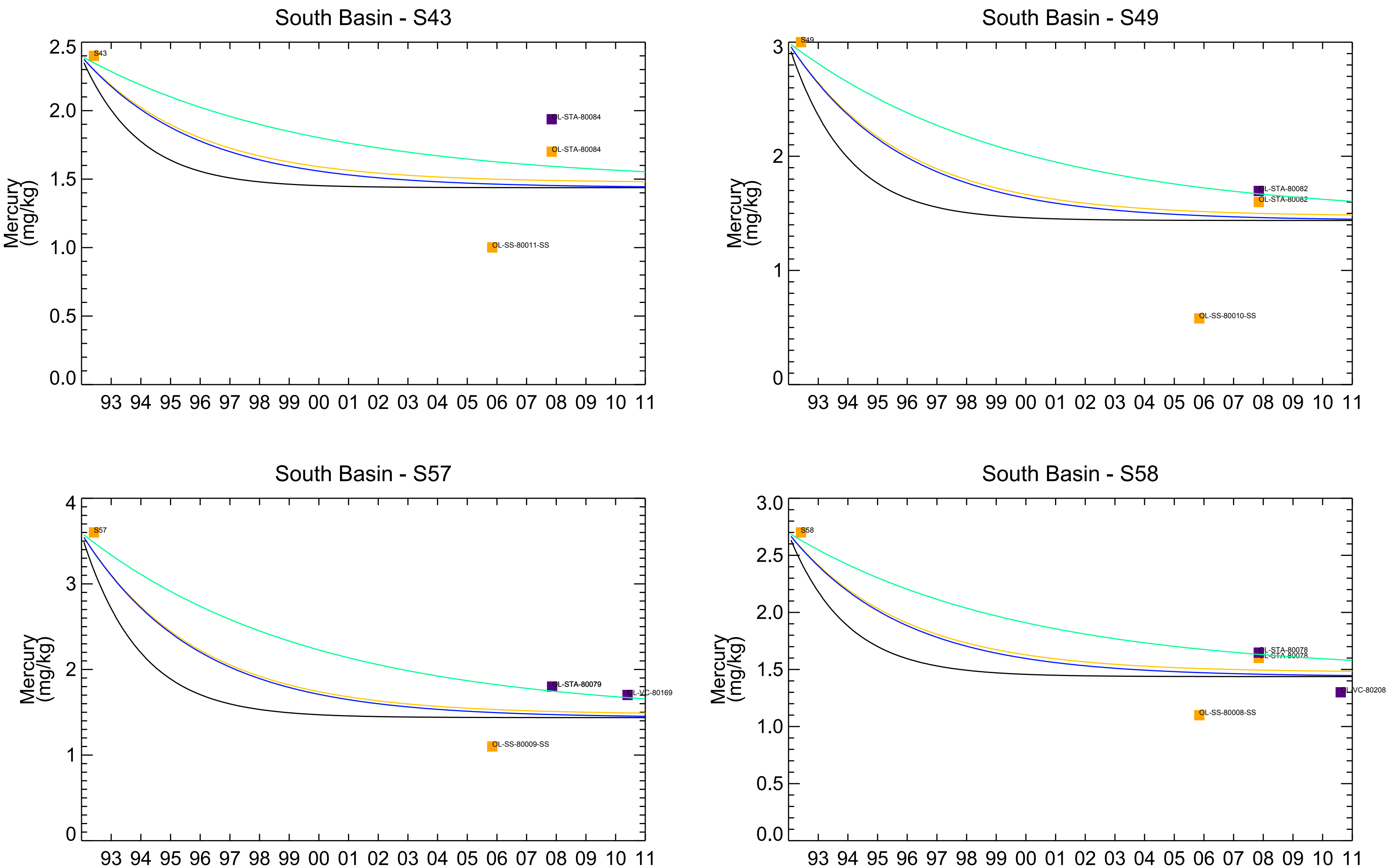


Figure M.11. MNR Model Sensitivity: Sedimentation Rate and Mixing Depth Sensitivity

Data: 2 cm
Data: 4 cm

Settling_Rate = 0.25 g/cm²/yr (2 cm mixing) g/cm²/yr
Settling_Rate = 0.125 g/cm²/yr (2 cm mixing) g/cm²/yr
Settling_Rate = 0.25 g/cm²/yr (4 cm mixing) g/cm²/yr
Settling_Rate = 0.125 g/cm²/yr (4 cm mixing) g/cm²/yr

Honeywell

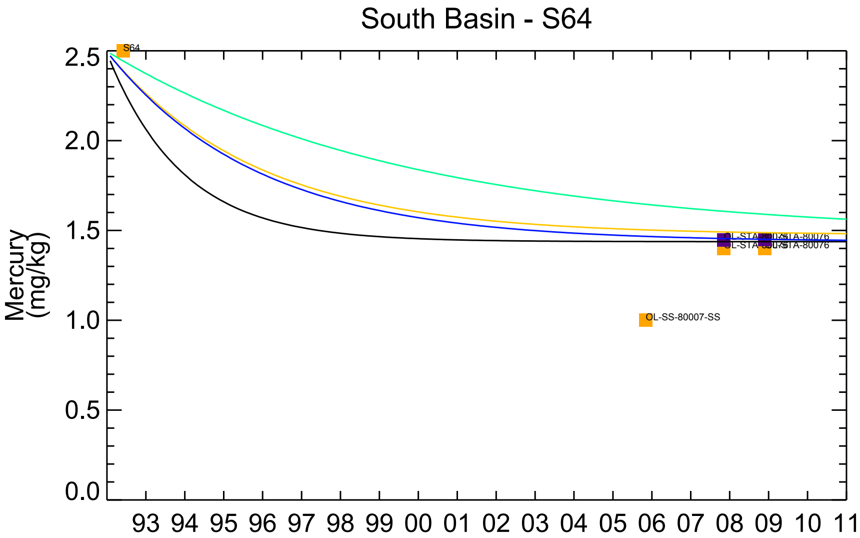


Figure M.11. MNR Model Sensitivity: Sedimentation Rate and Mixing Depth Sensitivity

Data: 2 cm
Data: 4 cm

Settling_Rate = 0.25 g/cm²/yr (2 cm mixing) g/cm²/yr
Settling_Rate = 0.125 g/cm²/yr (2 cm mixing) g/cm²/yr
Settling_Rate = 0.25 g/cm²/yr (4 cm mixing) g/cm²/yr
Settling_Rate = 0.125 g/cm²/yr (4 cm mixing) g/cm²/yr

Honeywell

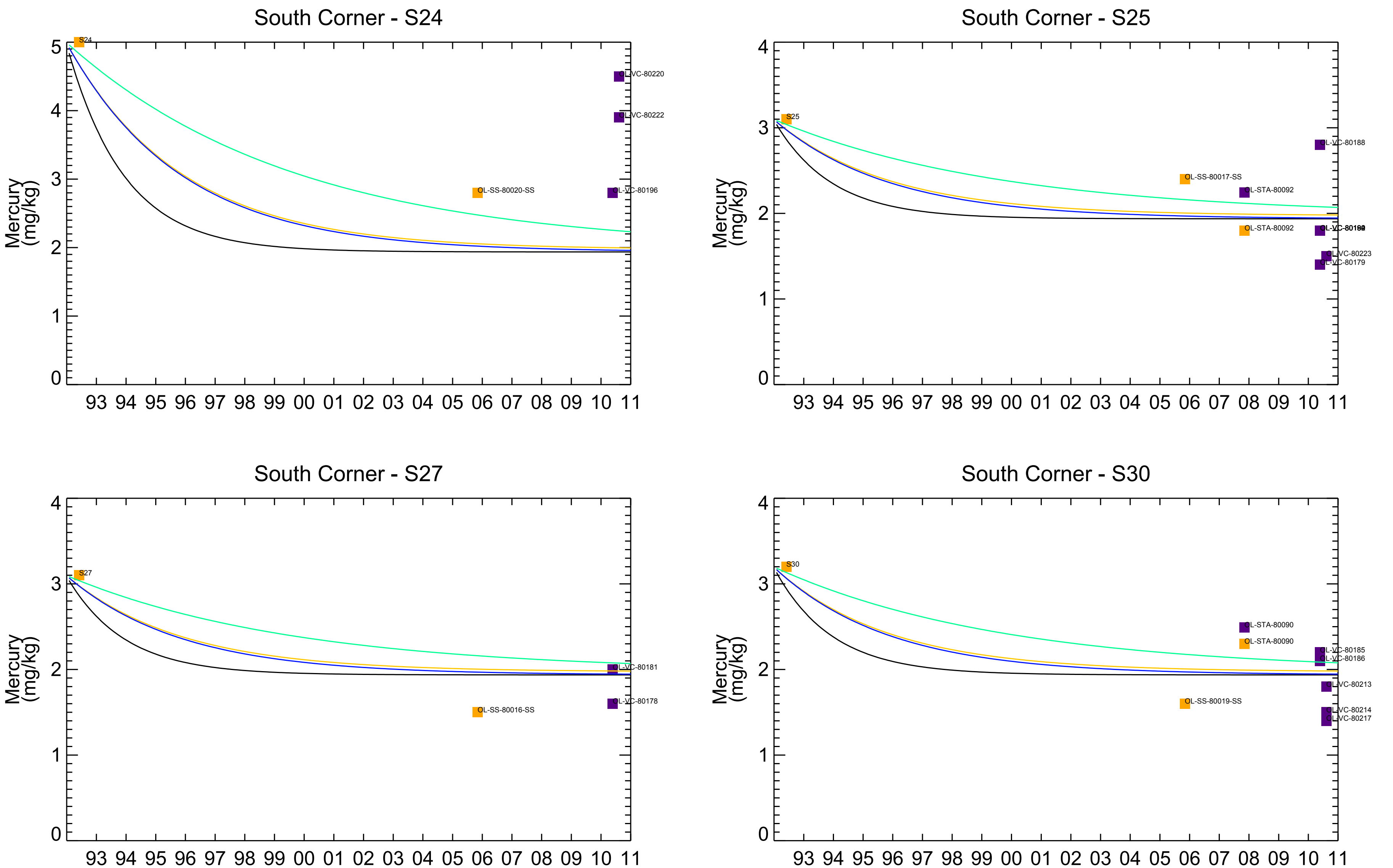


Figure M.11. MNR Model Sensitivity: Sedimentation Rate and Mixing Depth Sensitivity

Data: 2 cm
Data: 4 cm
Settling_Rate = 0.25 g/cm²/yr (2 cm mixing) g/cm²/yr
Settling_Rate = 0.125 g/cm²/yr (2 cm mixing) g/cm²/yr
Settling_Rate = 0.25 g/cm²/yr (4 cm mixing) g/cm²/yr
Settling_Rate = 0.125 g/cm²/yr (4 cm mixing) g/cm²/yr

Honeywell

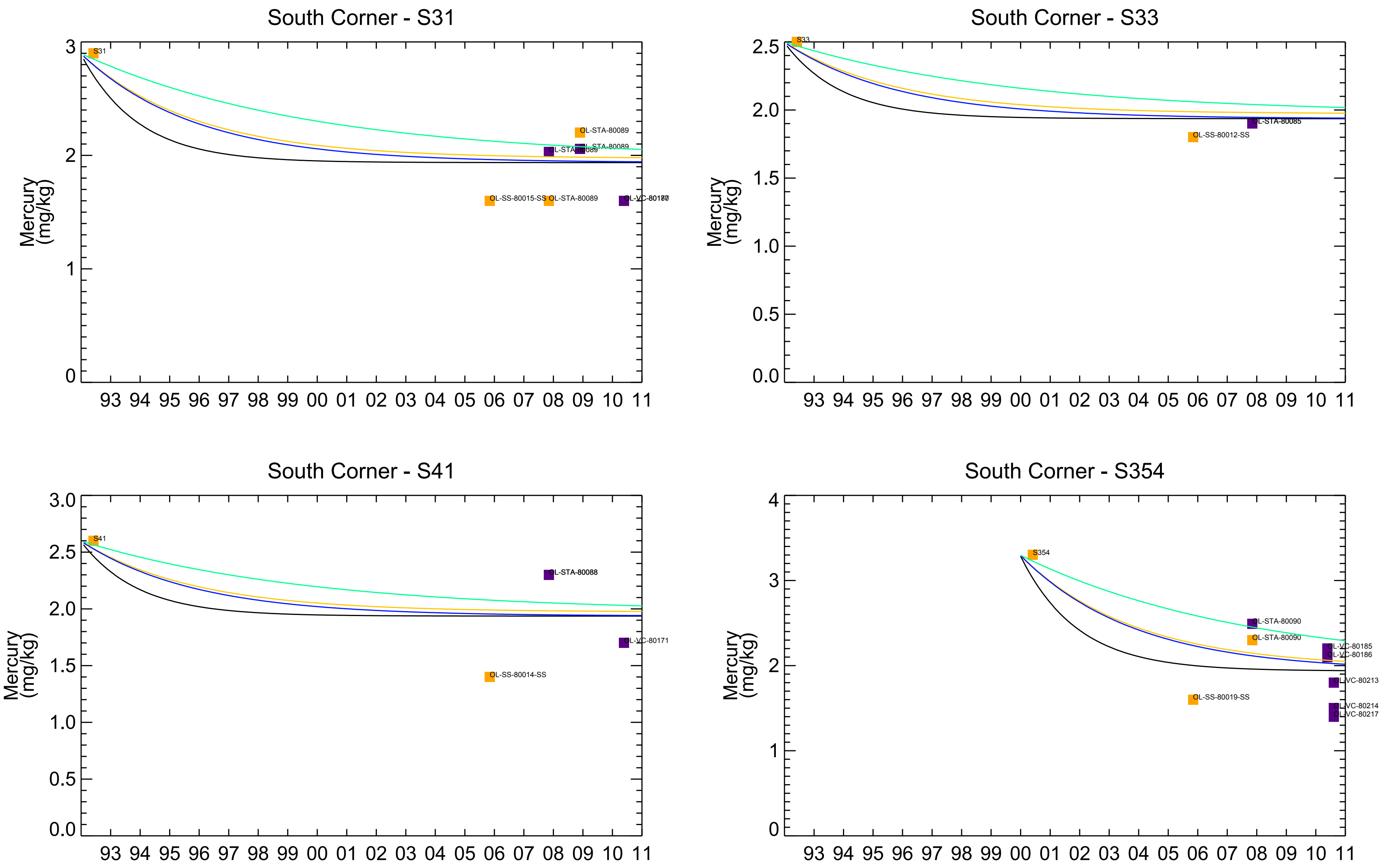


Figure M.11. MNR Model Sensitivity: Sedimentation Rate and Mixing Depth Sensitivity

Data: 2 cm
Data: 4 cm

Settling_Rate = 0.25 g/cm²/yr (2 cm mixing) g/cm²/yr
Settling_Rate = 0.125 g/cm²/yr (2 cm mixing) g/cm²/yr
Settling_Rate = 0.25 g/cm²/yr (4 cm mixing) g/cm²/yr
Settling_Rate = 0.125 g/cm²/yr (4 cm mixing) g/cm²/yr

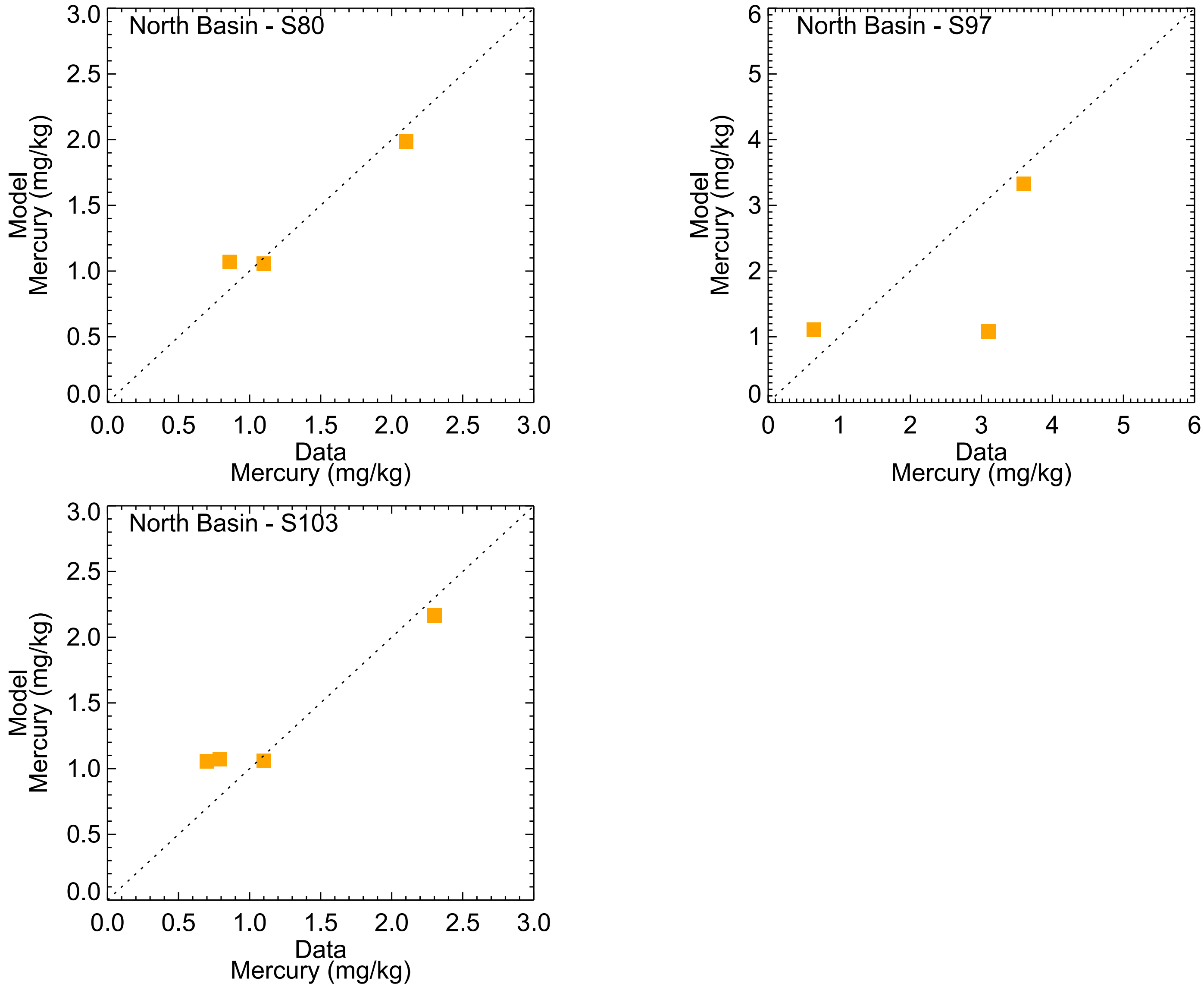


Figure M.12 MNR Model Calibration: Model Predictions Versus Data
SMU 8 Onondaga Lake

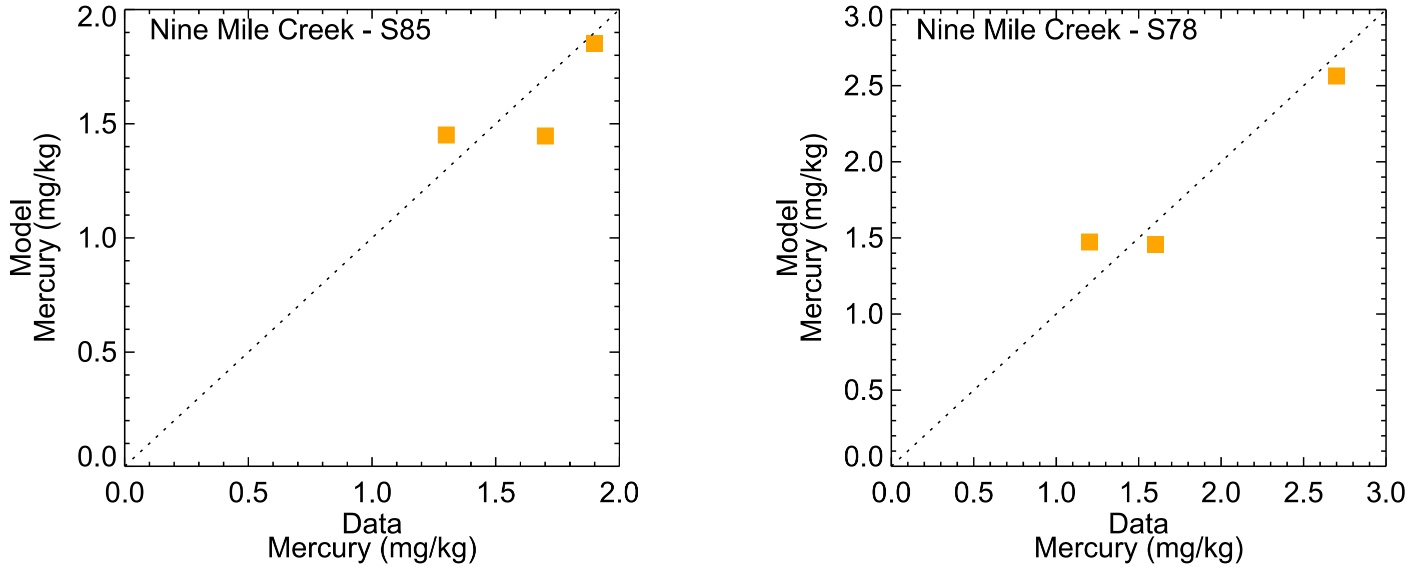


Figure M.12 MNR Model Calibration: Model Predictions Versus Data
SMU 8 Onondaga Lake

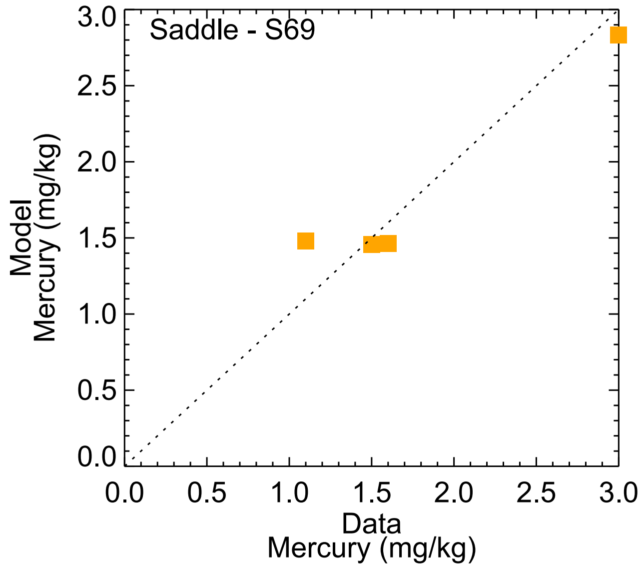


Figure M.12 MNR Model Calibration: Model Predictions Versus Data
SMU 8 Onondaga Lake

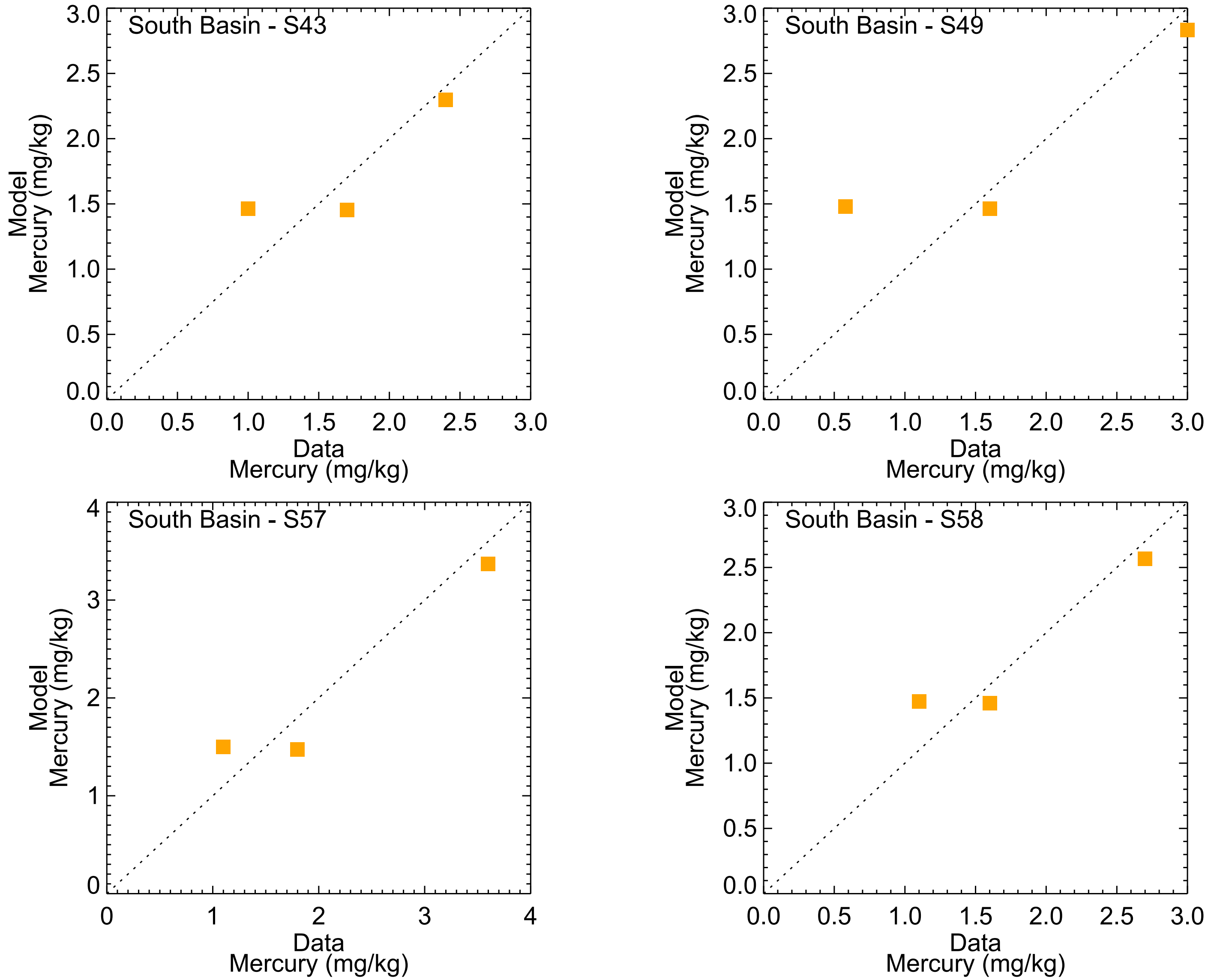


Figure M.12 MNR Model Calibration: Model Predictions Versus Data
SMU 8 Onondaga Lake

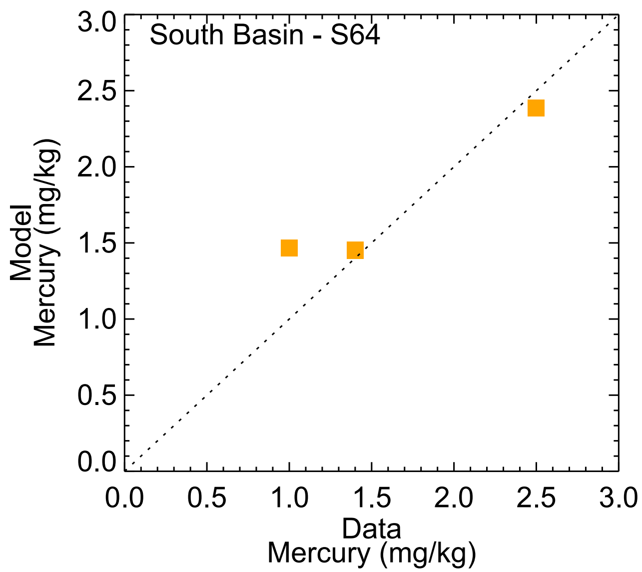


Figure M.12 MNR Model Calibration: Model Predictions Versus Data
SMU 8 Onondaga Lake

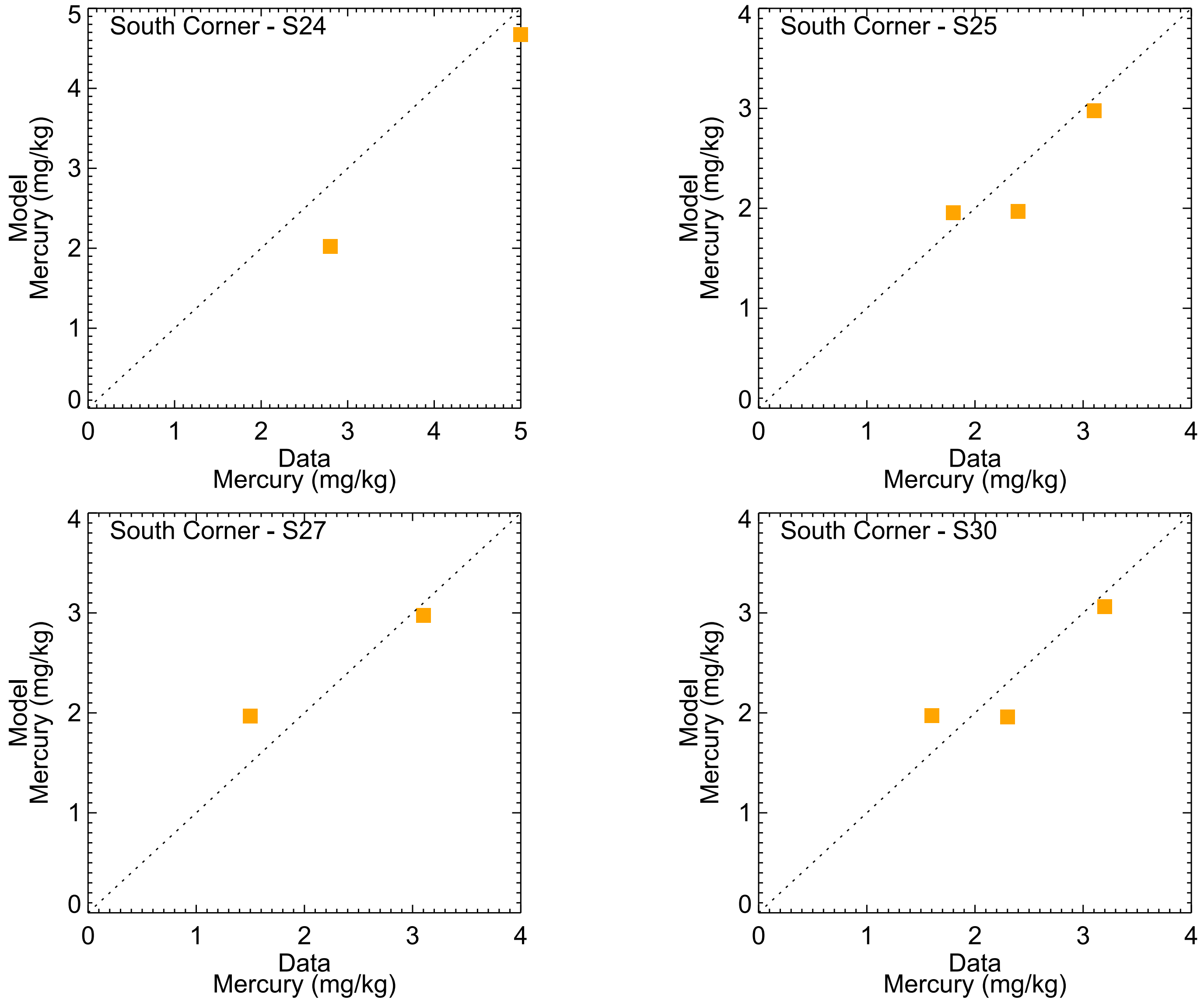


Figure M.12 MNR Model Calibration: Model Predictions Versus Data
SMU 8 Onondaga Lake

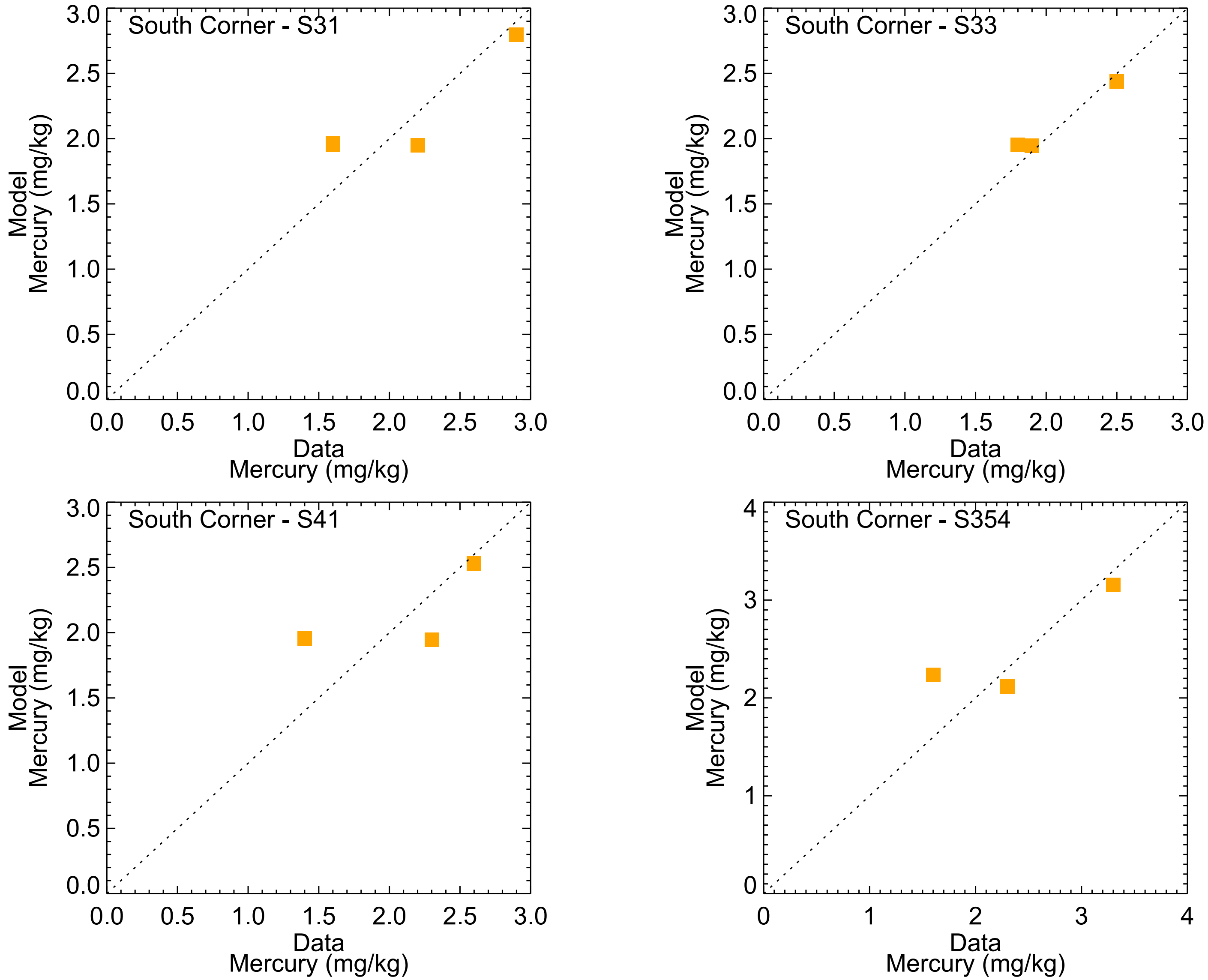


Figure M.12 MNR Model Calibration: Model Predictions Versus Data
SMU 8 Onondaga Lake

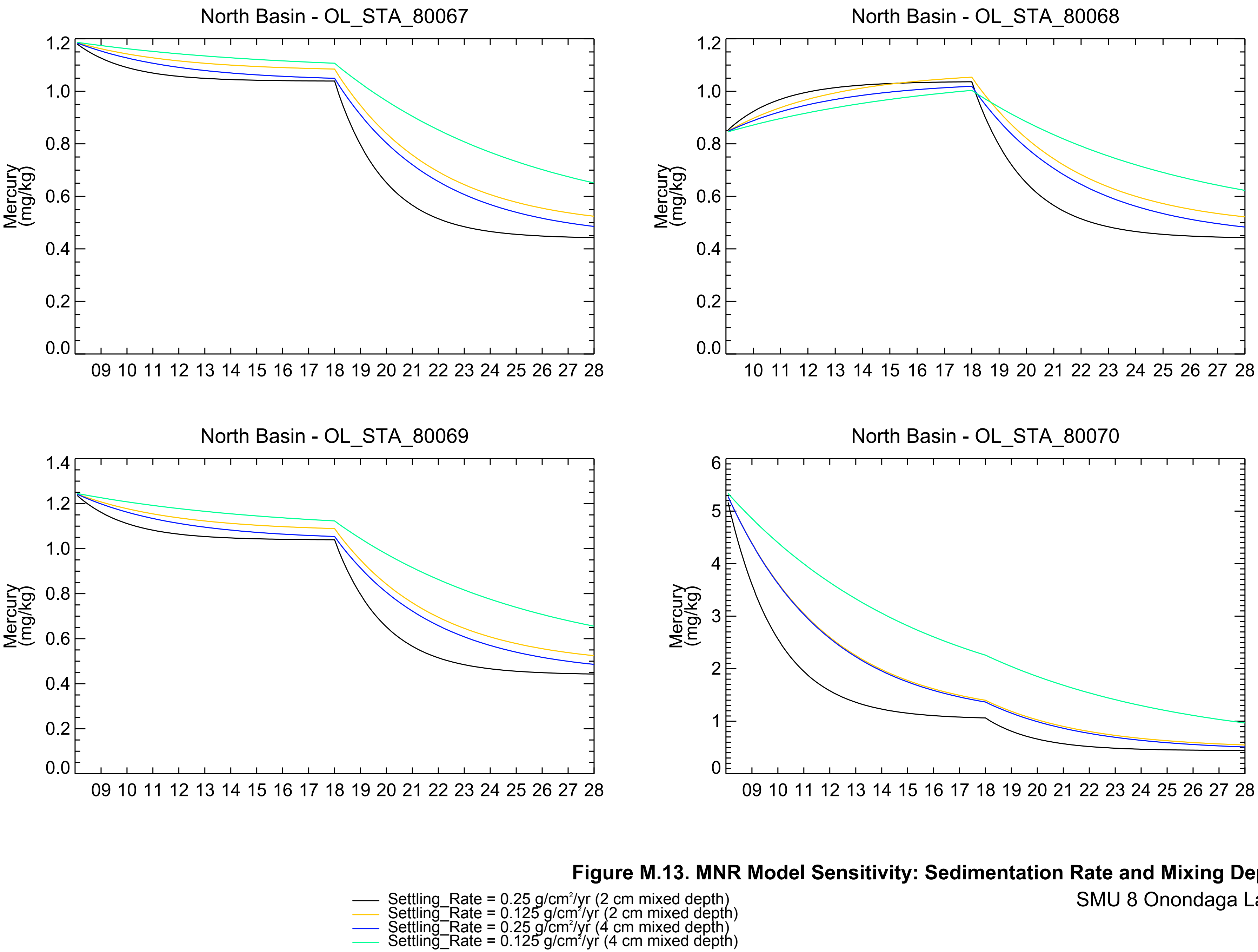


Figure M.13. MNR Model Sensitivity: Sedimentation Rate and Mixing Depth

SMU 8 Onondaga Lake

Honeywell

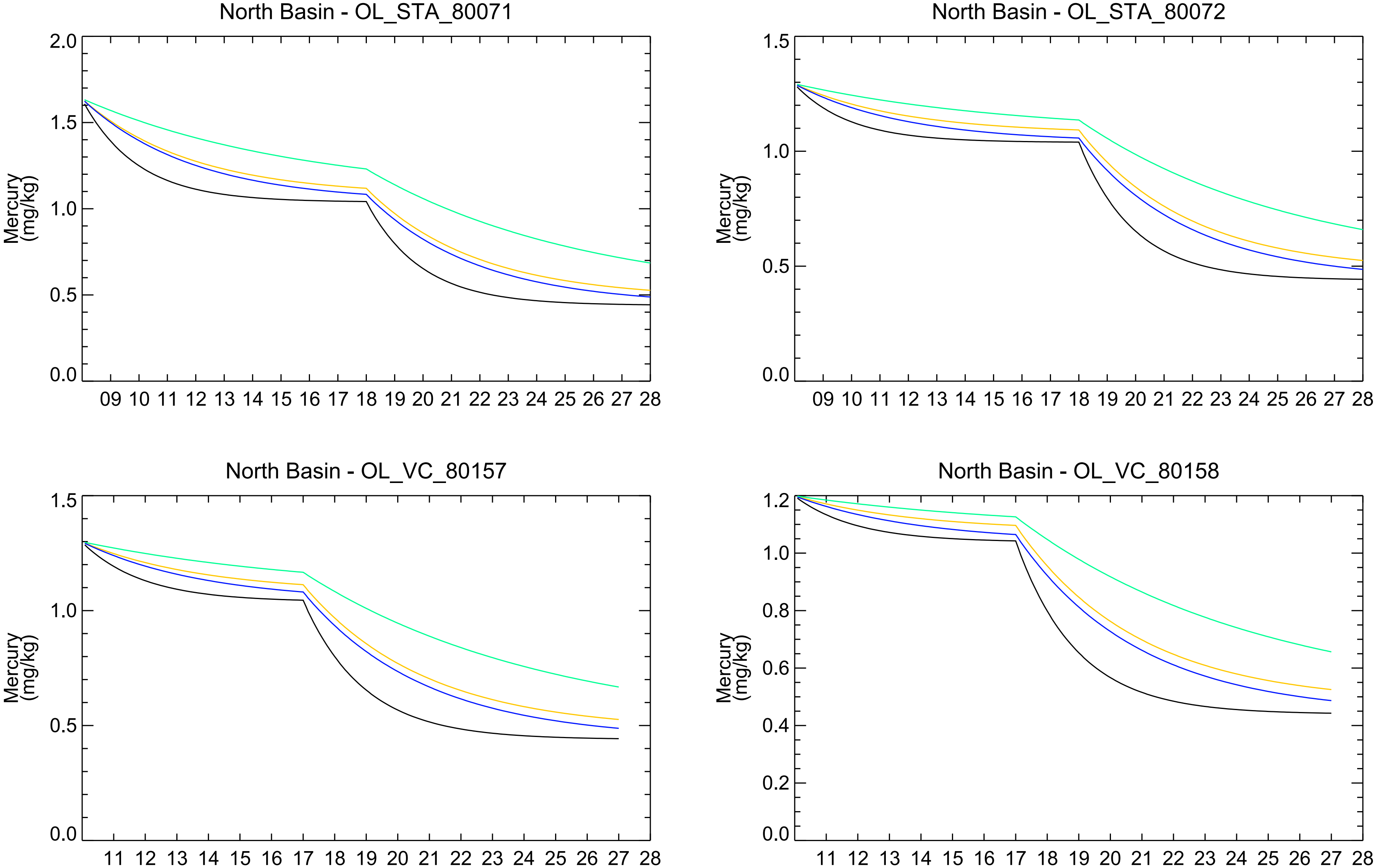
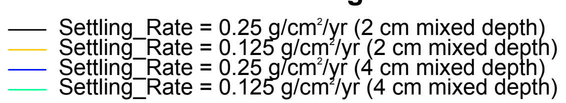


Figure M.13. MNR Model Sensitivity: Sedimentation Rate and Mixing Depth

SMU 8 Onondaga Lake



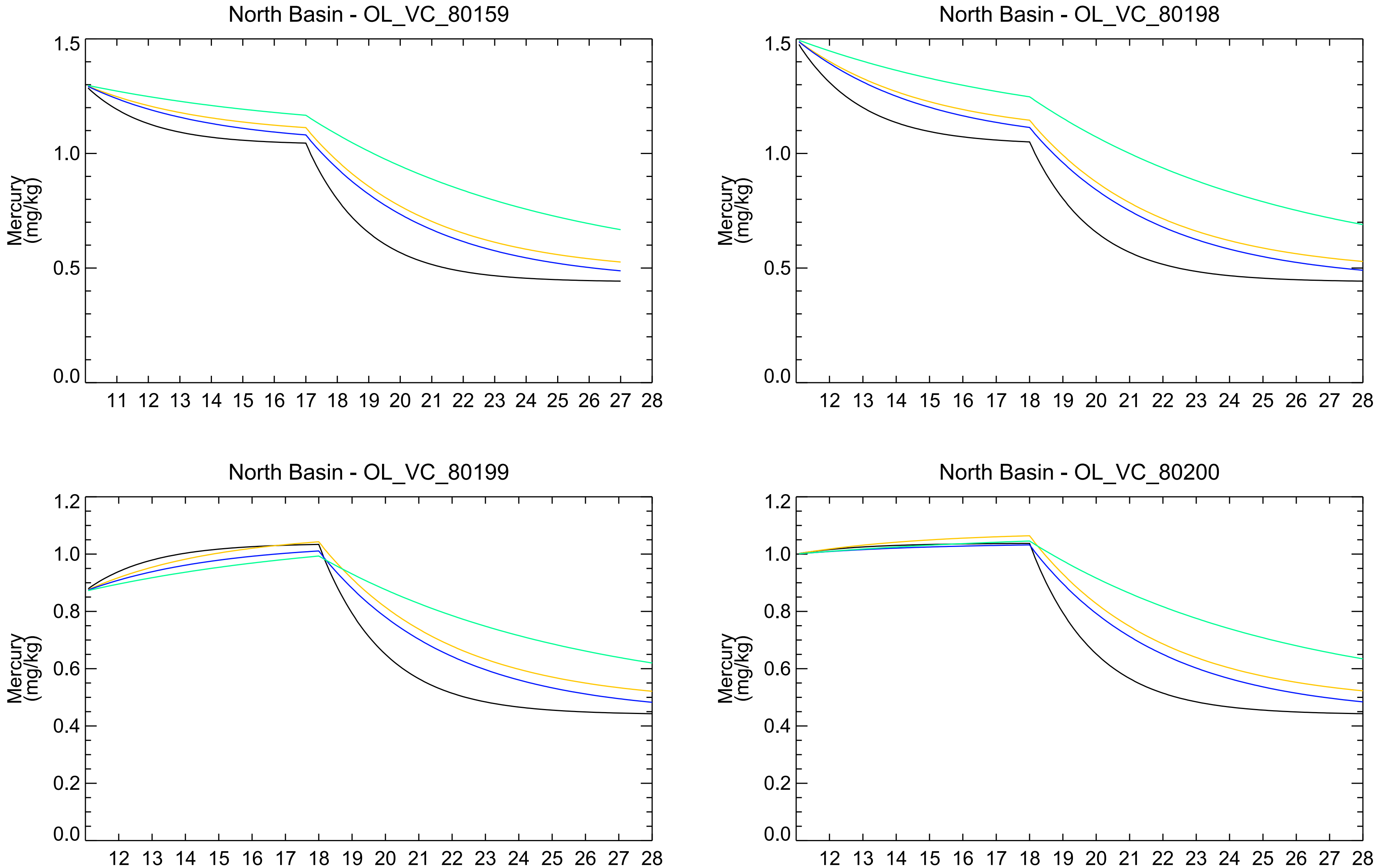
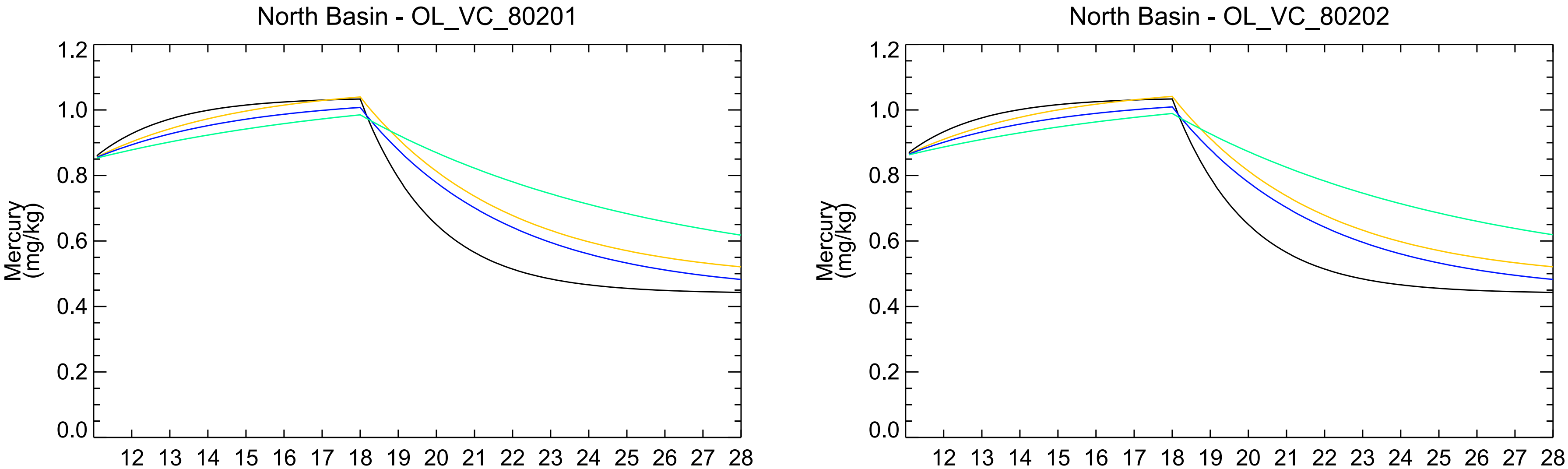


Figure M.13. MNR Model Sensitivity: Sedimentation Rate and Mixing Depth

SMU 8 Onondaga Lake

Honeywell



Settling_Rate = 0.25 g/cm²/yr (2 cm mixed depth)
Settling_Rate = 0.125 g/cm²/yr (2 cm mixed depth)
Settling_Rate = 0.25 g/cm²/yr (4 cm mixed depth)
Settling_Rate = 0.125 g/cm²/yr (4 cm mixed depth)

Figure M.13. MNR Model Sensitivity: Sedimentation Rate and Mixing Depth

SMU 8 Onondaga Lake

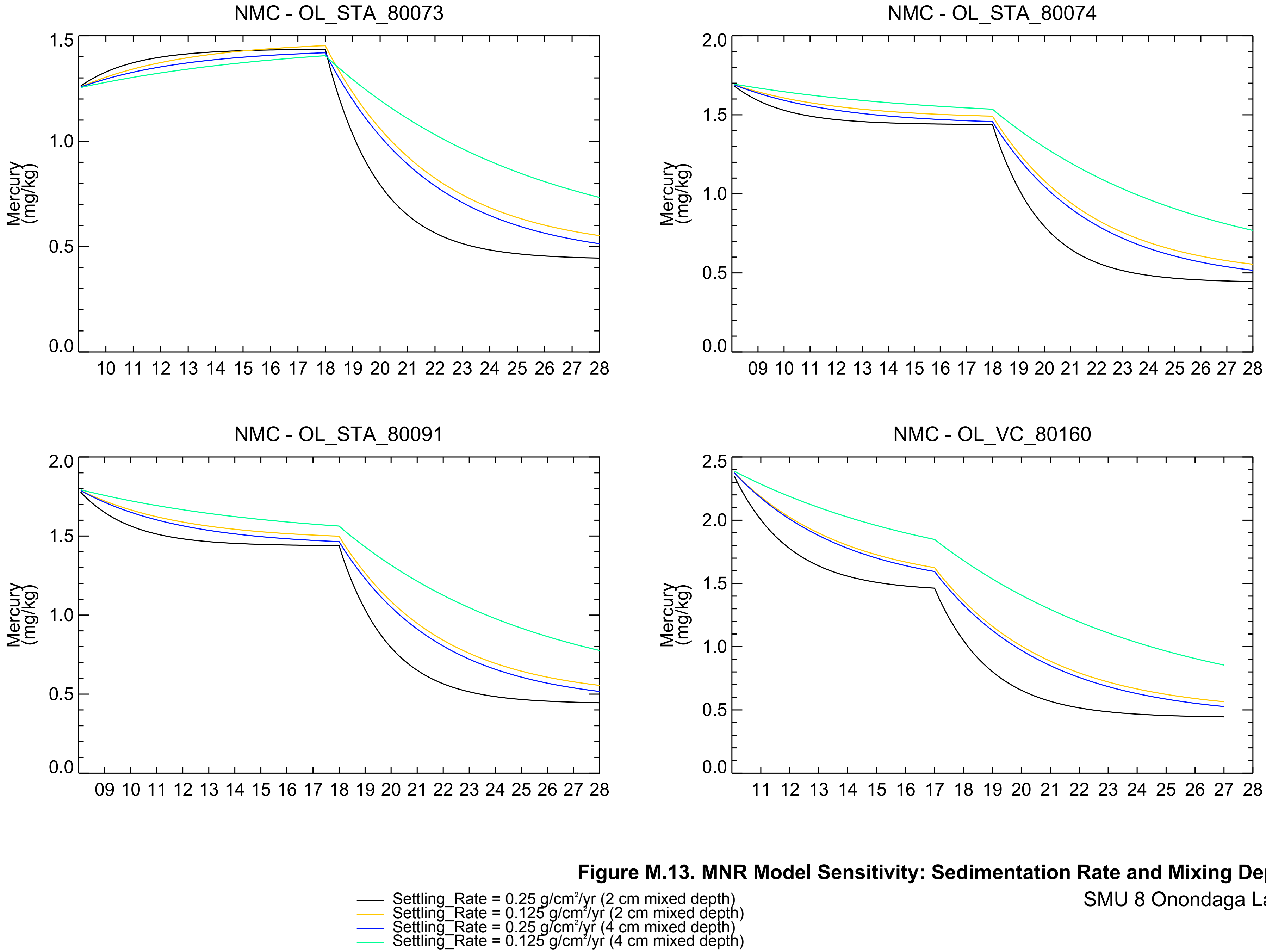


Figure M.13. MNR Model Sensitivity: Sedimentation Rate and Mixing Depth

SMU 8 Onondaga Lake

Honeywell

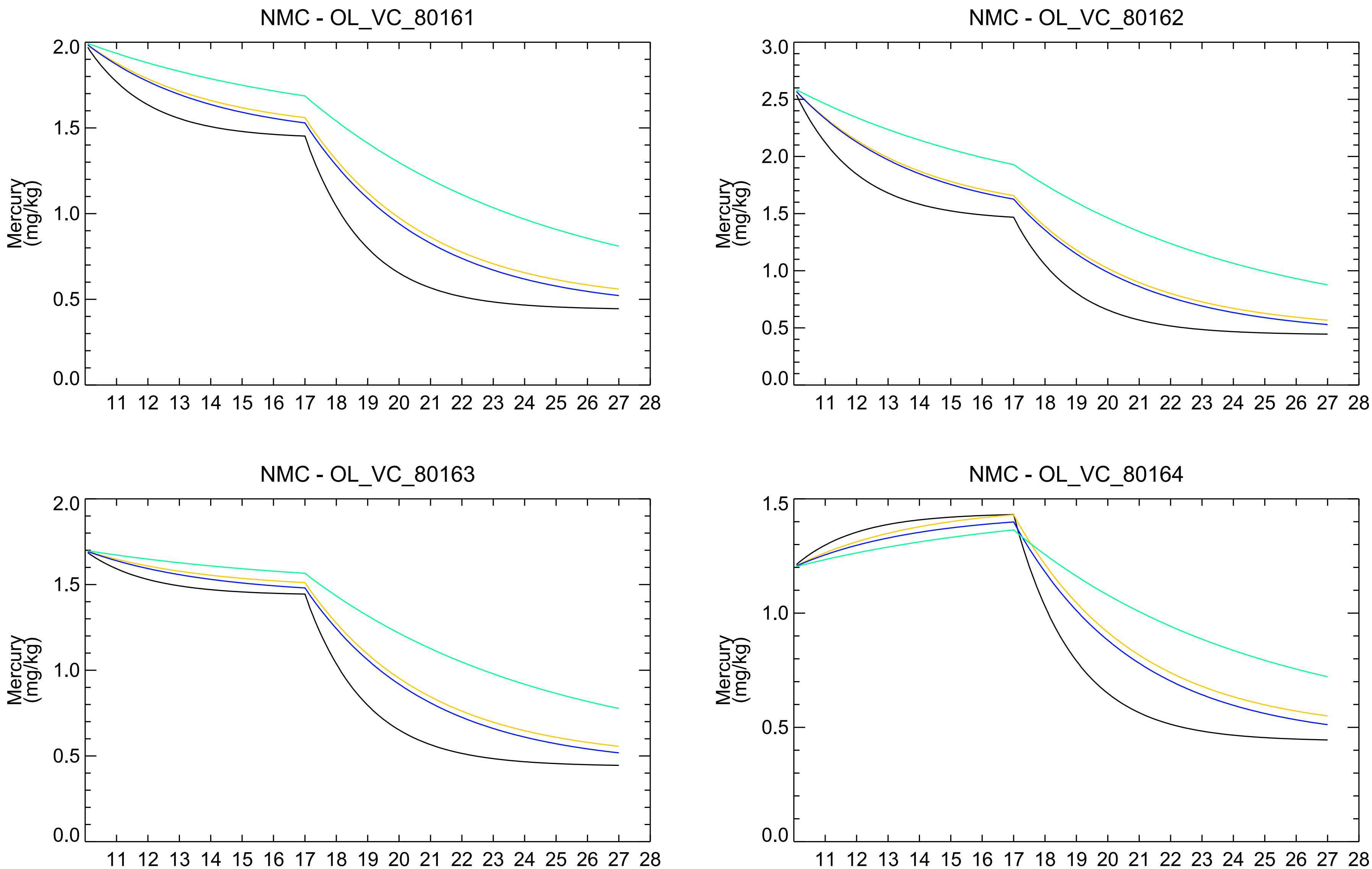


Figure M.13. MNR Model Sensitivity: Sedimentation Rate and Mixing Depth

SMU 8 Onondaga Lake

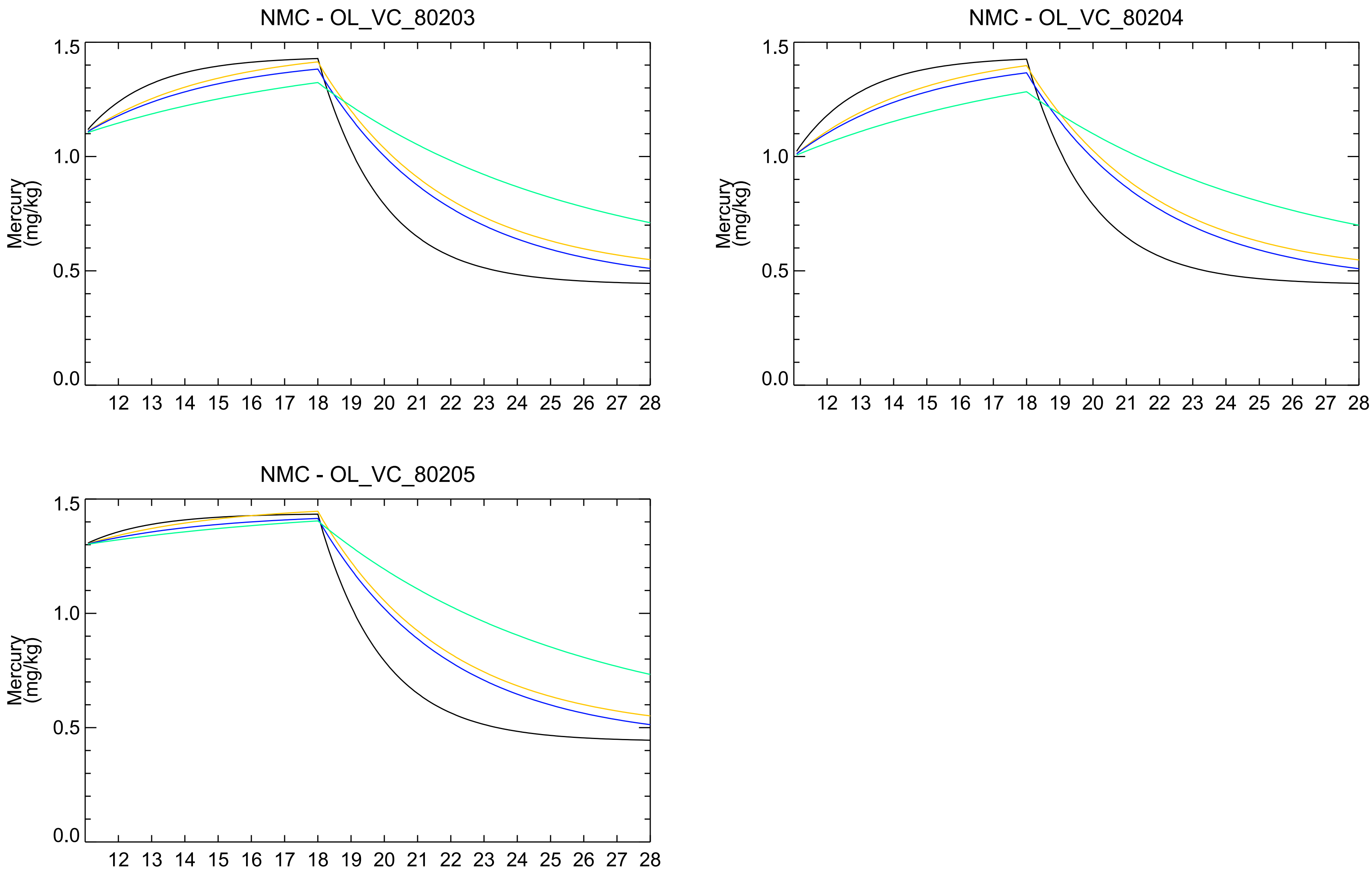


Figure M.13. MNR Model Sensitivity: Sedimentation Rate and Mixing Depth
SMU 8 Onondaga Lake

Settling_Rate = 0.25 g/cm²/yr (2 cm mixed depth)
Settling_Rate = 0.125 g/cm²/yr (2 cm mixed depth)
Settling_Rate = 0.25 g/cm²/yr (4 cm mixed depth)
Settling_Rate = 0.125 g/cm²/yr (4 cm mixed depth)

Honeywell

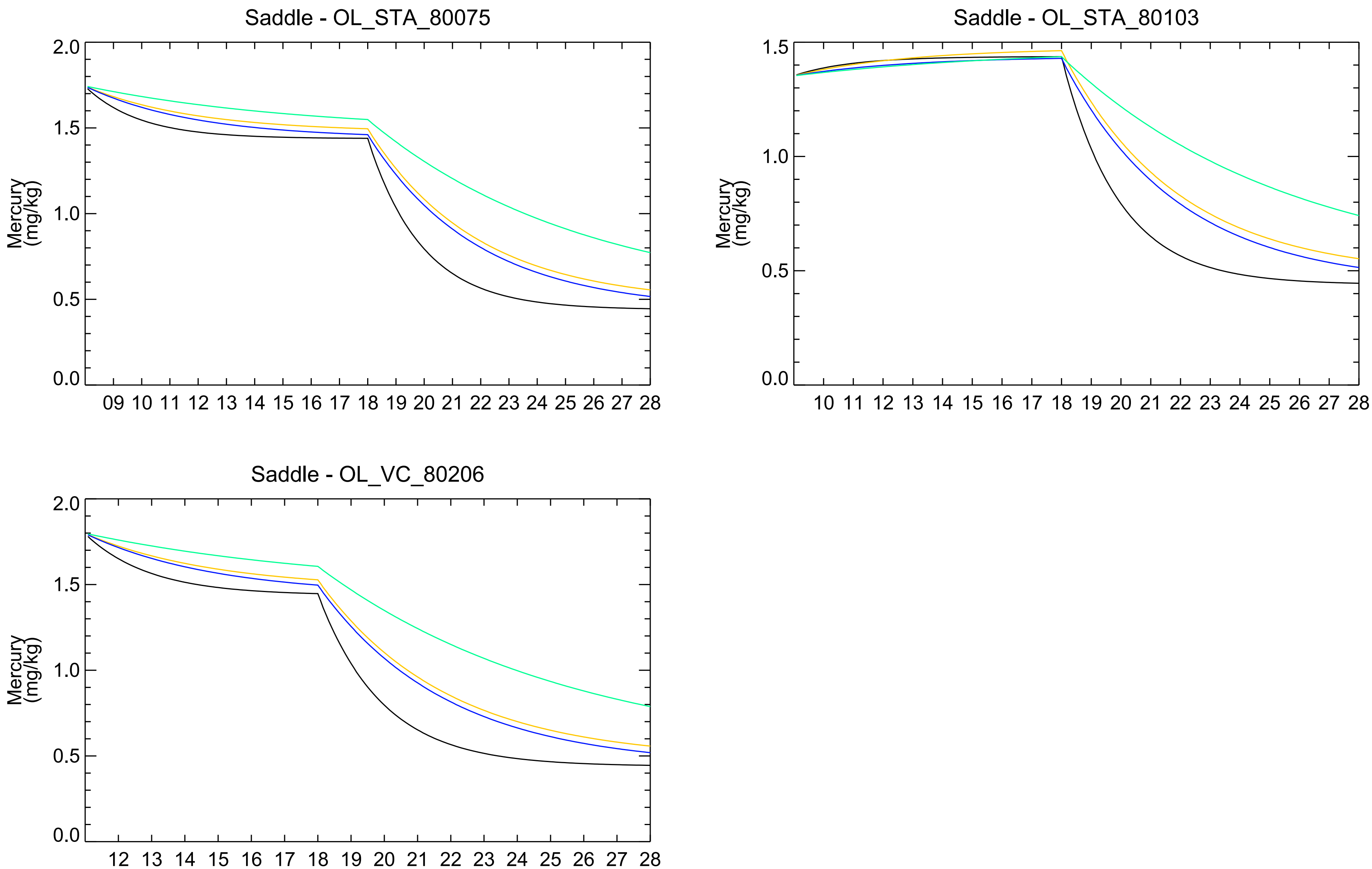


Figure M.13. MNR Model Sensitivity: Sedimentation Rate and Mixing Depth

SMU 8 Onondaga Lake

Settling_Rate = 0.25 g/cm^2/yr (2 cm mixed depth)
Settling_Rate = 0.125 g/cm^2/yr (2 cm mixed depth)
Settling_Rate = 0.25 g/cm^2/yr (4 cm mixed depth)
Settling_Rate = 0.125 g/cm^2/yr (4 cm mixed depth)

Honeywell

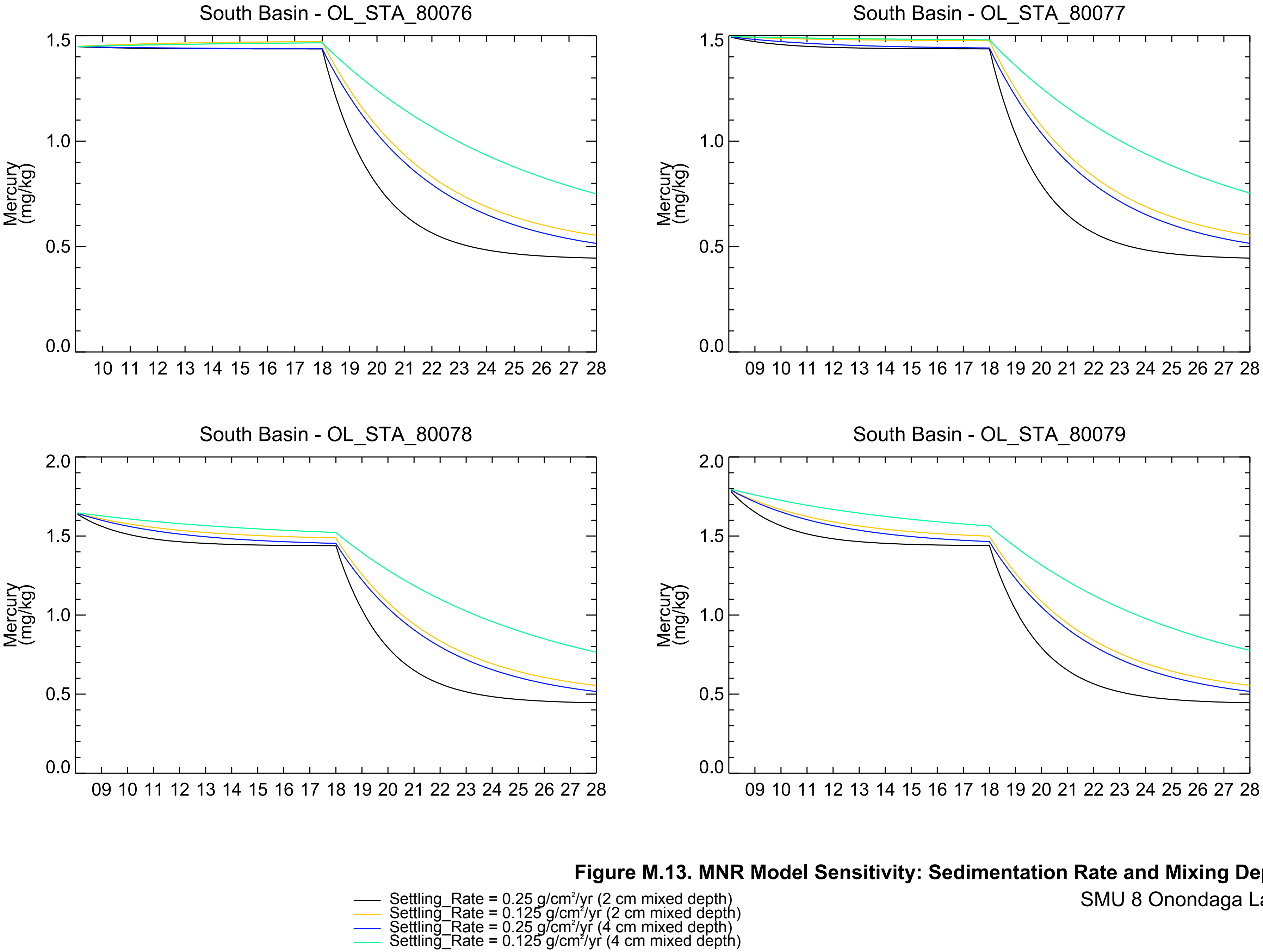


Figure M.13. MNR Model Sensitivity: Sedimentation Rate and Mixing Depth

SMU 8 Onondaga Lake

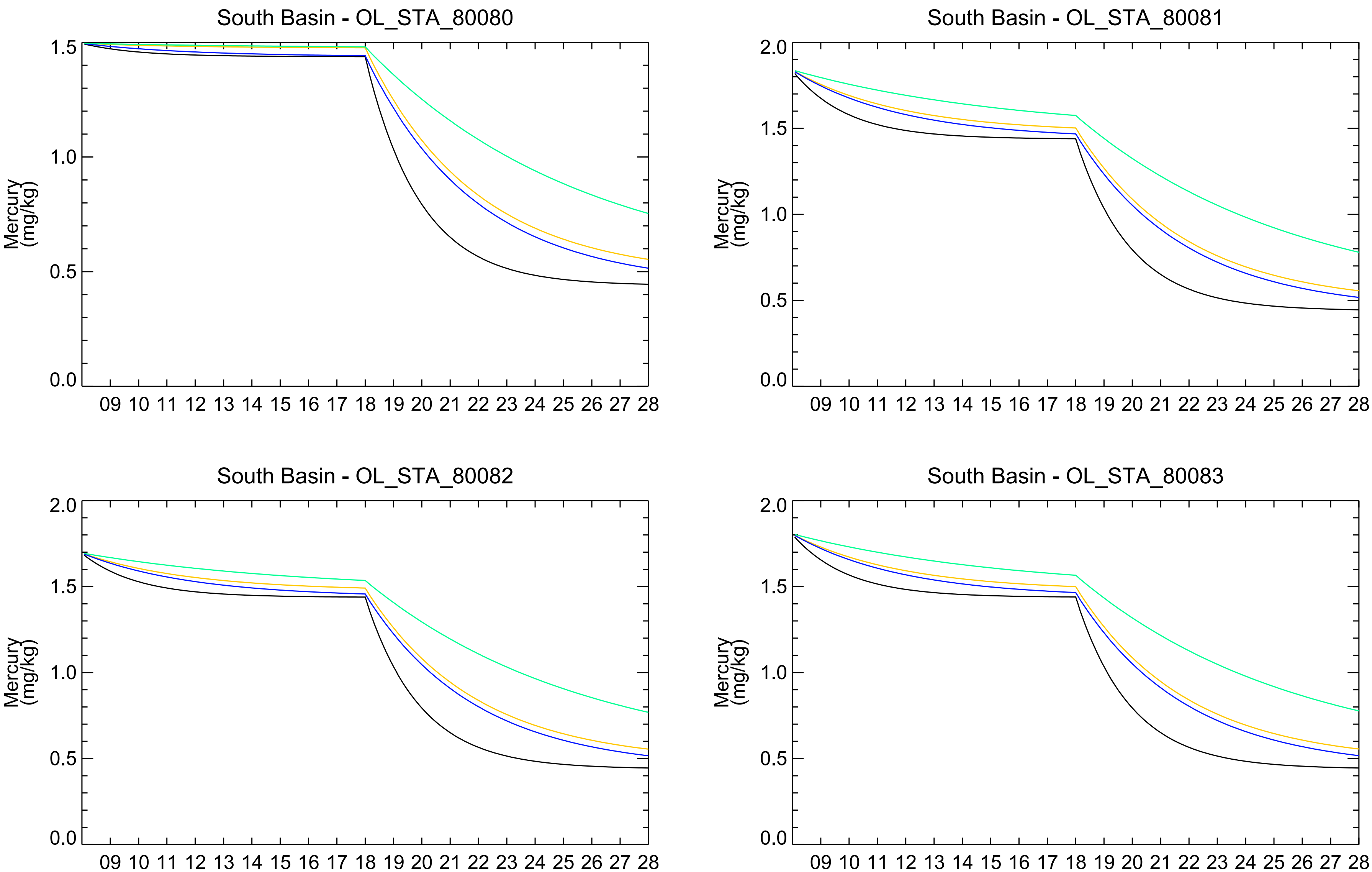


Figure M.13. MNR Model Sensitivity: Sedimentation Rate and Mixing Depth

SMU 8 Onondaga Lake

Honeywell

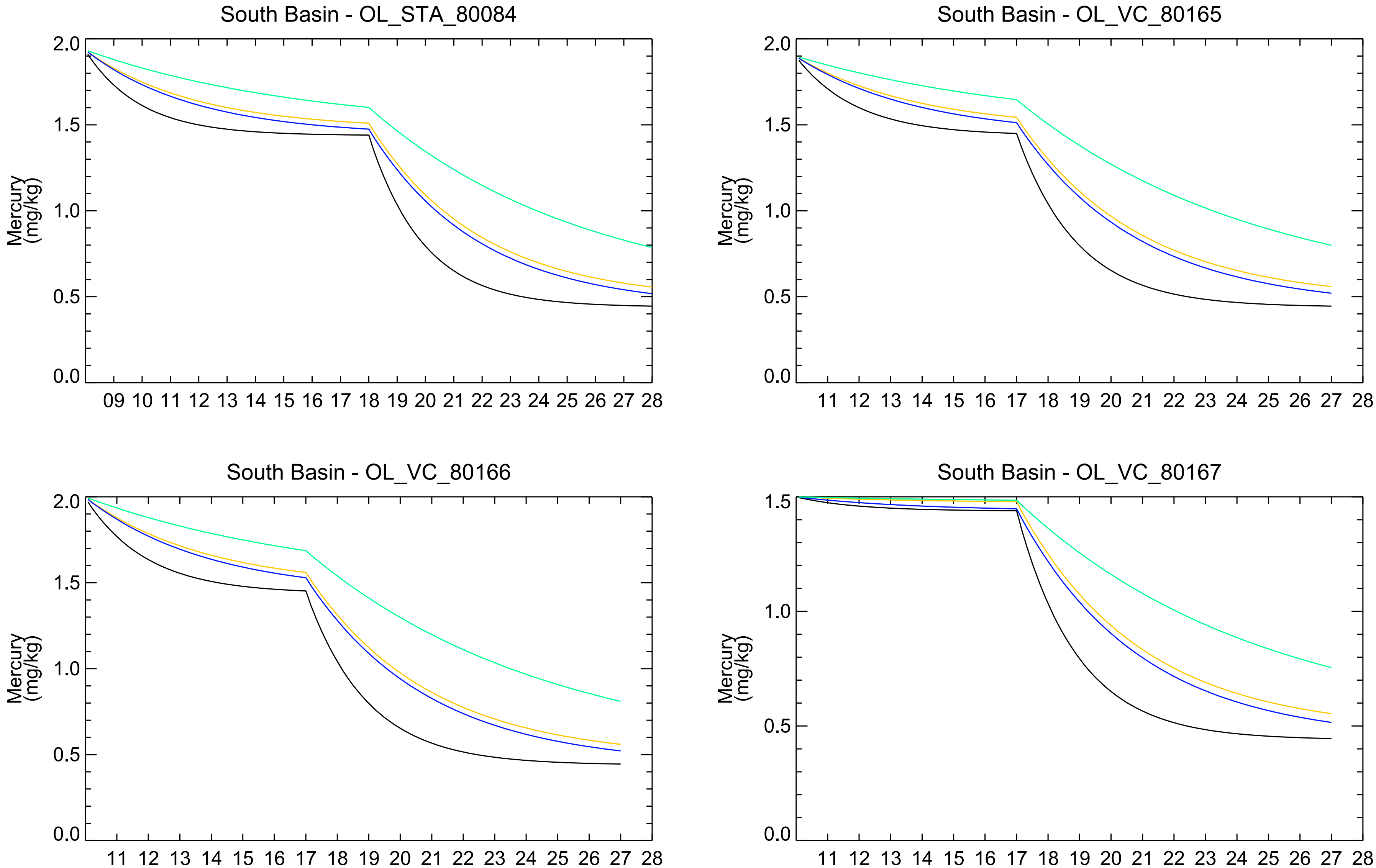


Figure M.13. MNR Model Sensitivity: Sedimentation Rate and Mixing Depth

SMU 8 Onondaga Lake

Settling_Rate = 0.25 $g/cm^2/yr$ (2 cm mixed depth)
Settling_Rate = 0.125 $g/cm^2/yr$ (2 cm mixed depth)
Settling_Rate = 0.25 $g/cm^2/yr$ (4 cm mixed depth)
Settling_Rate = 0.125 $g/cm^2/yr$ (4 cm mixed depth)

Honeywell

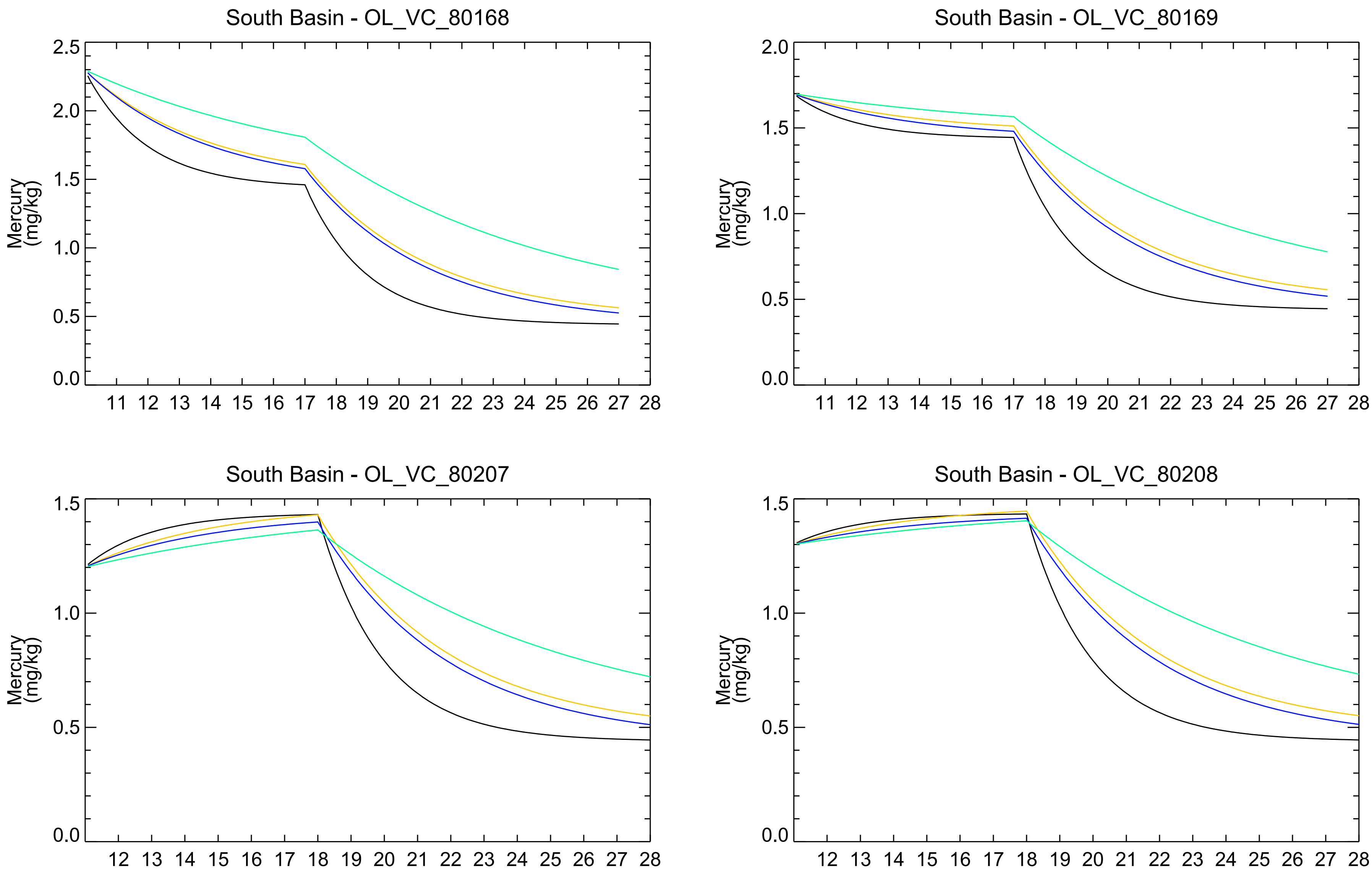


Figure M.13. MNR Model Sensitivity: Sedimentation Rate and Mixing Depth

SMU 8 Onondaga Lake

Settling_Rate = 0.25 g/cm²/yr (2 cm mixed depth)
Settling_Rate = 0.125 g/cm²/yr (2 cm mixed depth)
Settling_Rate = 0.25 g/cm²/yr (4 cm mixed depth)
Settling_Rate = 0.125 g/cm²/yr (4 cm mixed depth)

Honeywell

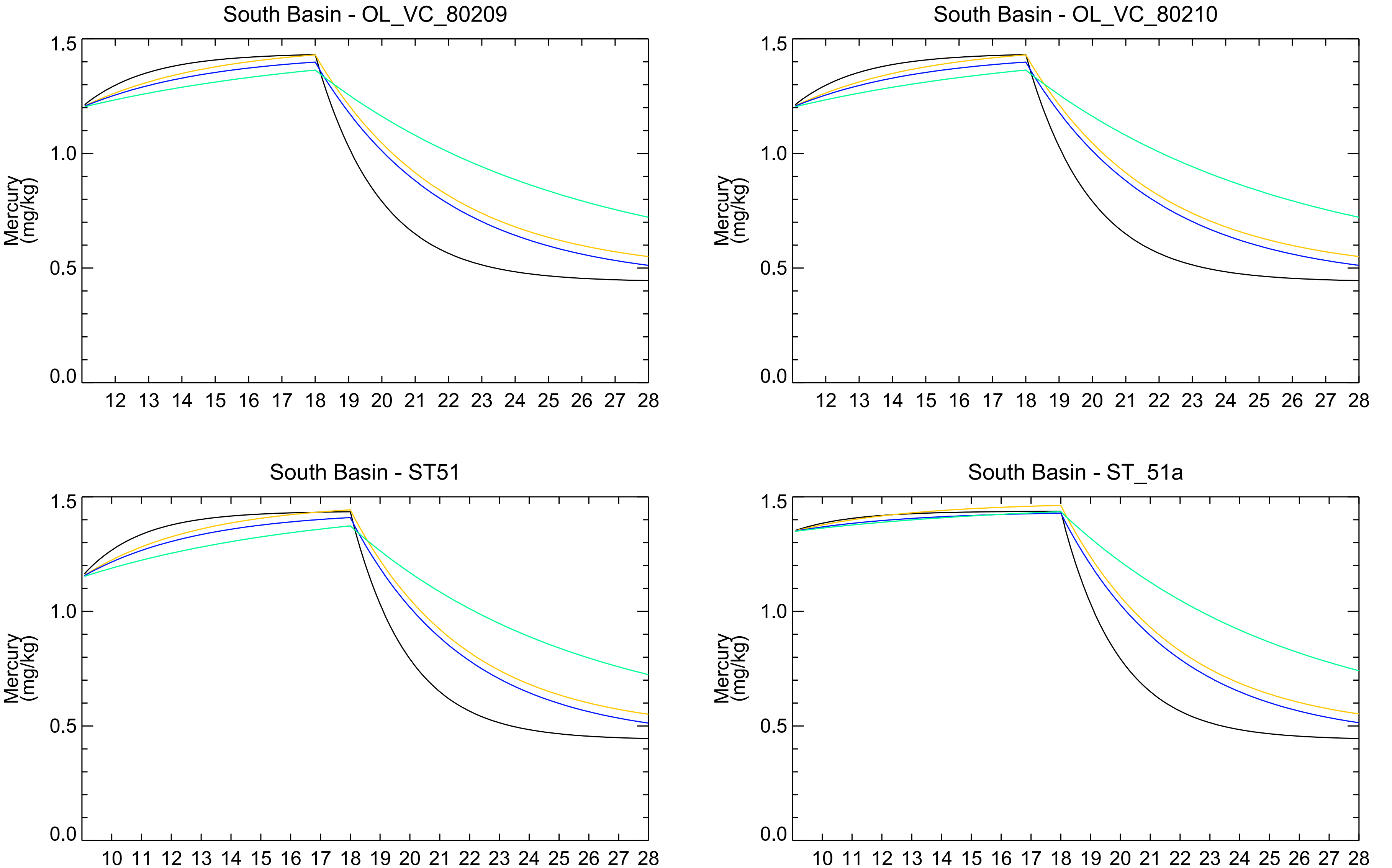


Figure M.13. MNR Model Sensitivity: Sedimentation Rate and Mixing Depth

SMU 8 Onondaga Lake

Settling_Rate = 0.25 g/cm²/yr (2 cm mixed depth)
Settling_Rate = 0.125 g/cm²/yr (2 cm mixed depth)
Settling_Rate = 0.25 g/cm²/yr (4 cm mixed depth)
Settling_Rate = 0.125 g/cm²/yr (4 cm mixed depth)

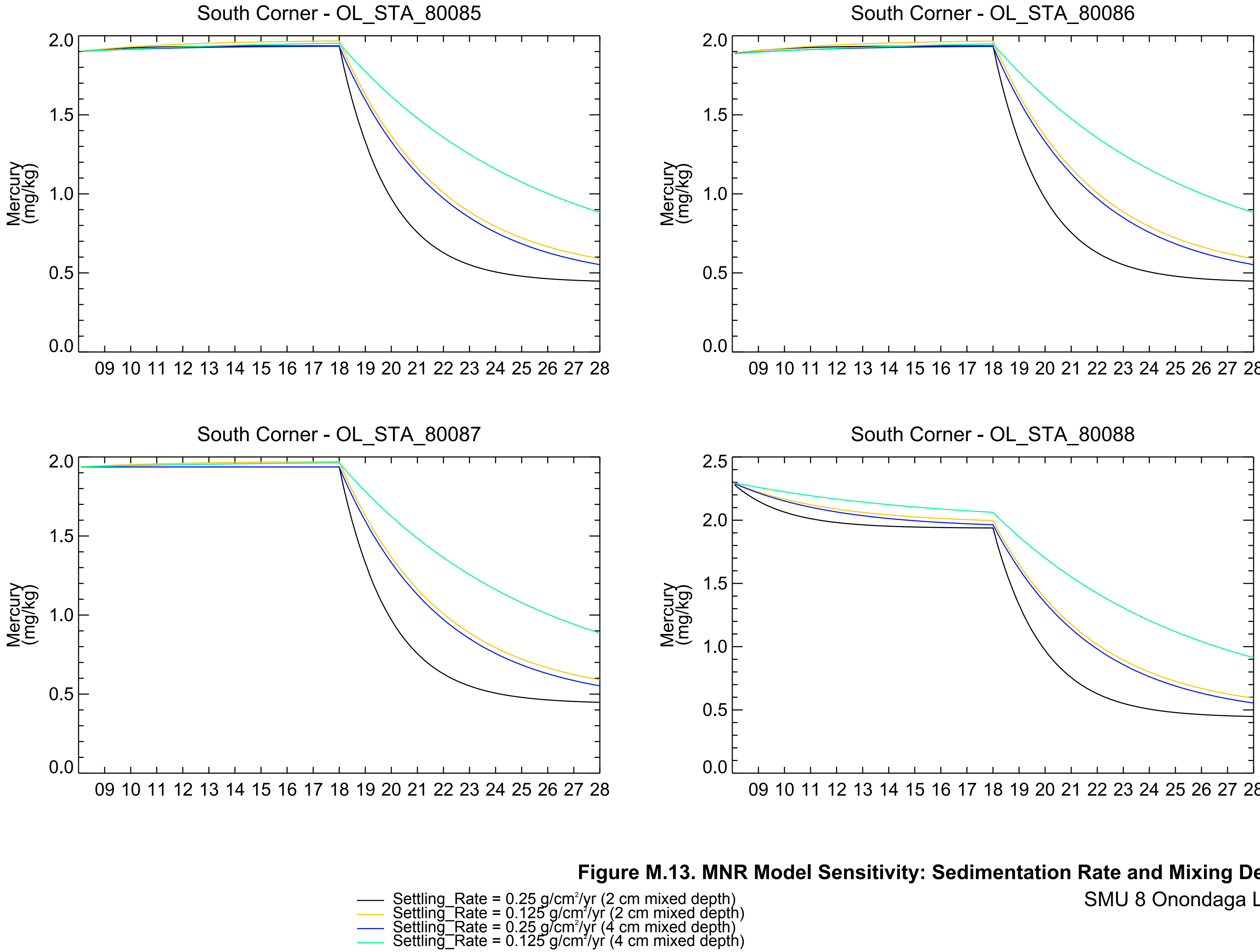


Figure M.13. MNR Model Sensitivity: Sedimentation Rate and Mixing Depth

SMU 8 Onondaga Lake

Honeywell

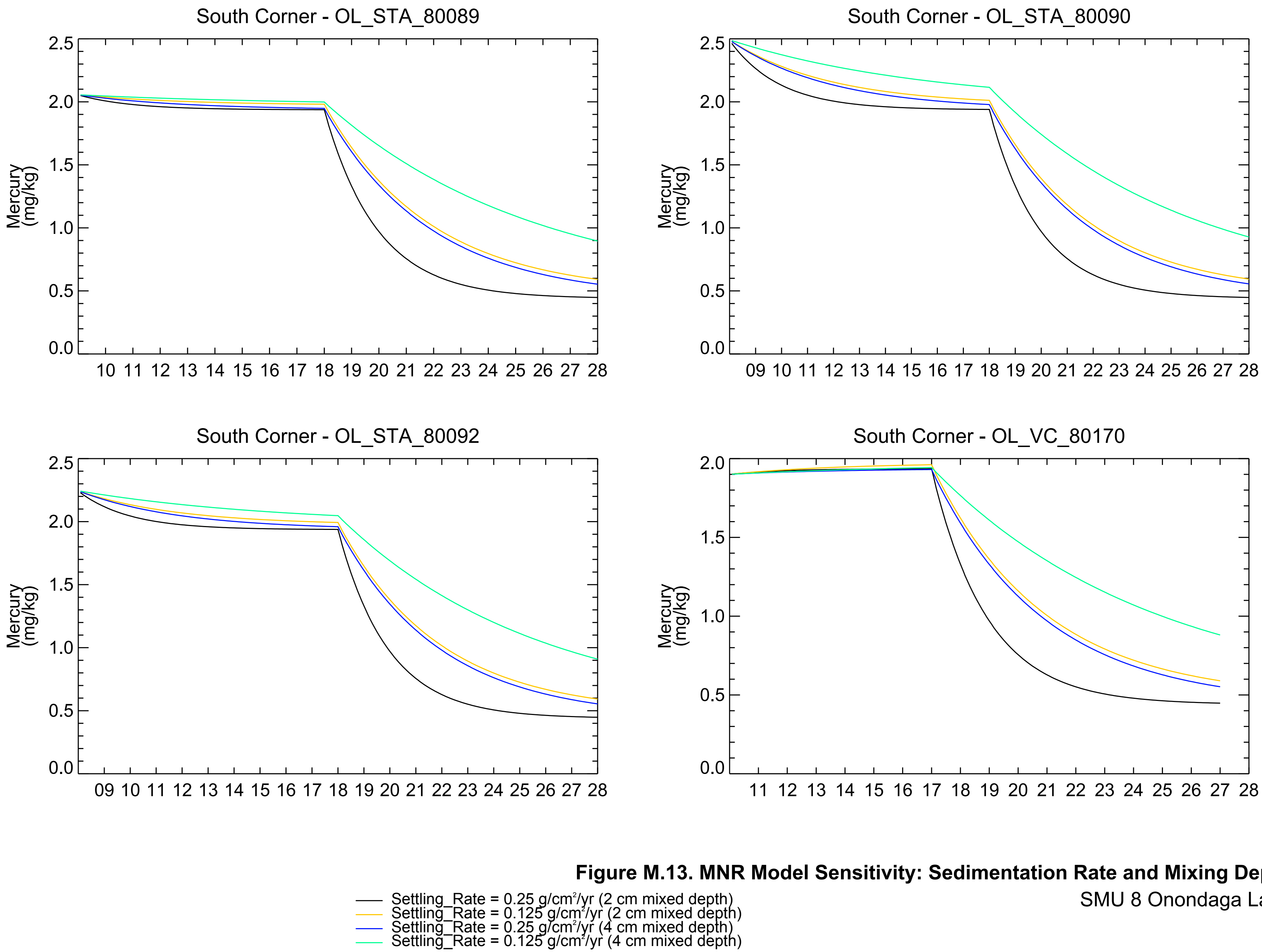


Figure M.13. MNR Model Sensitivity: Sedimentation Rate and Mixing Depth

SMU 8 Onondaga Lake

Honeywell

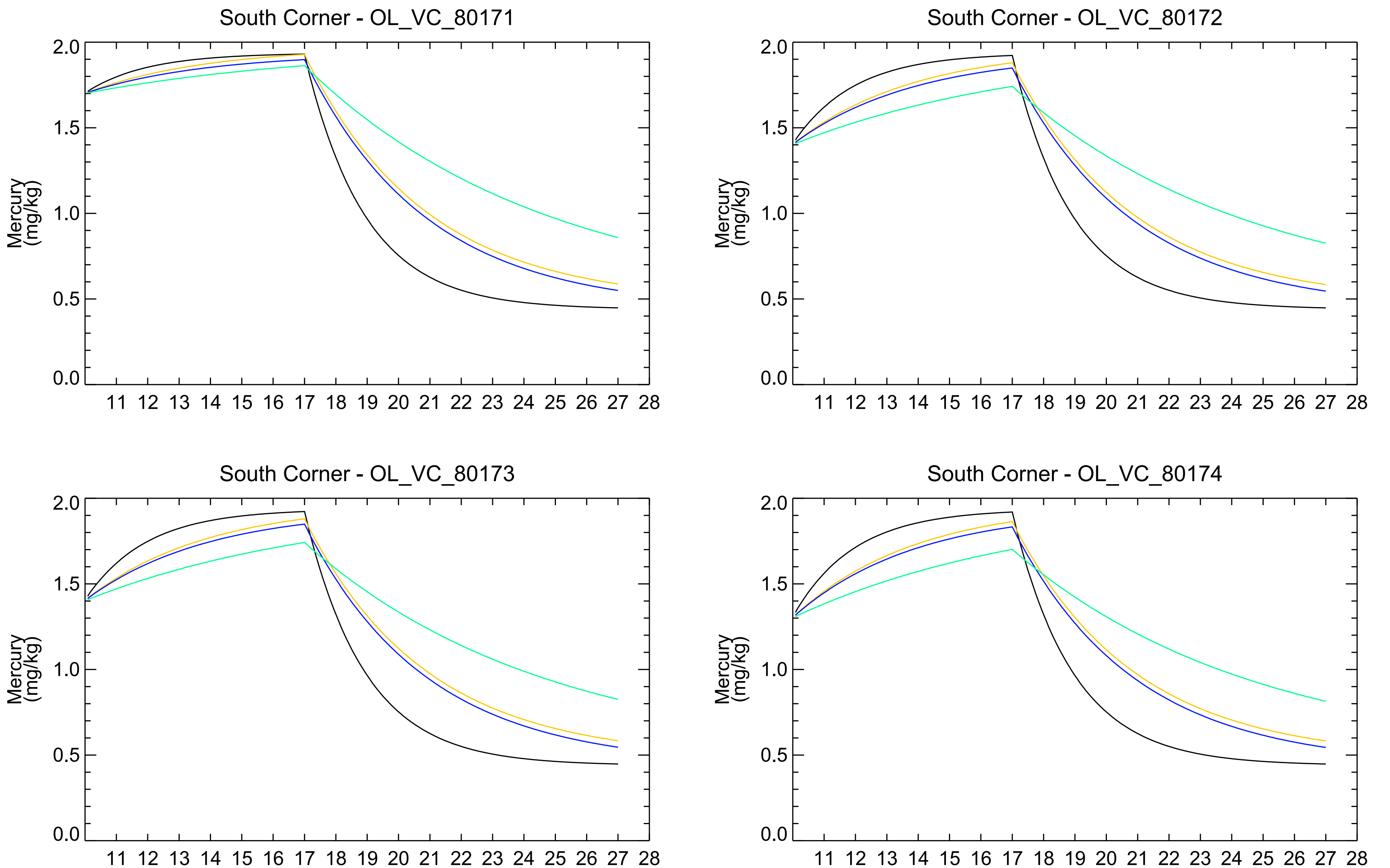


Figure M.13. MNR Model Sensitivity: Sedimentation Rate and Mixing Depth

SMU 8 Onondaga Lake

Settling_Rate = 0.25 g/cm²/yr (2 cm mixed depth)
Settling_Rate = 0.125 g/cm²/yr (2 cm mixed depth)
Settling_Rate = 0.25 g/cm²/yr (4 cm mixed depth)
Settling_Rate = 0.125 g/cm²/yr (4 cm mixed depth)

Honeywell

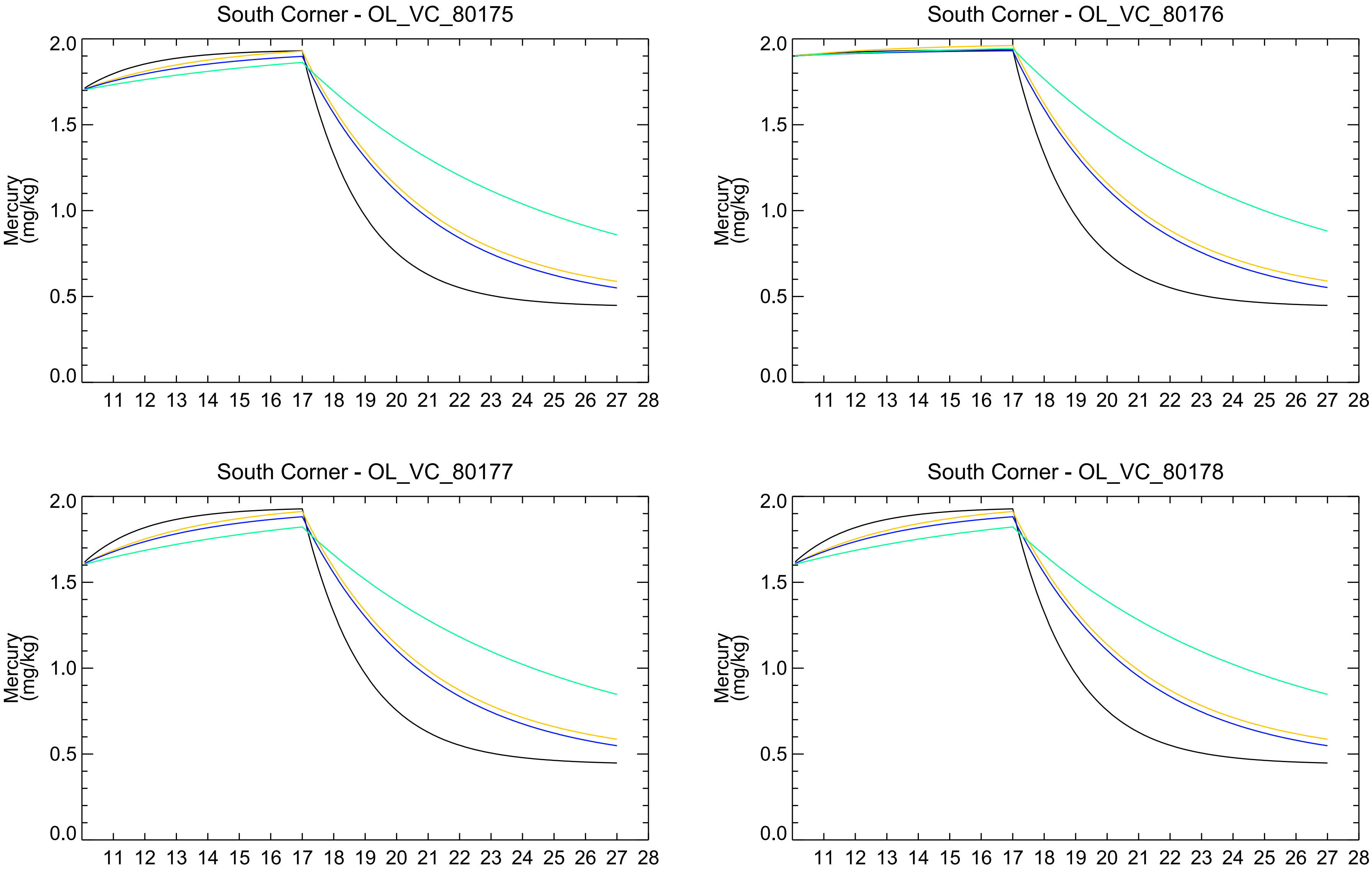


Figure M.13. MNR Model Sensitivity: Sedimentation Rate and Mixing Depth

SMU 8 Onondaga Lake

Honeywell

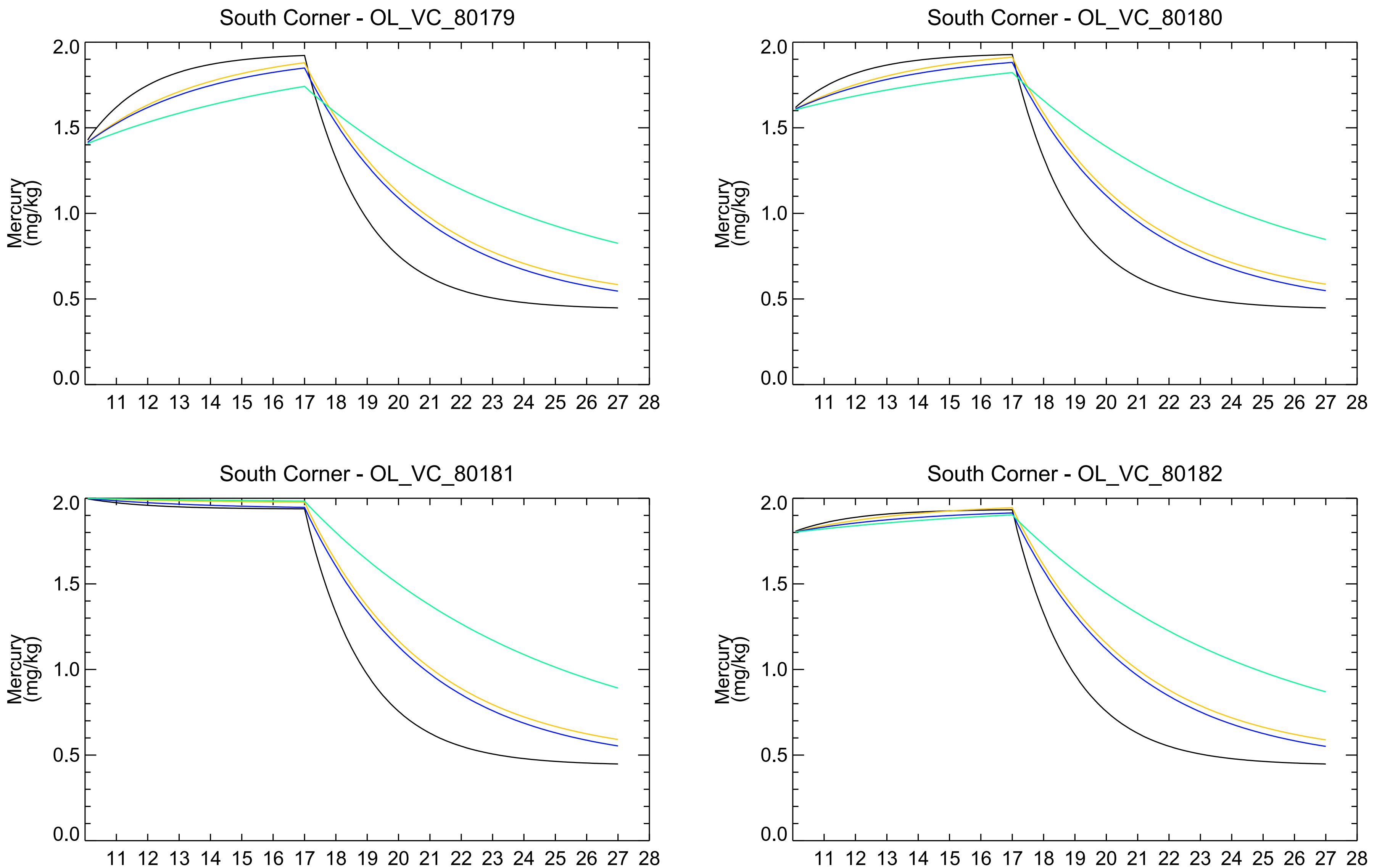


Figure M.13. MNR Model Sensitivity: Sedimentation Rate and Mixing Depth

SMU 8 Onondaga Lake

Honeywell

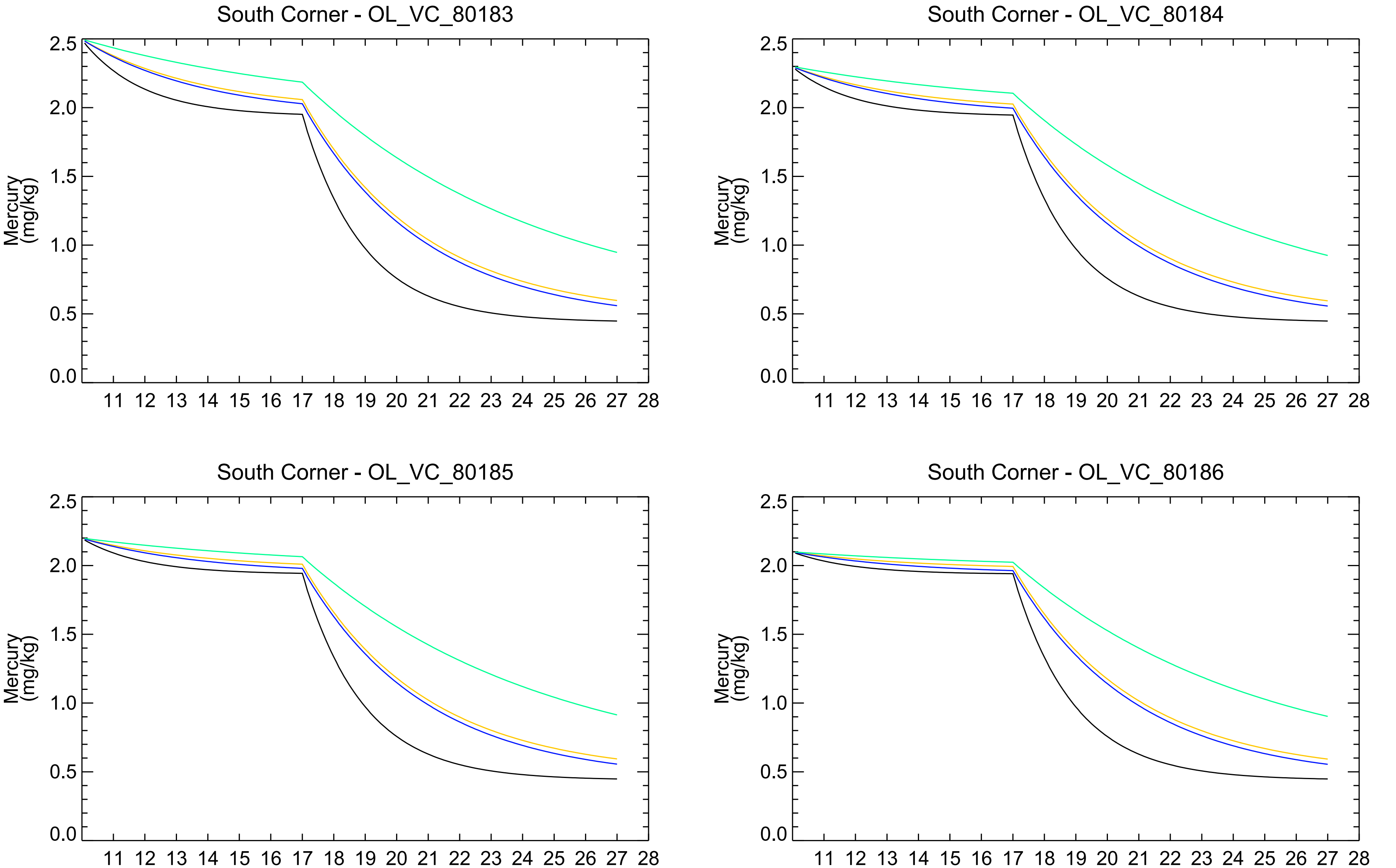


Figure M.13. MNR Model Sensitivity: Sedimentation Rate and Mixing Depth

SMU 8 Onondaga Lake

Settling_Rate = 0.25 g/cm²/yr (2 cm mixed depth)
Settling_Rate = 0.125 g/cm²/yr (2 cm mixed depth)
Settling_Rate = 0.25 g/cm²/yr (4 cm mixed depth)
Settling_Rate = 0.125 g/cm²/yr (4 cm mixed depth)

Honeywell

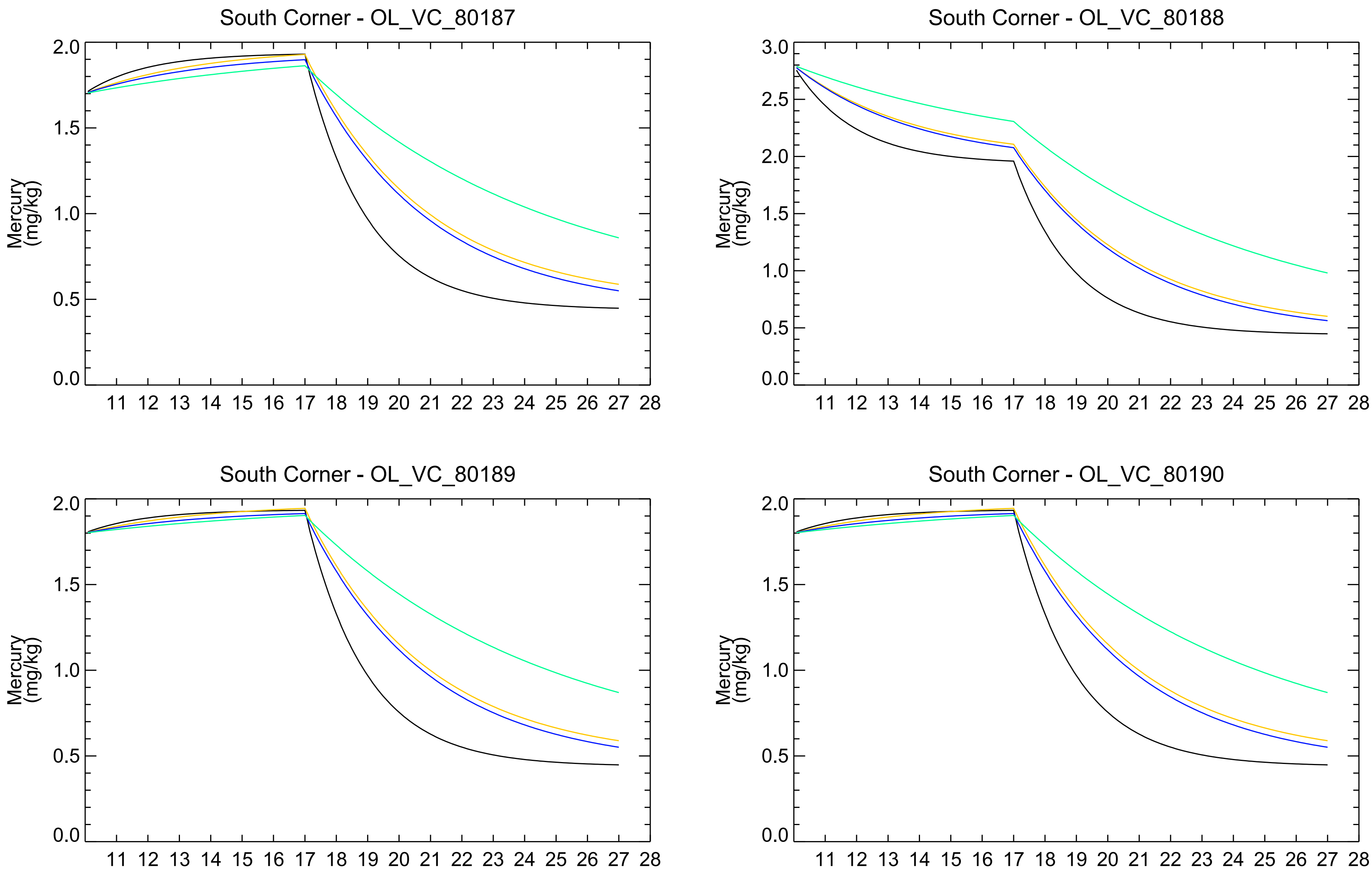


Figure M.13. MNR Model Sensitivity: Sedimentation Rate and Mixing Depth

SMU 8 Onondaga Lake

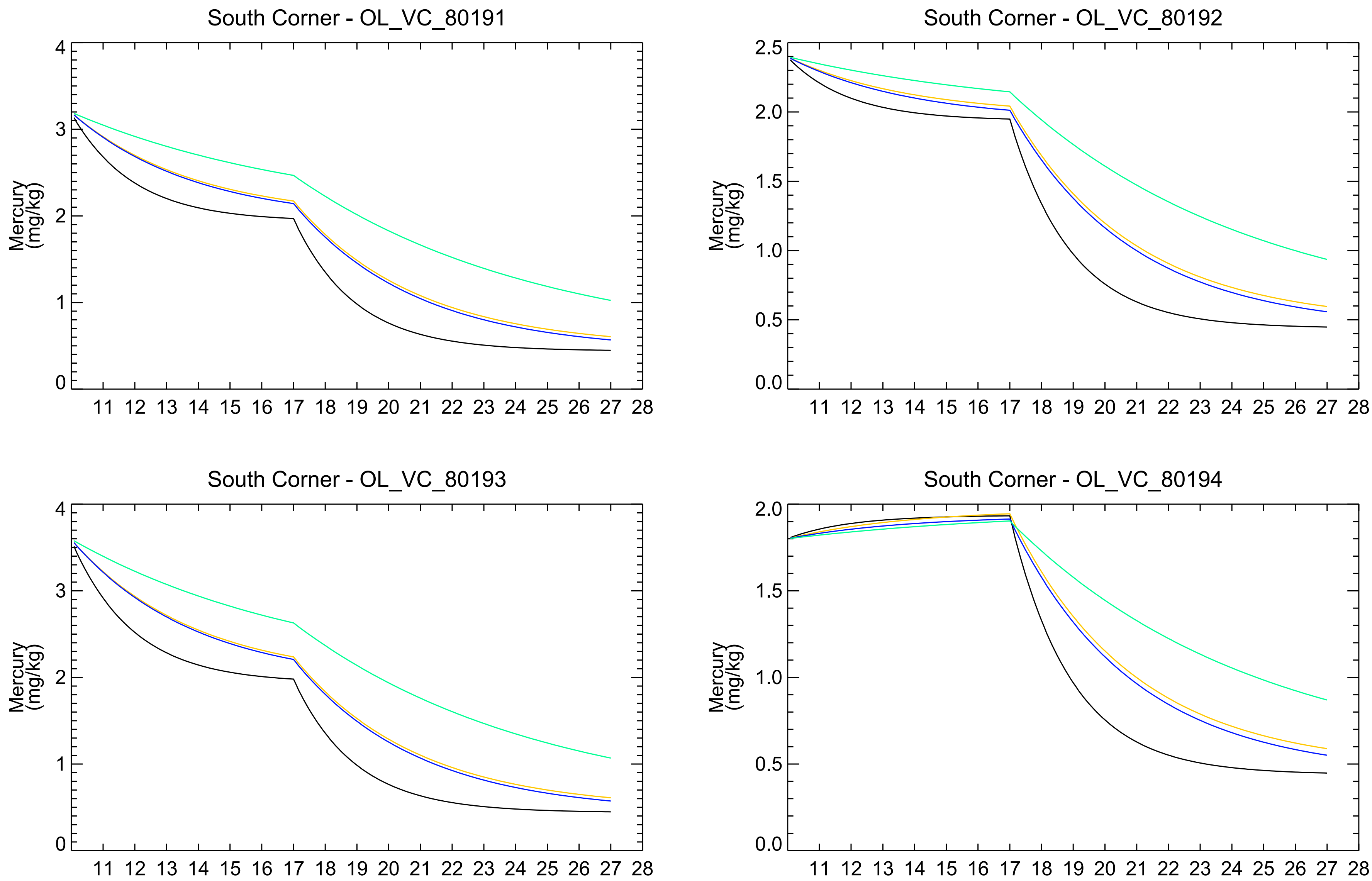
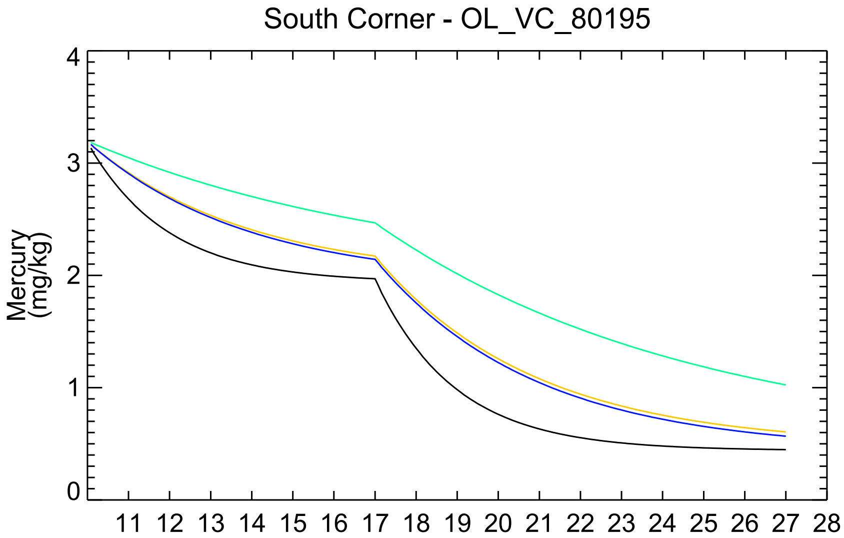
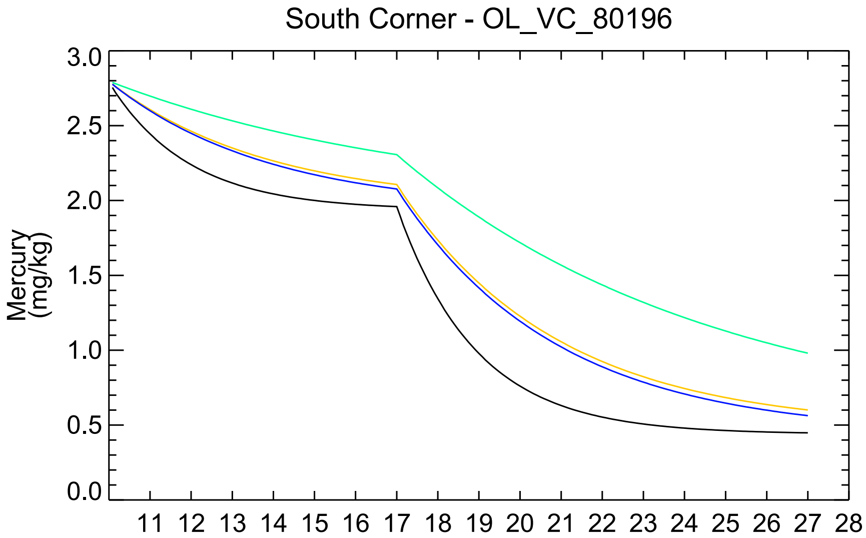
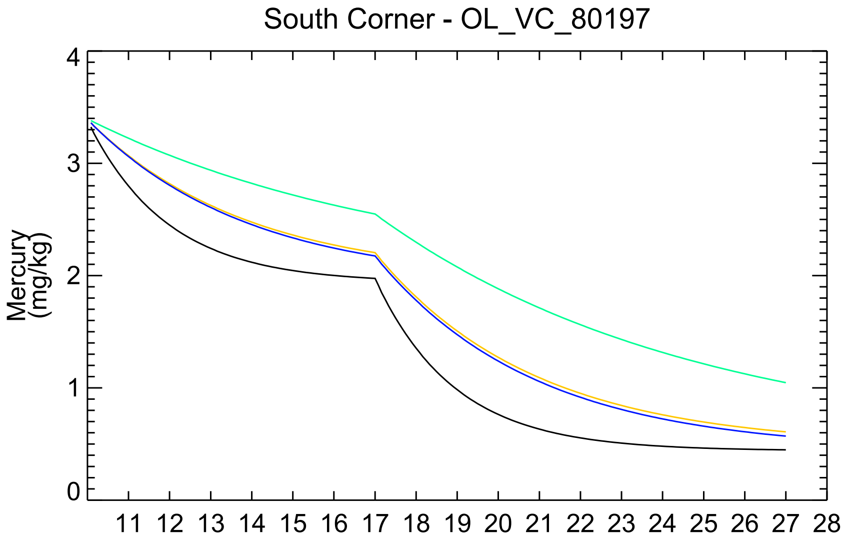


Figure M.13. MNR Model Sensitivity: Sedimentation Rate and Mixing Depth

SMU 8 Onondaga Lake







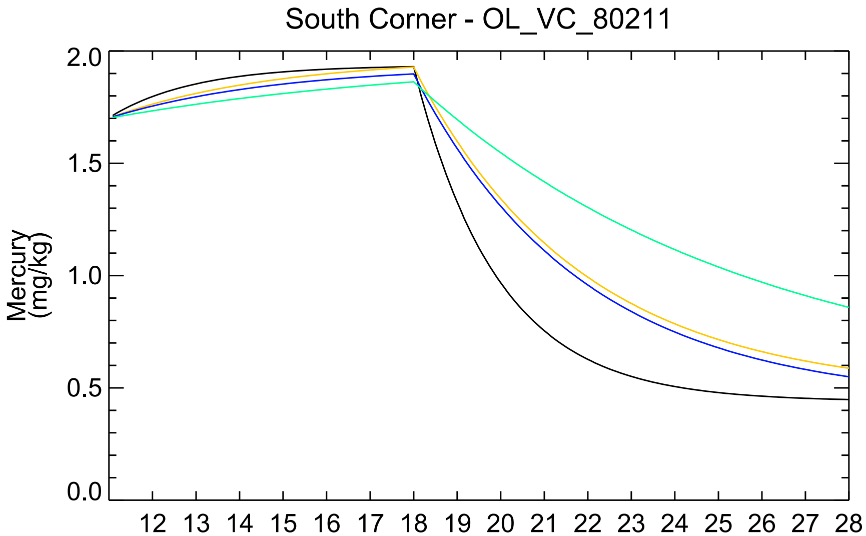


Figure M.13. MNR Model Sensitivity: Sedimentation Rate and Mixing Depth

SMU 8 Onondaga Lake

Settling_Rate = 0.25 g/cm²/yr (2 cm mixed depth)
Settling_Rate = 0.125 g/cm²/yr (2 cm mixed depth)
Settling_Rate = 0.25 g/cm²/yr (4 cm mixed depth)
Settling_Rate = 0.125 g/cm²/yr (4 cm mixed depth)

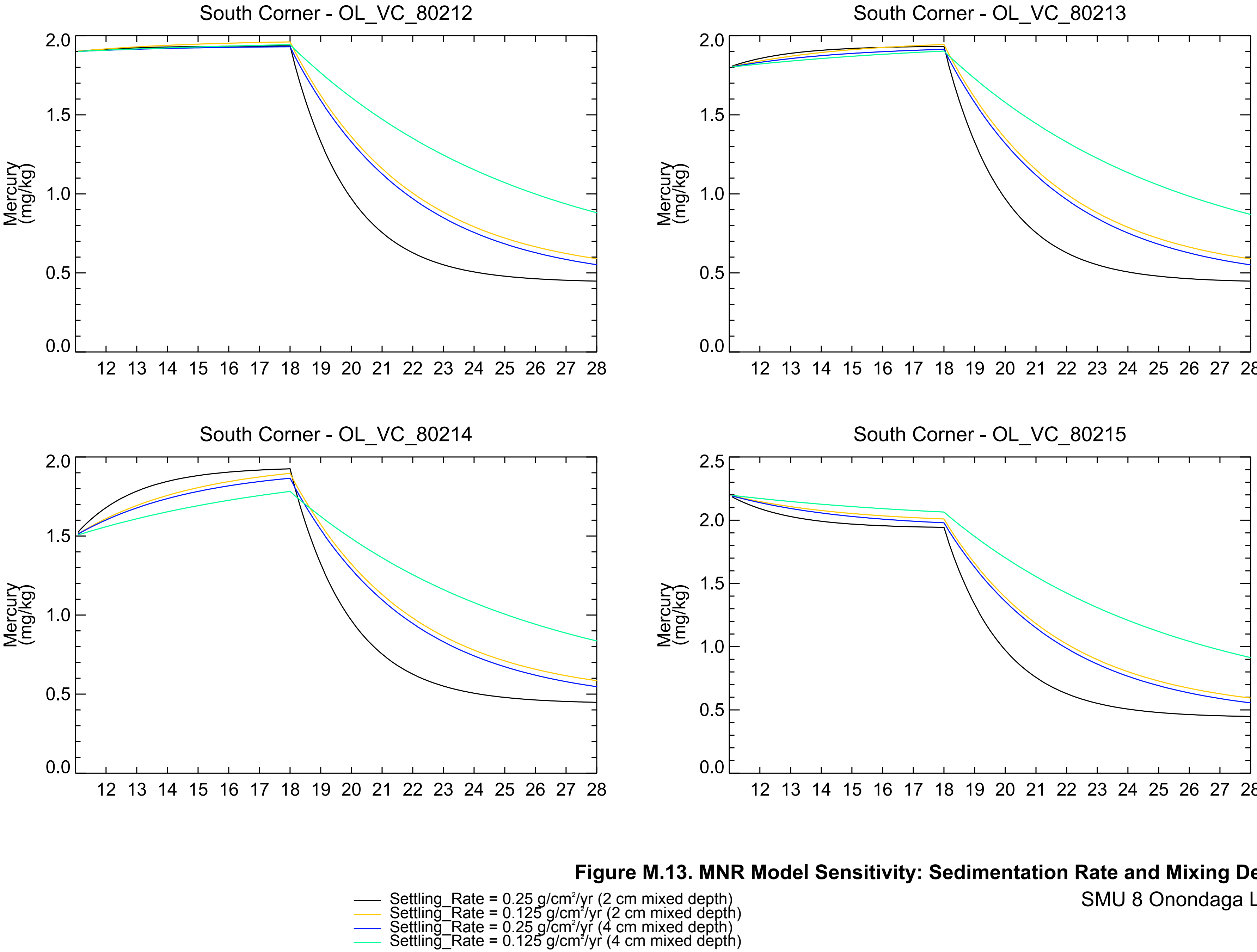


Figure M.13. MNR Model Sensitivity: Sedimentation Rate and Mixing Depth

SMU 8 Onondaga Lake

Honeywell

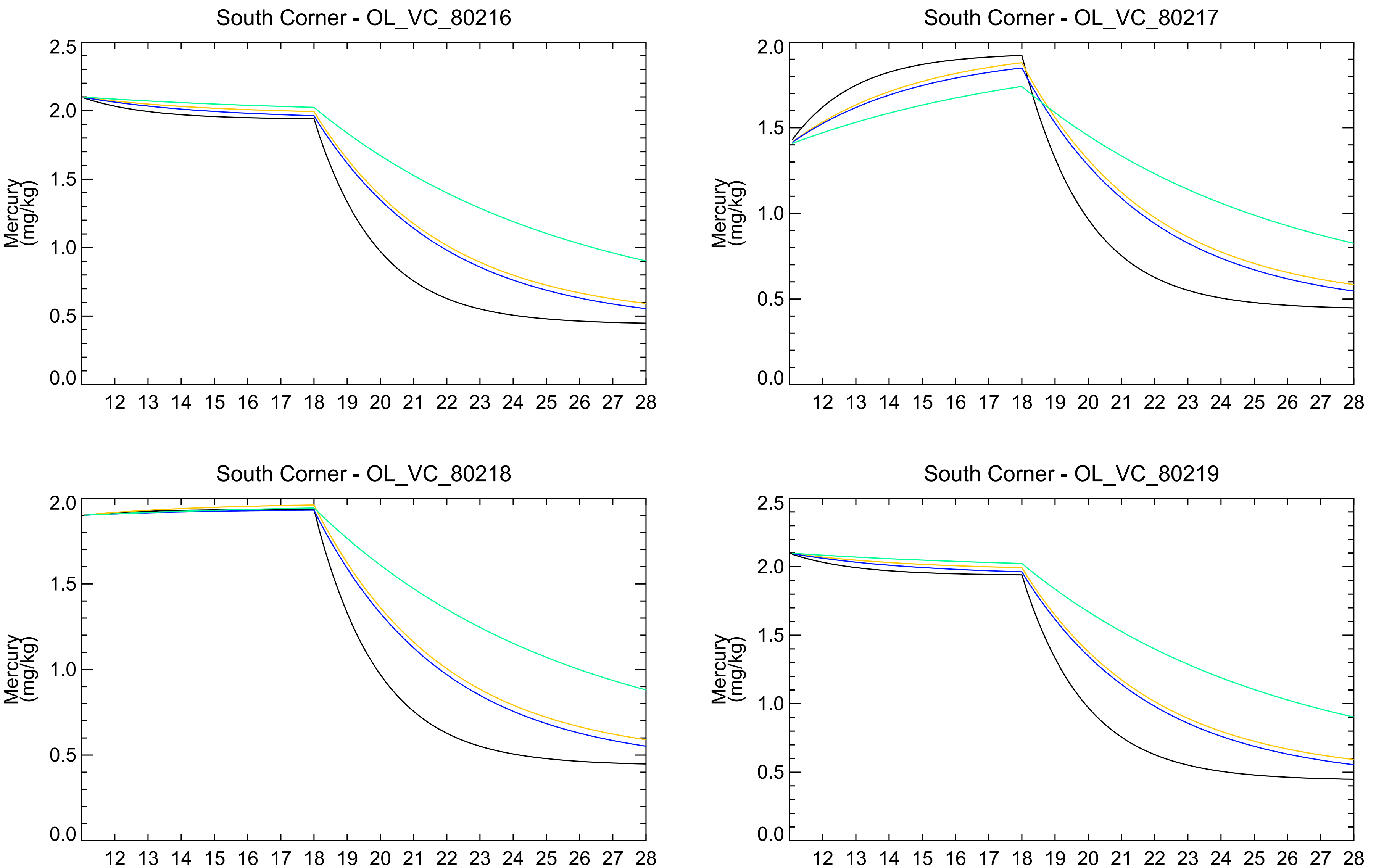


Figure M.13. MNR Model Sensitivity: Sedimentation Rate and Mixing Depth

SMU 8 Onondaga Lake

Settling_Rate = 0.25 g/cm²/yr (2 cm mixed depth)
Settling_Rate = 0.125 g/cm²/yr (2 cm mixed depth)
Settling_Rate = 0.25 g/cm²/yr (4 cm mixed depth)
Settling_Rate = 0.125 g/cm²/yr (4 cm mixed depth)

Honeywell

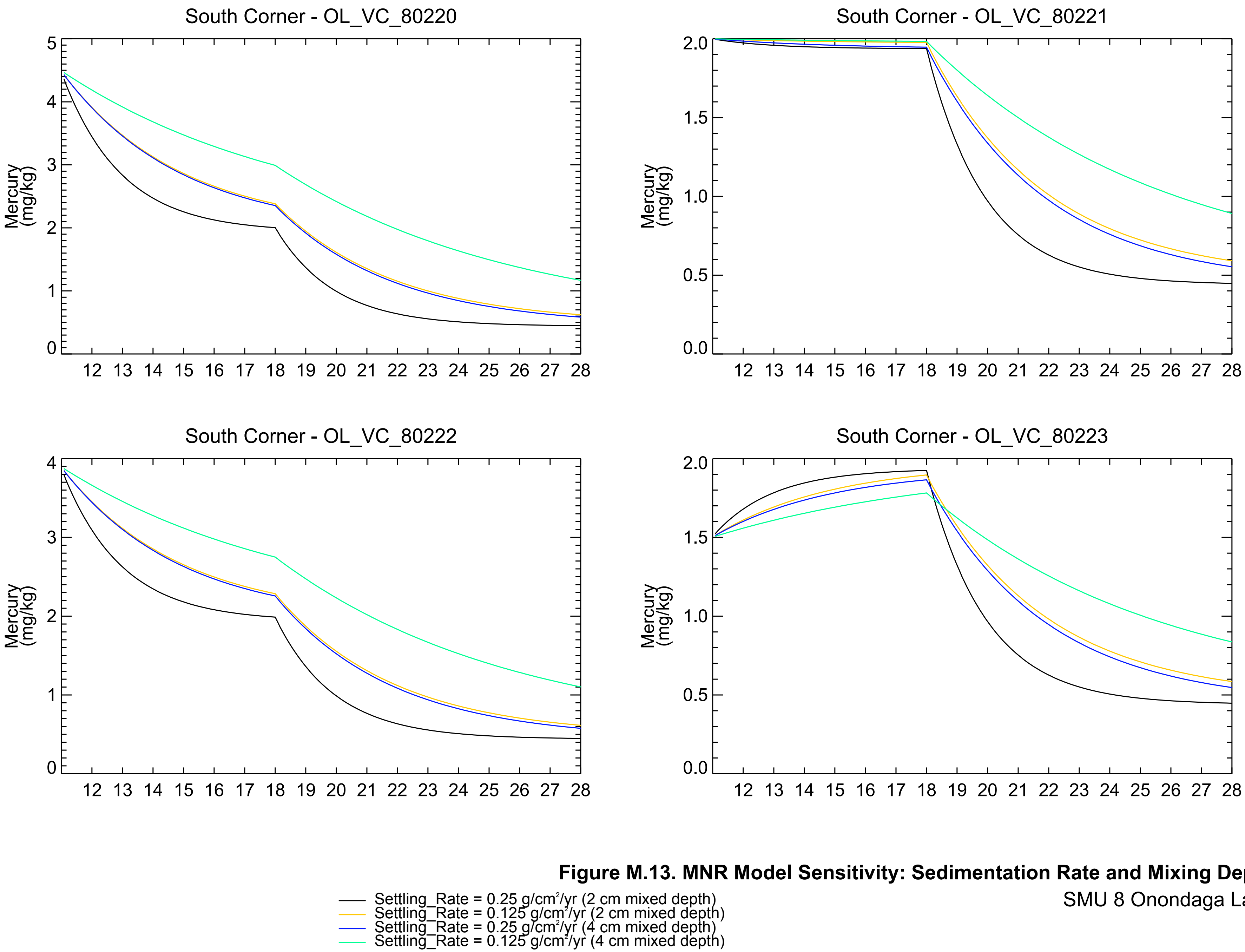
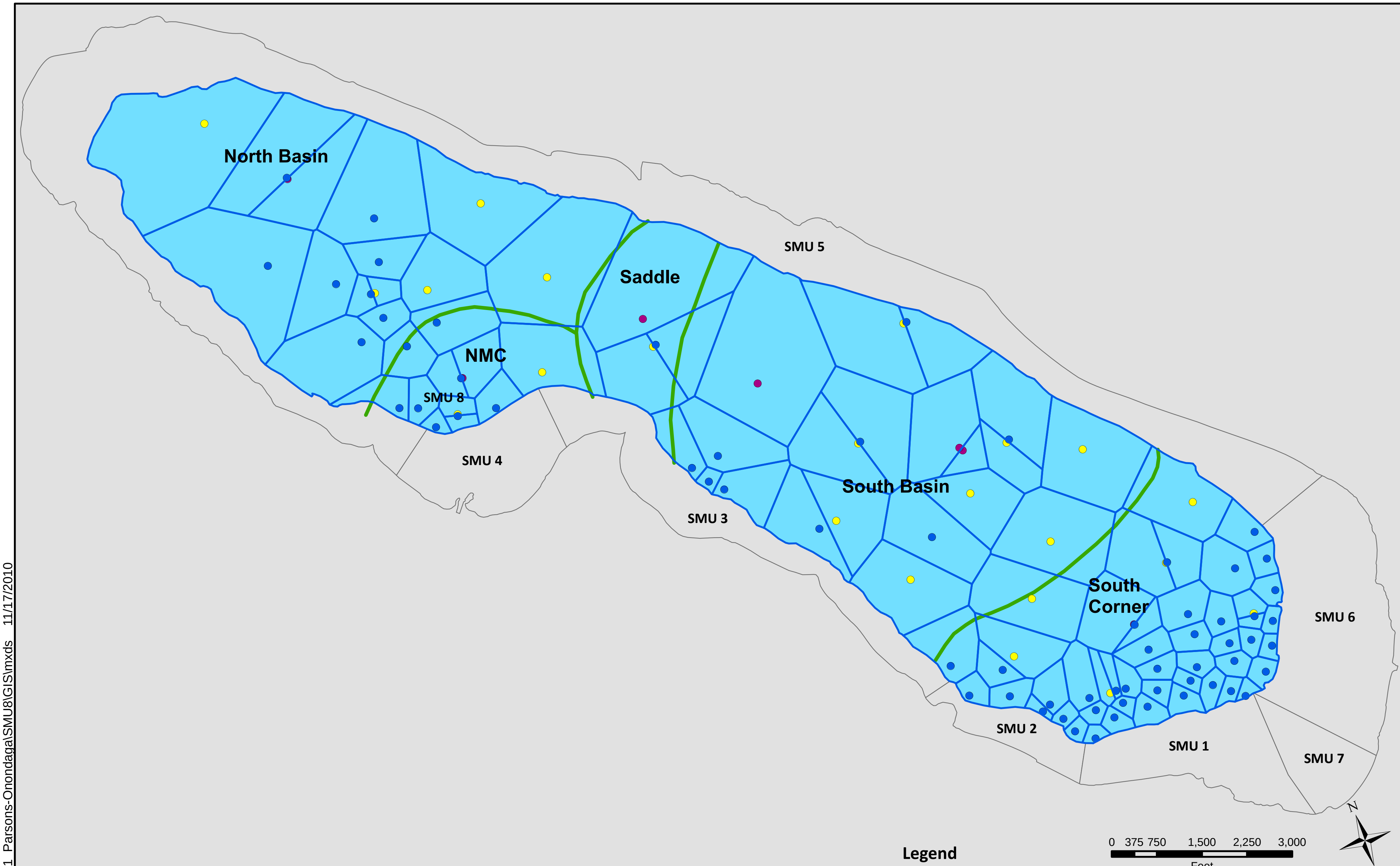


Figure M.13. MNR Model Sensitivity: Sedimentation Rate and Mixing Depth

SMU 8 Onondaga Lake



Legend

Model Locations

- 2007
- 2008
- 2010
- Thiessen Polygons

FIGURE M.14

Honeywell

Onondaga Lake
Syracuse, New York

MNR Model Locations for Evaluation of BSQV Compliance

\\helios\agld drive\jobs\090139-01 Parsons-Onondaga\SMU8\GIS\mxds 11/17/2010