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Toxicity and bioaccumulation in benthic organisms

R.P. Mason and E.T. Porter

Toxic chemicals that enter coastal oceans accumulate in the water, sediments, and biota. Sediments act as a repository for toxins and exchange processes between sediments and the overlying water affect the bioavailability of toxins for organisms. Mesocosm experiments are particularly important for toxicant studies because of the various feedback mechanisms between water, sediment, and biota. Mesocosm results combined with modeling results demonstrated the importance of organic matter binding of toxicants. For example, higher organic matter content of sediments correlated with less methylmercury bioaccumulation in zooplankton. The implication of these results is that food web dynamics and eutrophication status in coastal waters has a larger impact on toxicant dynamics than physical processes such as sediment resuspension. Toxicant bioaccumulation needs to be monitored closely when restoration efforts result in changes in nutrient loading.

Problem description

The coastal zone has been highly affected by human activities, which has resulted in a large insult of chemicals being introduced into these fragile ecosystems. Estuarine sediments may contain a complex mixture of contaminants because estuaries are often near urban areas and may have received substantial inputs over time from human activities, either by direct discharge or from runoff from the watershed.[87,88] The intense physical mixing of the water column and the strong benthic-pelagic coupling that exists in estuaries has important effects on contaminant fate and burial within the sediment. Sediment burial is often the main removal mechanism of pollutants from the system. Consequently, sediments are the long-term

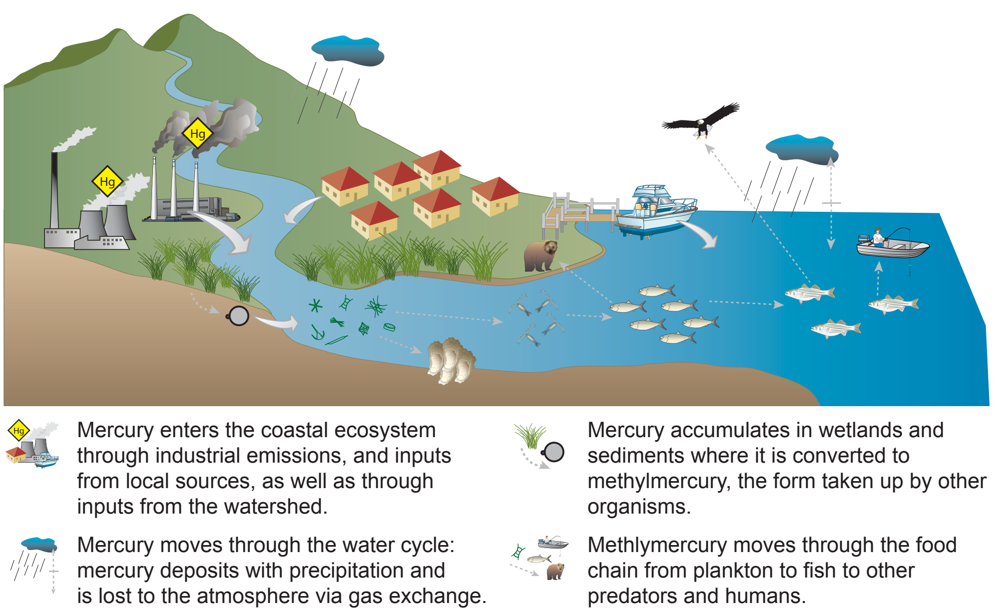

***Figure 235**: Toxic chemical such as mercury enter coastal ecosystems and accumulate in the food chain.*

87. EPA 2004, 88. Bianchi 2007

sink for many chemicals of concern, such as mercury, polycyclic aromatic hydrocarbons (PAHs), and polychlorinated biphenyls (PCBs), because of their toxicity to and bioaccumulation by aquatic organisms, birds, and mammals, including humans.[89,90]

Because sediments act as a repository for contaminants in coastal systems, processes that govern the exchange of chemicals between the water, sediment, and biota will have a large effect on the ecosystem (Fig. 235). Sediment burial will eventually remove these contaminants from interacting with the biota in the ecosystem but this is a long-term process. Sediment organic carbon content and sulfide levels play an important role in metal sequestration such that metals are more available for bioaccumulation, are more toxic, and are released to the overlying water more readily in low organic content sediments than in highly organic-rich media.[91,92] However, many contaminant concentrations increase with increasing organic content and thus there is a complex interaction between organic matter content and contaminant bioaccumulation from sediments, especially for trace metals. Even though government regulation may reduce contaminant inputs from point and diffuse sources such as watershed sediment loading and agricultural runoff, it is still important to understand the cycling of contaminants within the ecosystem, and particularly within the sediment, because of the potential for release back to the ecosystem, and for their bioaccumulation and toxicity to benthic organisms.

Nutrient loads and system productivity in the water column also influence the concentration and growth dynamics of primary producers and other microbial organisms, and this dictates to a large degree the contaminant concentration at the base of the food chain. Thus, contaminant fate is influenced by factors such as the degree of eutrophication, sediment resuspension, and other ecosystem disturbances. The interactions are often non-linear, and may have secondary effects and feedback interactions (Fig. 236). Thus these complexities cannot be examined in small-scale microcosms or in beakers. In addition, many contaminants are adsorbed to container walls, or by the microbial growth that often forms on the walls of experimental systems during long-term studies. Thus, small microcosms may

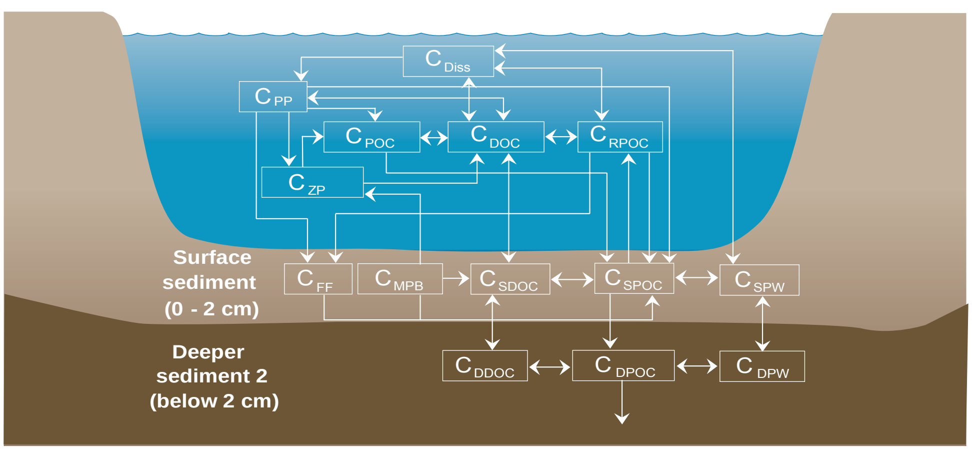

Figure 236: *Conceptual model showing the interactions and pathways for a chemical, C, cycling between the sediment and water under the influence of sediment resuspension and the interaction of the chemical with the various phases in the system. The following abbreviations are used: PP = phytoplankton; ZP = zooplankton; POC = particulate organic carbon; DOC = dissolved (colloidal) organic carbon; RPOC = resuspended particulate matter; Diss = dissolved constituents; FF = filter feeders; MPB = microphytobenthos; S = surface sediment; D = deep sediment; PW = porewater.*

89. EPA 2004, 90. Bianchi 2007, 91. Di Torro et al. 2005, 92. Mason 2002

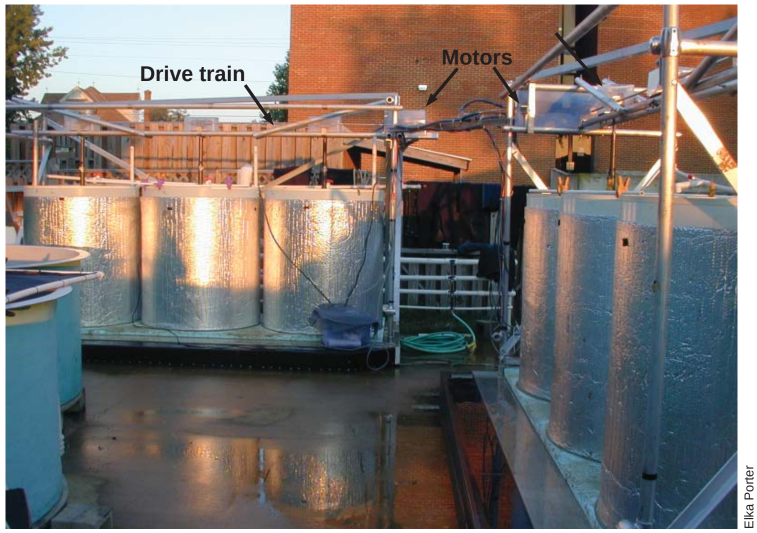

Figure 237: *The MEERC STORM (Shear, Turbulence Resuspension Mesocosm) system was used to investigate physicochemical processes, sediment-water interactions, and transfer of contaminants between media. The system was more effective at studying benthic-pelagic coupling processes than typical mesocosms because the STORM system mimicked realistically episodic and tidal resuspension with high bottom shear and water column turbulence levels.*

experience substantial wall effects associated with their use due to their large wall-surface-area-to-volume ratio. Such wall effects are more important in the absence of sediment in the experimental ecosystem. Experiments that include sediment are the most realistic for the examination of the effect of contaminants and toxicants on coastal ecosystems, and on contaminant bioaccumulation and health effects on fish consumers.

Sources of contaminants (from water, sediment, or food) to biota cannot be determined from field studies or simple in-lab exposure experiments because of the importance of feedback mechanisms and the interaction between species (Fig. 236). Recent studies have shown that mesocosms have substantial advantages for the examination of contaminant biogeochemical cycling within complex systems such as estuaries.[93,94] Benthic-pelagic coupling by physical (advection, diffusion, and resuspension), chemical (sediment redox changes), or biological (biota migration across the sediment-water interface) processes can strongly affect the rate of bioaccumulation. System growth rate and productivity influenced by nutrient loadings also influences both organism feeding rates and bioaccumulation (growth dilution effects). Finally, uptake of contaminants may be slow such that longer-term experiments need to be performed over weeks to months to ascertain clearly the uptake mechanisms and the effects of contaminants and their potential toxicity. Again, small-scale systems cannot maintain their integrity for sufficient time for many of the studies that need be done to examine bioaccumulation and trophic transfer of contaminants in aquatic food chains. Studies[95-97] have demonstrated, for example, the importance of longer term experiments for examining estuarine mercury cycling.

Overall, the real world is complex and requires detailed, long-term experiments to adequately examine all the interactions in systems with realistic physical mixing and disturbance and representative food chains. Many important questions cannot be examined through the collection of field data. Therefore, mesocoms provide a system that can be manipulated to examine the complex processes

93. Orihel et al. 2006, 94. Bromilow et al. 2006, 95. Kim 2004, 96. Kim et al. 2004, 97. Kim et al. 2006

discussed above in long-term controlled experiments. Limitations of size need to be heeded, especially if higher level food chain organisms are to be included. A typical 1 m^3 mesocosm would not be suitable for fish studies except with the smallest fish. However, the typical mesocom size is suitable for the examination of the effect of contaminants on invertebrates, both benthic and pelagic. Benthic-pelagic coupling studies have recently become more realistic by using experimental ecosystems with realistic bottom shear and water column turbulence.[98]

Research findings

Mesocosm studies using the MEERC STORM system (Fig. 237) have been used to investigate purely physicochemical processes in relatively simple systems, and to examine sediment-water interactions and the diffusive and advective transfer of contaminants between media (e.g., water and sediment).[99] In addition, they have been shown to be suitable for examination of the bioaccumulation of contaminants in a relatively complex system with substantial benthic-pelagic coupling and where there are both water-column and sediment-based primary consumers.[100-103] More specifically, they have been used to examine the role of physical disturbance, such as tidal and episodic sediment resuspension, on contaminant transport from sediment to the water column, and on bioaccumulation by biota for mercury.[101-104]

MEERC STORM systems of $1m^3$ volume were designed for mesocosm experiments that include episodic and tidal resuspension in short-term and longer-term mesocosm experiments. Episodic resuspension due to increased bottom shear can be induced by storms yet its effects on the ecosystem, the nutrient, and the contaminant cycling are difficult to assess in the field. In addition, bottom shear varies over the tidal cycle. Sediment resuspension is induced regularly during the tidal cycle when critical erosional thresholds are surpassed, such as, for example, during highest flood or ebb tides.

Bottom shear was carefully controlled in the STORM systems and set to levels of bottom shear found in the field and that induced sediment resuspension. However, at all times water column turbulence levels were kept at realistic levels and the water column was not over-mixed. Bottom shear in standard mesocosms is unrealistically low, with consequences for benthic-pelagic coupling.[98] In the STORM systems, using a special mixing apparatus, much higher bottom shear without over-mixing the water column can be induced.

Bottom shear in the STORM systems can be programmed to vary smoothly over

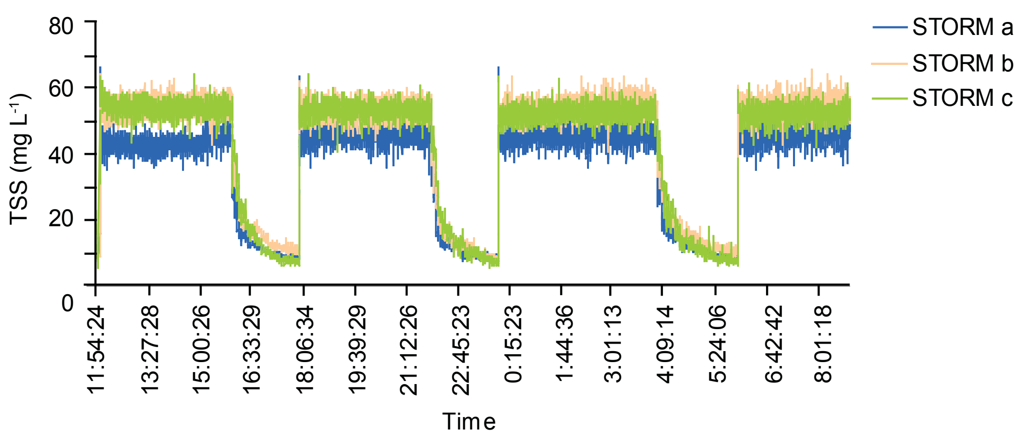

Figure 238: *1Tidal cycles in three replicate STORM tanks induce sediment resuspension during three 4h mixing_ON phases and induce particle settling during 2h mixing_OFF phases. TSS = total suspended solids.*

98. Porter et al. 2004b, 99. Schneider 2005, 100. Adapted from Kim et al. In review, 101. Kim 2004, 102. Kim et al. 2004, 103. Kim et al. 2006, 104. Bergeron 2005

any desired cycling pattern. For logistical reasons related to sampling a large number of ecosystem variables, the system mimicked tidal resuspension as 4h "mixing on" periods with high bottom shear during which sediment resuspension occurred, and a 2h mixing off phase (Fig. 238, 4h on 2h off cycling) where mixing was turned off and particle settling was allowed. This "4h resuspension_ON" and "2 h Resuspension_OFF" cycling was maintained over experiments of about 4 weeks duration (e.g., Fig. 239). Sediment resuspension was measured continuously in the STORM tanks using optical backscatter sensors (OBS-3) deployed 50 cm above the bottom and calibrated with direct total suspended solids (TSS) water samples taken during the course of the experiment.

Researchers performed four 4-week long comparative ecosystem experiments with tidal resuspension using the STORM mesocosm facility (Fig. 237). These experiments examined the effect of tidal resuspension on the ecosystem and on the contaminant and nutrient dynamics.[105,106] In addition, experiments have varied benthic fauna in comparative ecosystem experiments with the STORM systems. Finally, experiments focusing on the effect of episodic sediment resuspension of Hudson River sediments on PCB release[107] and particle dynamics were performed.

Physicochemical studies and contaminant transport studies that have been done include:

1) Examination of the partitioning of contaminants between the particulate phase and the dissolved phase, and the importance of *kinetic* (slow response time to changes in concentration) versus *equilibrium* (rapid attainment of steady state) control over the chemical distribution. These studies have shown that the notion of equilibrium partitioning between natural solids and dissolved constituents in coastal waters is not valid. [108]
2) Examination of the rate of oxidation of sediments upon resuspension and the effect of such processes on metal and other contaminant release to solution.

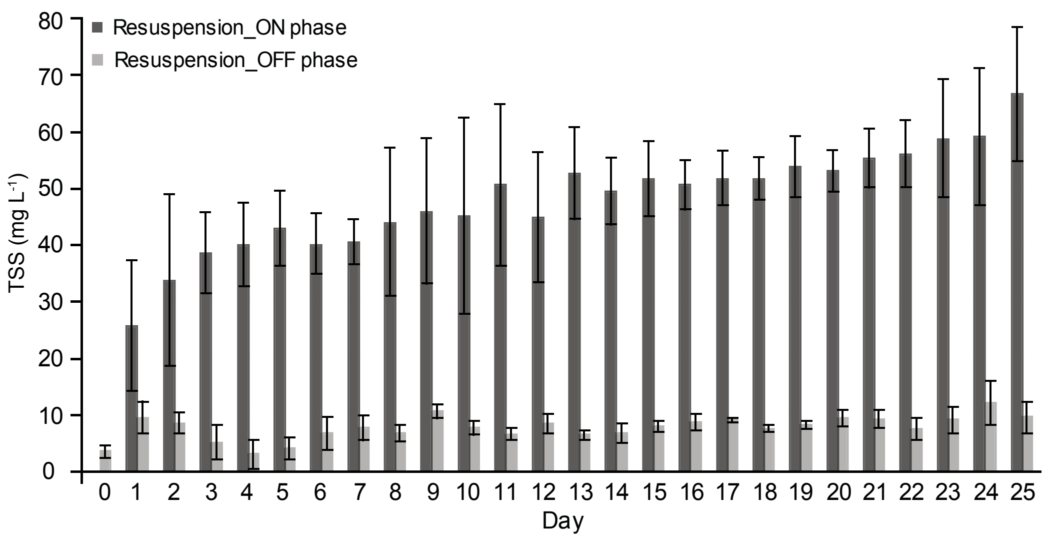

Figure 239*: Tidal resuspension (4h mixing on, 2h mixing off) maintained over a ~ 4 week period in the STORM systems during an ecosystem experiment, as measured using OBS-3 sensors. Representative data from one of the 4h mixing_on phase and from the end of a 2h mixing_off phase are shown for each day of the experiment. Four of such mixing_on and mixing_off phases were programmed for each day for the duration of the experiment. The data show average TSS levels from three STORM tanks (± standard deviations). TSS = total suspended solids.*

105. Kim et al. 2004, 106. Kim et al. 2006, 107. Schneider et al. 2007, 108. Schneider 2005

Mesocosm studies have found that the effect of resuspension is less than that obtained in smaller-scale studies where the energy for resuspension, and therefore the water column particulate load, is typically unrealistic.[109] For methylmercury, the effect of resuspension on sediment mercury methylation rate can be effectively examined using STORM mesocosms because this process is dependent on physical, chemical, and biological factors in a complex fashion.

3) Studies of the effect of tidal resuspension and other physical processes on sediment chemistry and contaminant mobility across the sediment-water interface (Fig. 240). These have shown that resuspension has less effect on water-column metal concentrations than previously thought, although the effect is different for different metals. For example, cadium, which has a relatively low affinity for the solid phase, is bioaccumulated more strongly than metals that bind strongly with sediment, such as zinc and lead.[110-112] The effect of resuspension on mercury dynamics is shown in Fig. 240.

Management implications

Mesocosm studies have shown that trace metals in particular, and by analogy other strongly-bound sediment contaminants, are not released to any significant degree by sediment resuspension. Release may occur during the initial resuspension of sediment but continual resuspension appears to result in decreased release to the water for PCB's, with the extent of release being a function of contaminant partitioning.[109,113] This notion is consistent with the idea of the contaminant being distributed in both easily available and strongly bound pools in sediment. In addition, it appears that without continual input of the chemical from external sources, the fraction that remains in the easily available form decreases over time. These results reinforce the idea that contaminants that are strongly particle reactive are not readily bioavailable to aquatic organisms, and that the legacy of contamination in sediments may be less important than first expected.

However, because organic matter is often the major binding phase for these metals, and because organic content indirectly affects sediment redox state, changes in the ecosystem that result in a decrease in sediment organic carbon could lead to an increase in the release and availability of these contaminants to the food chain.[110, 113-115] Model results extrapolating the mesocosm results to the Chesapeake Bay ecosytem show the impact of sediment chemistry and resuspension on methylmercury bioaccumulation[116,117] (Fig. 241). The

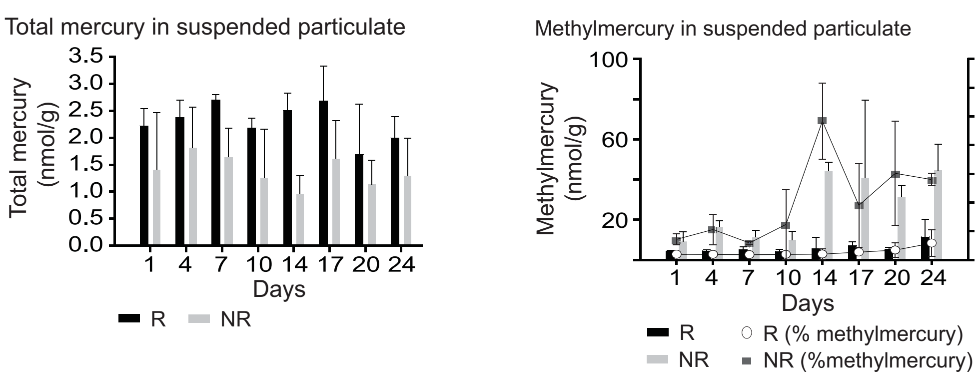

Figure 240: *Graphs showing the impact of sediment resuspension on the amount of total mercury and methylmercury in suspended particulate. While resuspension (R) increased the total mercury concentration, the percent methylmercury was higher for the non-resuspended (NR) mesocosms.*[109]

109. Kim et al. 2006, 110. Mason 2002, 111. Langston et al. 1999, 112. Schneider et al. 2007, 113. Schneider 2005, 114. Di Torro et al. 2005, 115. Bianchi 2007, 116. Kim et al. In review, 117. Kim et al. 2004

bioavailability of mercury and other metals both in the water and sediment to invertebrates and microbes is a strong inverse function of the organic content of the water or porewater. Thus, reductions in eutrophication may have a negative effect on contaminants by increasing bioavailability and bioaccumulation.

Finally, modeling studies have shown the importance of the rate of primary productivity in influencing the bioaccumulation of mercury, methylmercury, and likely other contaminants in food-limited environments.[118,119-121] The implication is that food web structure and competition for resources are important considerations that are often not examined in sufficient detail when attempting to understand contaminant fate and bioaccumulation in coastal systems.

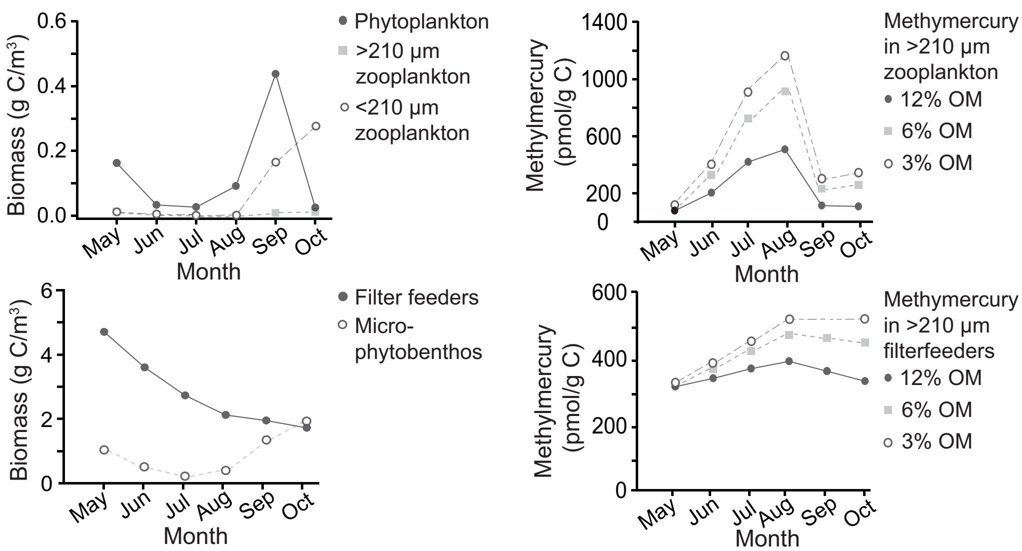

Figure 241: *Model simulation of Chesapeake Bay showing the effect of sediment organic content (OM) on the bioaccumulation of methylmercury into organisms. The model output also shows the biomass estimates for the different biota.*[120]

118. Di Torro et al. 2005, 119. Kim et al. In review, 120. Kim et al. 2004, 121. Ashley 1998

References

Anderson, D.A., P.M. Glibert, and J.M. Burkholder. 2002. Harmful algal blooms and eutrophication: Nutrient sources, composition, and consequences. Estuaries 25:562-584.

Ashley, J.F.A. 1998. Habitat use and trophic status as determinants of hydrophobic organic contaminant bioaccumulation within shallow systems. Ph.D. Dissertation, University of Maryland, College Park, 318 pp.

Bartleson, R. D., W. M. Kemp, and J. C. Stevenson. 2005. Use of a simulation model to examine effects of nutrient loading and grazing on *Potamogeton perfoliatus* L. communities in microcosms. Ecol. Model. 185: 483-512.

Berg, G.M., P.M. Glibert, and C.C. Chen. 1999. Dimension effects of enclosures on ecological processes in pelagic systems. Limnol. Oceanogr. 44: 1331-1340.

Berg, G.M., P.M. Glibert, M.W. Lomas, and M. Burford. 1997. Organic nitrogen uptake and growth by the Chrysophyte Aureococcus anophagefferens during a brown tide event. Mar. Biol. 129: 377-387

Bergeron, C.M. 2005. The impact of sediment resuspension on mercury cycling and the bioaccumulation og methylmercury into benthic and pelagic organisms. M.S. Thesis, University of Maryland, College Park, 108 pp.

Berman, T. and D.A. Bronk. 2003. Dissolved organic nitrogen: a dynamic participant in aquatic ecosystems. Aq. Microb. Ecol. 31: 279-305.

Bianchi, T.S. 2007. Biogeochemistry of Estuaries. Oxford University Press, New York, 706 pp.

Borum, J. 1985. Development of epiphytic communities on eelgrass (*Zostera marina L*.) along a nutrient gradient in a Danish estuary. Mar. Biol. 87: 211-218.

Bricker, S., B. Longstaff, W. Dennison, A. Jones, K. Boicourt, C. Wicks, and J. Woerner. 2007. Effects of Nutrient Enrichment in the Nation's Estuaries: A decade of change. NOAA Coastal Ocean Program Decision Analysis Series No. 26. National Centers for Coastal Ocean Science, Sliver Spring, MD. 322pp.

Brooks, M. T. 2004. Trophic complexity, transfer efficiency and microbial interactions in pelagic ecosystems: A modeling study. MS Thesis, Univ. of Maryland, College Park.

Bromilow, R.H., R.F. de Carvalho, A.A. Evans, and P.H. Nicholls. 2006. Behavior of Pesticides in Sediment/Water Systems in Outdoor Mesocosms. Journal of Environmental Science and Health Part B, 41:1–16.

Caddy, J.F. 1993. Towards a comparative evaluation of human impact on fishery ecosystems of enclosed and semi-enclosed seas. Rev Fish Sci 1: 57-95

Cerco, C. and K. Moore. 2001. System-wide submerged aquatic vegetation model for Chesapeake Bay. Estuaries. 24: 522-534.

Cosper, E.M., W.C. Dennison, E.J. Carpenter, V. M. Bricelj, J.G. Mitchell, S.H. Kuenstner, D. Colflesh, and M. Dewey. 1987. Recurrent and Persistent Brown Tide Blooms Perturb Coastal Marine Ecosystem. Estuaries.10(4): 284-290.

de Leiva Moreno J.I., Agostini V.N., Caddy J.F., and Carocci F. 2000. Is the pelagic-demersal ratio from fishery landings a useful proxy for nutrient availability? A preliminary data exploration for the semi-enclosed seas around Europe. ICES Journal of Marine Science. 57: 1091-1102.

Dennison W.C., R.J. Orth, K.A. Moore, J.C. Stevenson, V. Carter, S. Kollar, P.W. Bergstrom, R.A. Batiuk. 1993. Assessing Water-Quality with Submersed Aquatic Vegetation. Bioscience 43(2):86–94.

Di Torro, D.M., J.A. McGrath, D.J. Hansen, W.J. Berry, P.R. Paquin, R. Mathew, K.B. Wu, and R.C. Santore. 2005. Predicting sediment metal toxicity using a sediment biotic ligand model: Methodology and initial application. Environ. Toxicol. Chem. 24: 2410-2427.

Duarte, C. 1995. Submerged aquatic vegetation in relation to different nutrient regimes. Ophelia 41: 87-112.

Dugdale, R.C. and J.J. Goering. 1967. Uptake of new and regenerated forms of nitrogen in primary production. Liminlology and Oceanography. 12(2): 196-206.

EPA. 2000. Chesapeake Bay Program. *Chesapeake 2000.*

EPA. 2004. The incidence and severity of sediment contamination in surface waters of the United States. USEPA Office of Science and Technology, Washington DC, Report # EPA-823-R-04-007.

Gacia, E. and C. Duarte. 2001. Sediment retention by a Mediterranean Posidonia ocanica meadow: The balance between deposition and resuspension. Estuar. Coast. Shelf Sci. 52: 505-514.

Glibert, P. M. and D. G. Capone. 1993. Mineralization and assimilation in aquatic, sediment, and wetland systems. Pages 243-272 in R. Knowles and T. H. Blackburn, editors. Nitrogen Isotope Techniques. Academic Press, San Diego.

Glibert, P.M., J. Harrison, C. Heil, and S. Seitzinger. 2006. Escalating worldwide use of urea – a global change contributing to coastal eutrophication. Biogeochemistry 77:441-463.

Glibert, P.M. and C. Heil. 2005. Use of urea fertilizers and the implications for increasing harmful algal blooms in the coastal zone. Contributed papers, the 3rd International Nitrogen Conference, Science Press USA Inc.,2005, 539-544.

Glibert, P.M., C.A. Heil, D. Hollander, M. Revilla, A. Hoare, J. Alexander, and S. Murasko. 2004. Evidence for dissolved organic nitrogen and phosphorus uptake during a cyanobacterial bloom in Florida Bay. Mar. Ecol. Prog. Ser. 280: 73-83

Glibert, PM and C. Legrand. 2006. The diverse nutrient strategies of HABs: Focus on osmotrophy. pp 163-176 in: E. Graneli and J. Turner (eds), Ecology of Harmful Algae. Springer.

Glibert, P.M. S. Seitzinger, C.A. Heil, J.M. Burkholder, M.W. Parrow, L.A. Codispoti, and V. Kelly. 2005. The role of eutrophication in the global proliferation of harmful algal blooms: new perspectives and new approaches Oceanography 18 (2): 198-209

Hagy, J. D. 2002. Eutrophcation, hypoxia and trophic transfer efficiency in Chesapeake Bay. PhD Thesis, Univ. Maryland, College Park.

Hallagraeff, G. M. 1993. A review of harmful algal blooms and their apparent global increase. Phycologia 32:79-99.

Hengst, A. M. 2007. Restoration ecology of *Potamogeton perfoliatus* in mesohaline Chesapeake Bay: The nursery bed effect. MS Thesis, University of Maryland, College Park.

Hillbricht-Illkowska, A. 1977. Trophic relations and energy flow in pelagic plankton. Polish Ecological Studies. 3:3-98.

Kemp, W. M. 2000. Seagrass ecology and management: An introduction, pp. 1-8, In: S. Bortone (ed.) Seagrasses: Monitoring, ecology, physiology, and management. CRC Publ., Boca Raton, FL

Kemp, W.M., W.R. Boynton, J.C. Stevenson, R.R. Twilley and J.C. Means. 1983. The decline of submerged vascular plants in upper Chesapeake Bay: Summary of results concerning possible causes. Mar. Techn. Soc. J. 17:78-89.

Kemp, W. M., R. Batiuk, R. Bartleson, P. Bergstrom, V. Carter, G. Gallegos, W. Hunley, L. Karrh, E. Koch, J. Landwehr, K. Moore, L. Murray, M. Naylor, N. Rybicki, J. C. Stevenson, and D. Wilcox. 2004. Habitat requirements for submerged aquatic vegetation in Chesapeake Bay: Water quality, light regime, and physical-chemical factors. Estuaries 27: 363-377.

Kemp, W.M., W. Boynton, J. Adolf, D. Boesch, W. Boicourt, G. Brush, J. Cornwell, T. Fisher, P. Glibert, J. Hagy, L. Harding, E. Houde, D. Kimmel, W.D. Miller, R. I.E. Newell, M. Roman, E. Smith, and J.C. Stevenson. 2005. Eutrophication of Chesapeake Bay: Historical trends and ecological interactions. Mar. Ecol. Prog. Ser.303: 1-29.

Kemp W.M., M.T. Brooks and R.R. Hood. 2001. Nutrient enrichment, habitat variability and trophic transfer efficiency in simple models of pelagic ecosystems. Mar Ecol Prog Ser 223:73-87

Kim, E-H. 2004. The importance of physical mixing and sediment chemistry in mercury and methylmercury biogeochemical cycling and bioaccumulation within shallow estuaries. Ph.D. Dissertation, University of Maryland, College Park, 273 pp.

Kim, E.-H. R.P. Mason, and C.M. Bergeron. (In review). Modeling methylmercury bioaccumulation in an estuarine environment: An examination of the major controlling factors.

Kim, E-H., R.P. Mason, E.T. Porter and H.L. Soulen. 2004. The effect of resuspension on the fate of total mercury and methylmercury in a shallow estuarine ecosystem. Mar. Chem. 86: 121-137.

Kim, E-H., R.P. Mason, E.T. Porter, and H.L. Soulen. 2006. The impact of resuspension on sediment mercury dynamics, and methylmercury production and fate: A mesocosm study. Mar. Chem. 102: 300-315.

Kirk, J. T. O. 1994. Light and Photosynthesis in Aquatic Ecosystems. Second Edition. Cambridge University Press, Cambridge, UK. 525 pages.

Landry, M.R. 1977. A review of important concepts in the trophic organization of pelagic ecosystems. Helgolander wis Meeresunters 30: 8-17.

Langston, W.J., G.R. Burt and N.D. Pope. 1999. Bioavailability of metals in sediments of the Dogger Bank (central North Sea): A mesocosm study. Est. Coast. Shelf. Sci. 48: 519-540.

Luo, J. and S.B. Brandt. 1993. Bay anchovy *Anchoa mitchilli* production and consumption in mid-Chesapeake Bay based on a bioenergetics model and acoustic measures of fish abundance. Mar. Ecol. Prog. Ser. 98:223-236.

Orihel, D.M., M.J. Paterson, C.C. Gilmour, R.A. Bodaly, P.J. Blanchfield, H. Hintelmann, R.C. Harris, and J.W.M. Rudd. 2006. Effect of Loading Rate on the Fate of Mercury in Littoral Mesocosms. Environ. Sci. Technol. 40: 5992-6000.

Madsen, K.N., P. Nilsson, and K. Sundback. 1993. The influence of benthic microalgae on the stability of a subtidal sediment. Journal of Experimental Marine Biology Ecology. 170: 159–177.

Madden, C. J. and W. M. Kemp. 1996. Ecosystem model of an estuarine submersed plant community: Calibration and simulation of eutrophication responses. Estuaries. 19 (2B): 457-474.

Malone, T. C., H. W. Ducklow, E. R. Peele and S. Pike. 1991. Picoplankton carbon flux in Chesapeake Bay. Marine Ecology Progress Series 78:11-22.

Malone, T.C., D.J. Conley, P.M. Glibert, L.W. Harding, Jr., and K. Sellner. 1996. Scales of nutrient limited phytoplankton productivity: The Chesapeake Bay example. Estuaries.19: 371-385.

Mason, R.P. 2002. The bioaccumulation of mercury, methylmercury and other toxic elements into pelagic and benthic organisms. pp. 127-149. In: M.C. Newman, M.H. Roberts, and R.C. Hale [eds.], Coastal and Estuarine Risk Assessment, CRC/Lewis Publ.

Melton, J. H. 2002. Environmental quality and restoration of mesohaline submerged aquatic vegetation. MS Thesis, University of Maryland, College Park.

Naeem, S., J. Lindsey, P. Sharon, J.H. Lawton, and R.M. Woodfin. 1994. Declining biodiversity can alter performance of ecosystems. Nature. 368: 734-737.

Naeem, S., K. Hakansson, J.H. Lawton, M.J. Crawley, and L.J. Thompson. 1996. Biodiversity and plant productivity in a model assemblage of plant species. Oikos. 76: 259-264.

Nagel, J. 2007. Plant-sediment interactions and biogeochemical cycling for seagrass communities in Chesapeake and Florida Bays. PhD Thesis, University of Maryland, College Park.

Newell R.I.E. 1988. Ecological changes in Chesapeake Bay: Are they the result of overharvesting the American oyster, *Crassostrea virginica*? Pages 536-546 in M.P. Lynch and E.C. Krome (eds.). Understanding the estuary: Advances in Chesapeake Bay research. Chesapeake Research Consortium Publication 129 (CBP/TRS 24/88), Gloucester Point, VA.

Newell, R.I.E., J.C. Cornwell and M.S. Owens. 2002. Influence of simulated bivalve biodeposition and microphytobenthos on sediment nitrogen dynamics: A laboratory study. Limnol. Oceanogr. 47: 1367-1379.

Nixon S.W. and B.A. Buckley 2002. “A strikingly rich zone” – nutrient enrichment and secondary production in coastal marine ecosystems. Estuaries 25: 782-796

Oviatt, C. A. 1994. Biological considerations in marine enclosure experiments: Challenges and revelations. Oceanography 7:45-51.

Pauly D., V. Christensen, J. Dalsgaard, R. Froese, and F. Torres 1998. Fishing down the food chain. Science 279: 860-863

Porter E.T., J.C. Cornwell, and L.P. Sanford. 2004a. Effect of oysters *Crassostrea virginica* and bottom shear velocity on benthic-pelagic coupling and estuarine water quality. Marine Ecology Progressive Series. 271: 61-75.

Porter, E.T, L.P. Sanford, G. Gust, and F.S. Porter. 2004b. Combined water-column mixing and benthic boundary-layer flow in mesocosms: key for realistic benthic-pelagic coupling studies. Mar. Ecol. Prog. Ser. 271: 43–60.

Porter, E.T. 1999. Physical and Biological Scaling of Benthic-Pelagic Coupling in Experimental Ecosystem Studies. Ph.D. Thesis. Marine Estuarine Environmental Sciences program, University of Maryland, College Park, Maryland.
Romdhane, M. S., H. C. Eilertsen, O. K. D. Yahia and M. N. D. Yahia. 1998. Toxic dinoflagellate blooms in Tunisian lagoons: Causes and consequences for aquaculture. Pages 80–83 in B. Reguera, J. Blance, M. L. Fernandez, and T. Wyatt. (eds). Harmful Algae. Xunta de Galicia and Intergovernmental Oceanographic Commission of United Nations Educational, Scientific and Cultural Organization, Paris, France.
Schneider, A.R. 2005. PCB desorption from resuspended Hudson River sediment. Ph.D. Dissertation, University of Maryland, College Park, 211 pp.
Schneider, Porter E.T., Baker J.E. 2007. PCB release from resuspended Hudson River sediment. Environmental Science and Technology 41(4) 1097-1103.
Schulte, K. 2003. Spatial structure and heterogeneity in beds of the seagrass *Ruppia maritima* and comparison to ecological variables. MS Thesis, University of Maryland, College Park.
Short, F. T., D. Burdick, and J. E. Kaldy. 1995. Mesocosm experiments quantify the effects o eutrophication on eelgrass, *Zostera marina*. Limnology and Oceanography. 40: 740-749.
Short, F. T. and S. Wyllie-Echeverria. 1996. Natural and human-induced disturbance of seagrasses. Environmental Conservation. 23: 17-27.
Smayda, T. J. 1997. Harmful algal blooms: Their ecophysiology and general relevance to phytoplankton blooms in the sea. Limnology and Oceanography 42:1137–1153.
Stankelis, R. M., M. Naylor, and W. R. Boynton. 2003. Submerged aquatic vegetation in the mesohaline region of the Patuxent estuary: Past, present and future status. Estuaries. 26 (2A): 186-195.
Sturgis, R.B. and L. Murray. 1997. Scaling of nutrient inputs to submersed plant communities: Temporal and spatial variations. Marine Ecology-Progress Series 152:89-102.
Tilman, D. 1977. Resource competition between planktonic algae: An experimental and theoretical approach. Ecology 58:338–348.
Tomasko, D. C. Dawes, and M. O. Hall. 1996. The effects of anthropogenic nutrient enrichment on Turtle grass (*Thalassia testudinum*) in Sarasota Bay, Florida. Estuaries. 19 (2B): 448-456.
Twilley, R.R., W.M. Kemp, K.W. Staver, J.C. Stevenson and W.R. Boynton. 1985. Nutrient enrichment of estuarine submersed vascular plant communities: I. Algal growth and effects on production of plants and associated communities. Mar. Ecol. Progr. Ser. 23:179-191.
Yamamoto, T. 2003. The Seto Inland Sea—eutrophic or oligotrophic? Marine Pollution Bulletin 47: 37-42.
Zedler, J. B., and J. C. Callaway. 1999. Tracking wetland restoration: do mitigation sites follow desired trajectories? Restoration Ecology 7:69-73.