

PREDICTING CONTAMINANT EXPOSURE FROM DREDGING OPERATIONS

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ABSTRACT

Understanding the environmental risk of dredging operations requires modeling of contaminant release, fate and transport to perform exposure assessments. Contaminant releases occur from dredging, losses from the new sediment face and residuals, and dredged material disposal. The USACE ERDC has developed a suite of models in its Automated Dredging and Disposal Alternatives Modeling System (ADDAMS) and Surface-water Modeling System (SMS) to predict contaminant release, fate and transport.

Currently, ERDC is developing enhanced contaminant fate and transport features in its sediment-water interaction model (CAP/RECOVERY) to model reactive capping media, contaminant degradation, sediment bed consolidation and groundwater advection and in its Particle Tracking Model (PTM) to model contaminant/resuspended solids/water column interactions on local time and spatial scales. A module of the fate and effects of contaminants for PTM has been developed based on compatible formulations for non-conservative substances. The contaminant module incorporates basic water quality processes/kinetics, including adsorption/desorption, decay, and volatilization, as well as settling, deposition, and resuspension. Partitioning of contaminants to dissolved organic carbon and multiple particle types is being considered. Organic toxics degradation will be based on the SEAM3D model of sequential electron acceptance. Future developments will couple the CAP/RECOVERY and PTM models with ICM, a basin scale resource management tool, which will include living resources, submerged aquatic vegetation, and diagenesis to model contaminant cycling between the sediment bed and water column for prediction of long-term risk. Uncertainty methods and tools will be incorporated in the models to aid risk assessment.

Keywords: Contaminants, models, fate and transport, release, resuspension.

INTRODUCTION

Background

Dredged material or resuspended sediment plumes affect the environment in a variety of ways and on multiple temporal and spatial scales. Solids associated with plumes increase light attenuation and can transport sorbed toxicants. Oxygen-demanding substances potentially deplete the plume of oxygen. High ammonia or sulfide concentrations associated with dredged sediment can be toxic to aquatic organisms. Adsorbed and dissolved toxicants can have sub-lethal effects on organisms in the plume vicinity. Although the total release of solids, oxygen demand, and toxicants may be small from a whole-system perspective, local effects within the plume can adversely affect the habitat of valuable living resources including fish, shellfish, and aquatic vegetation.

Dredging contaminated sediments often generates concern from the public and the regulatory community about sediment resuspension and associated contaminant releases to the water column. Assessing exposure potential during the conduct of a risk assessment requires an understanding of contaminant exposure pathways as shown in Figure 1, and models for projecting the physically and biologically mediated movement of contaminants within the system. Evaluation of the environmental risk for the dredging operation requires estimates of the sediment and contaminant concentrations in the water column. Improved techniques for making these estimates will allow the consideration of operational controls and alternative dredges for reducing potential contaminant losses.

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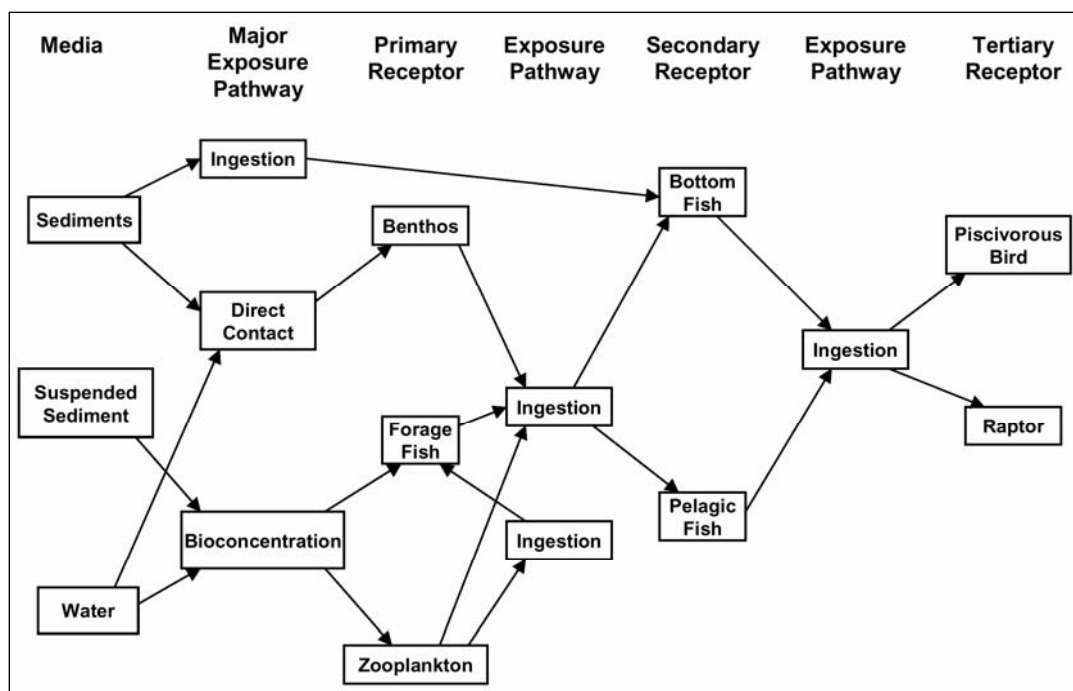


Figure 1. Contaminant pathways from source to ecological receptors.

The Corps' dredging projects also require regulatory approval and stakeholder concurrence that sediments suspended during dredging will not adversely impact the environment. The dredging community is continually faced with questions regarding the fate and effects of in-place contaminated sediments, the origins and destinations of such sediments, and the sediment related impacts of various channel deepening alternatives. Additionally, the Nation is required to manage total maximum daily loads (TMDLs). Suspended sediment fate, as well as the influence of contaminated sediments, factors into TMDL management. The range of issues presents a challenge since the processes occur across broad spatial and temporal scales. Impacts of dredged material plumes occur over shorter and smaller temporal and spatial scales, whereas the effects of contaminated in-place sediments can be manifested over longer and larger scales. Such questions can only be satisfactorily addressed through reliable and accurate predictive methods based on state-of-the-art mathematical models. A consistent, interacting set of predictive models is required that can act over spatial scales ranging from the immediate environment of a dredge to the extent of the system and over temporal scales ranging from the period of dredge operation to decades. This paper describes the USACE efforts in developing the broad set of models for evaluating environmental risk from dredging operations.

Conceptual Model

Contaminant releases occur from dredging, losses from the new sediment face and residuals, and dredged material disposal. The conceptual site model is shown in Figure 2. The primary release sources at the dredging site are resuspension of sediment particles and pore water by the dredging equipment; erosion of dredging residuals; and pore water releases, diffusion and bioturbation from residuals and cut faces. Resuspension and erosion contribute to short-term risk, while releases from residuals, contaminated deposition and cut faces also affect long-term risk. The release sources at open water disposal sites are very similar: dispersion of sediment particles and pore water from the dredged material discharge and plume; erosion of deposited dredged material; and pore water releases, diffusion and bioturbation from the consolidating deposited dredged material. Dispersion and erosion contribute to short-term risk, while releases from dredged material mound and contaminated redeposition also affect long-term risk. Potential pathways from the sources to the receptors include ingestion of the sediments or water, direct contact with sediment or water, bioconcentration from the water column, and biouptake of organisms. The receptors are humans, piscivorous birds, pelagic fish, forage fish, bottom fish, benthos, and zooplankton.

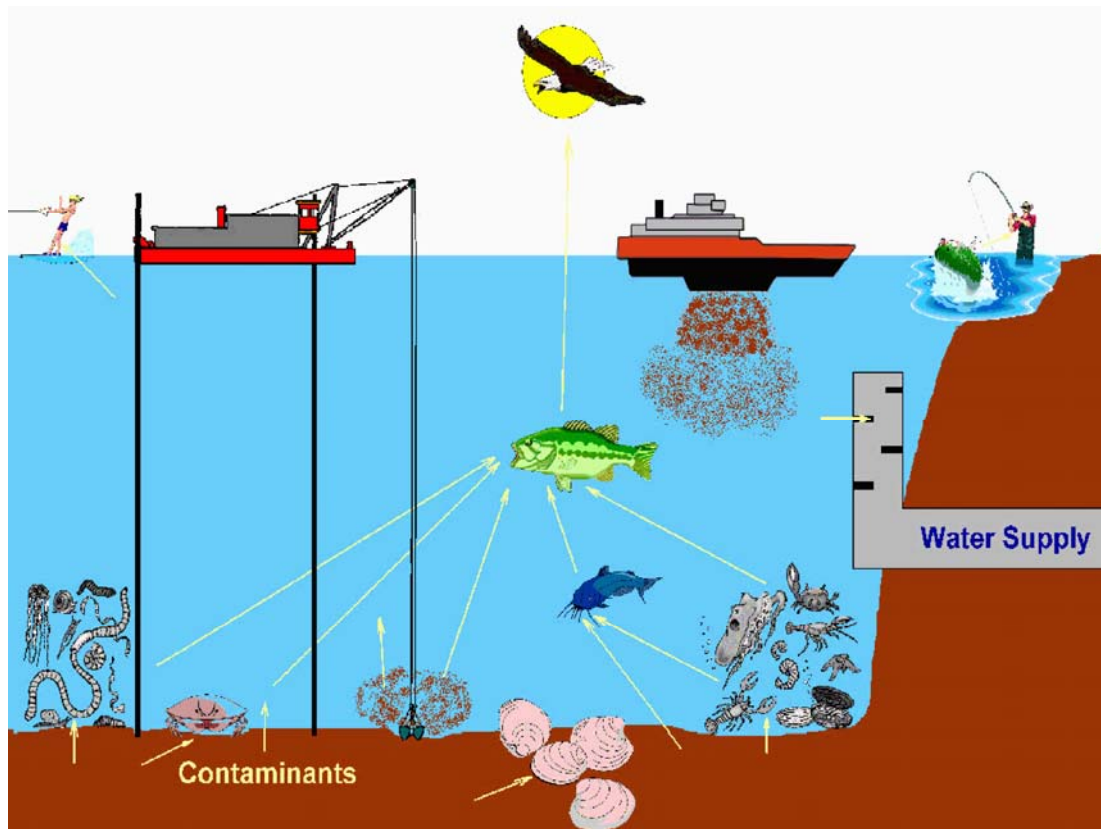


Figure 2. Conceptual site model for open-water disposal scenario.

Several types of dredges and disposal methods are employed as shown in Figure 3. Typical dredge types include hydraulic pipeline, mechanical clamshell and hopper. Dredging operations may include overflow of barges and hoppers as sources of contamination as well as resuspension by the dredge and losses from the disposal. Disposal methods include discrete dumps, multiple dumps, and pipeline discharges. During open-water disposal of dredged material, ambient water is entrained with dredged material during descent and collapse of the discharge plume, thus increasing the volume of the contaminated water released from the material that comes to rest at the bottom of the disposal site.

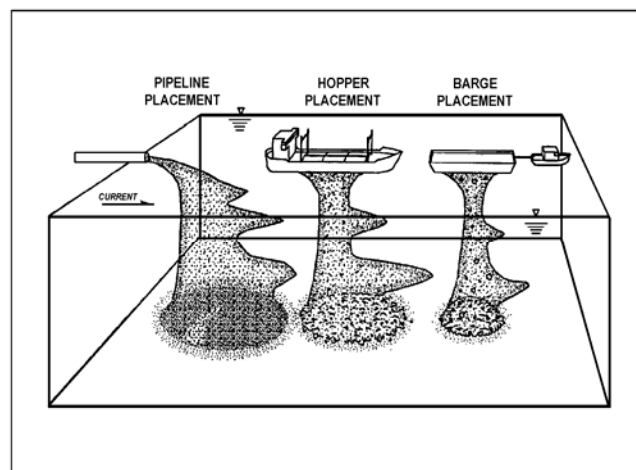


Figure 3. Open-water dredged material disposal methods.

Bucket or clamshell dredges remove the sediment being dredged at nearly its in situ density and place it on a barge or scow for transportation to the disposal area. Although several barges may be used so that the dredging is essentially continuous, disposal occurs as a series of discrete discharges. Whatever its form, the dredged material descends rapidly through the water column to the bottom, and only a small amount of the material remains suspended. After it hits bottom, most of the dredged material comes to rest. Some material enters the horizontally spreading bottom surge formed by the impact and is carried away from the impact point until the turbulence of the surge is sufficiently reduced to permit its deposition. For purposes of evaluation of initial mixing, barges or hopper dredge discharges are discrete discharges, while direct discharge from a pipeline dredge, overflow or upland placement sites should be considered continuous discharges. The STFATE model simulates discrete discharges while the CDFATE model addresses continuous discharges.

Another variation of the open-water disposal scenario is the use of contained aquatic dredged material disposal (CAD) sites as shown in Figure 4. Figure 5 shows a typical CAD pit with the different isolation layers in a cap (Ruiz et al. 2002). Dredged material not suitable for uncontrolled open-water disposal is disposed in the CAD pit. The material is then covered with a layer of clean material to isolate the potentially unsuitable material and slow the flux of contaminants to the water column. The unsuitable material is typically dredged mechanically and placed by bottom dumping from barges or by using a tremie tube. The final caps can be placed by a variety of means including bottom dumping from barges or hoppers or pumping through diffusers.

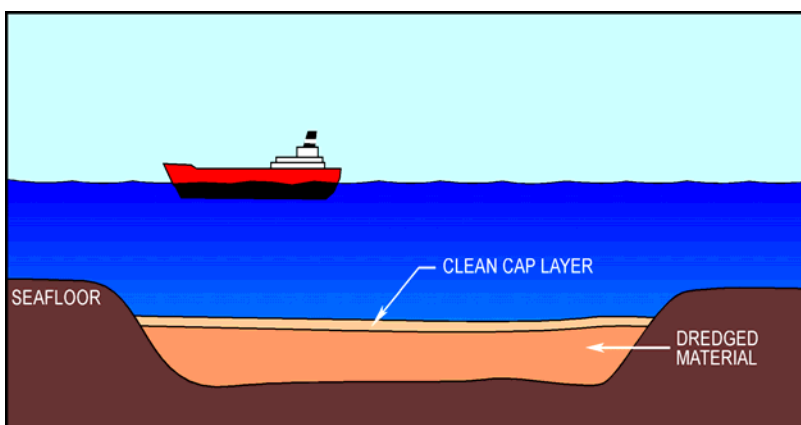


Figure 4. Capping/open-water disposal scenario.

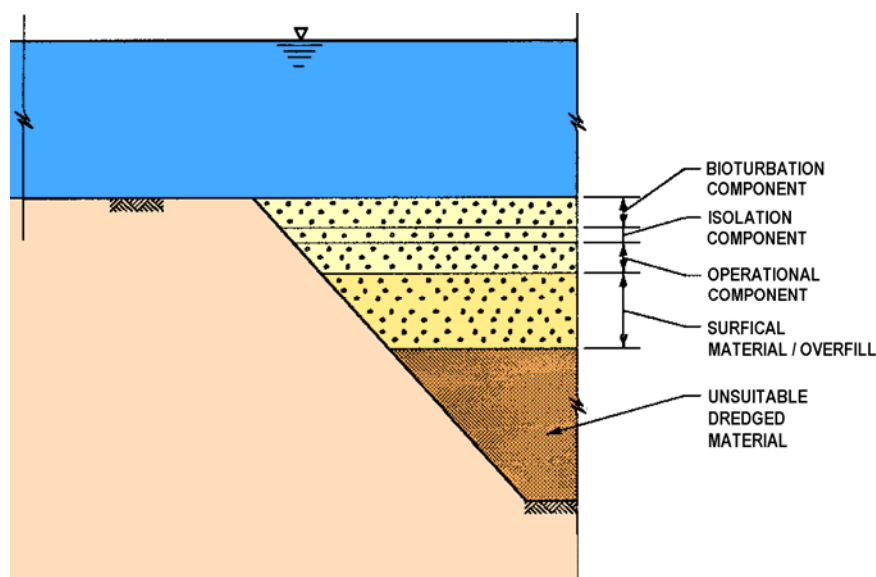


Figure 5. Typical cross section of CAD facility.

ESTABLISHED USACE MODELS AND DATABASES FOR EVALUATING DREDGING OPERATIONS

Approach

The USACE ERDC has developed a suite of models, tools and databases in its Automated Dredging and Disposal Alternatives Modeling System (ADDAMS) and Surface-water Modeling System (SMS) to predict contaminant release, fate and transport and in the Adaptive Risk Assessment Modeling System (ARAMS) to evaluate risk. ADDAMS (Schroeder et al. 2004) contains PC models of source descriptions and exposure including STFATE, CDFATE, DREDGE, and RECOVERY/CAP. SMS contains STFATE, LTFATE, and PTM. ARAMS/FRAMES (Deliman et al. 2001) contains tools for conducting human health and ecological risk assessment, including the Multimedia Environmental Pollution Assessment System (MEPAS) for conducting human health risk, the Wildlife Ecological and Assessment Program (WEAP) and the Theoretical Bioaccumulation Potential (TBP) model for conducting ecological risk, the databases Health Effects Assessment Summary Tables (HEAST) and Integrated Risk Information System (IRIS) for human health effects, and the Environmental Residue Effects Database (ERED) and Biota Sediment Accumulation Factor (BSAF) for aquatic ecological effects.

STFATE, CDFATE, and DREDGE model the short-term contaminant interactions between the suspended sediment and water column for evaluating dredging and disposal. Output from the exposure models are linked to ecological and human exposure models. These exposure models are discussed in greater detail below. The contaminant fate and transport (exposure) models simulate contaminant concentration in the water column and the bio-zone in the sediment as a function of time at the dredge site and at the disposal site. RECOVERY simulates sediment water column interactions driven by existing contaminated sediments, residuals from dredging operations (simulated by DREDGE), discrete discharges from open-water disposal (simulated by STFATE), and/or continuous discharges from a CDF or pipeline (simulated by CDFATE). In a capping scenario RECOVERY/CAP would be the model used for interactions between the cap, sediment, water column, and atmosphere for different cap configurations.

USACE Models

STFATE

STFATE (Short-Term FATE of dredged material disposed in open water) model (Johnson 1990) was developed from the DIFID (Disposal From an Instantaneous Discharge) model originally prepared by Koh and Chang (1973). The model is appropriate for instantaneous discharges from barges or scows and sequential discharges from hopper dredges. STFATE is a module of the Automated Dredging and Disposal Alternatives Modeling System (ADDAMS) (Schroeder et al. 2004).

STFATE addresses the fate and transport of solids and contaminants associated with the open-water disposal of dredged material. The contaminants are assumed to be “conservative” with no further adsorption on or desorption from the solids in the water column or deposited on the bottom. The distribution of dissolved phase to particulate phase remains constant over the period of the descent and collapse. The rationale for this assumption is that the time of the dump (release to final collapse) is very small, and thus, the solids are not going to reach a new equilibrium with the surrounding fluid.

The behavior of the dredged material during disposal is assumed to be separated into three phases: convective descent, during which the disposal cloud falls under the influence of gravity and its initial momentum is imparted by gravity; dynamic collapse, occurring when the descending cloud either impacts the bottom or arrives at a level of neutral buoyancy where descent is retarded and horizontal spreading dominates; and passive transport-dispersion, commencing when the material transport and spreading are determined more by ambient currents and turbulence than by the dynamics of the disposal operation (Johnson 1990). Figure 6 illustrates these phases.

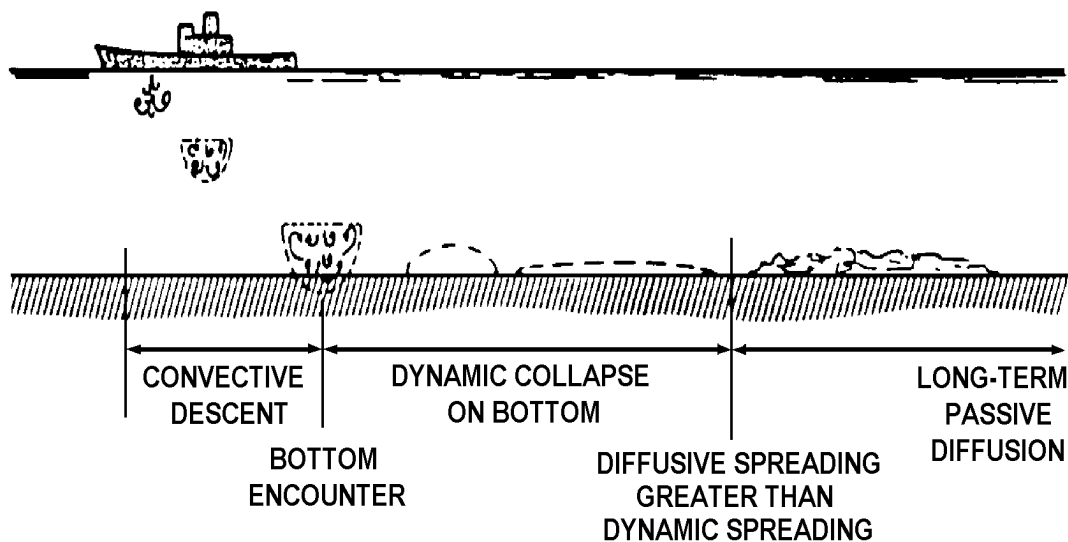


Figure 6. Illustration of open-water disposal processes.

CDFATE

The CDFATE computer program is one of the ADDAMS (Schroeder et al. 2004) modules that assist engineers, planners, and dredging operations managers in predicting the fate and effects of dredged material disposal. CDFATE (DROPMIX) is used to predict the fate and transport of dredged material discharges of a continuous nature into receiving water. These discharges may include pipeline discharges of dredged material slurries, overflows from hopper dredges and barges, and discharges of CDF effluents. Results from CDFATE typically consist of the physical limits of the mixing zone and the concentration of pollutants within the zone under steady-state conditions (Chase 1994). CDFATE is based on the EPA CORMIX model (Doneker and Jirka 1990); the Windows version of CDFATE contains the D-CORMIX model (Jirka et al. 1996). The discharge need not be toxic or highly polluted as suspended solids also degrade water quality.

When discharges into a receiving body are made, a pollutant cloud or plume will form. Because dredged material is generally heavier than water, the sediment plume will usually sink to the bottom, spread as a density flow, and then diffuse upward and outward. As the material diffuses into the water column, receiving water currents may transport the material some distance away from the original discharge point. In addition, the density of the receiving water can also influence the movement of discharged material. Finally, the initial velocity associated with the effluent itself may play a role in the final resting place for disposed sediments.

The size, location, and movement of the plume, and hence, the impact on the receiving water can be estimated using the CDFATE model. In addition, the program can be used to determine the extent of the mixing zone, that is, that area in the immediate vicinity of the discharge point where water quality parameters may exceed established criteria. Information describing the receiving water environment, the dredged material disposal method, and the mixing zone is provided to the model. The model will use this data in concert with numerical computations to generate information describing the plume location, plume geometry, and pollutant concentration within the plume as a function of time.

DREDGE

DREDGE (Hayes and Je 2000) utilizes empirical and analytical models to estimate the resuspension and transport of sediments and associated contaminants during dredging operations. DREDGE combines empirical sediment resuspension (near-field) models and simple suspended sediment transport (far-field) models to estimate suspended sediment concentrations at specified water column locations. DREDGE then utilizes a linear equilibrium partitioning model to convert initial contaminant concentrations on in situ sediment and downstream suspended

sediment concentrations to downstream water column particulate and dissolved contaminant concentrations. All calculations made by DREDGE assume steady-state time-invariant conditions. DREDGE predicts the short-term contaminant concentration distribution in the water column for determination of the acute effects from exposure to dredging and the spatial extent of the acute effects.

DREDGE utilizes empirical formulations developed from field studies to estimate the rate of sediment resuspension that results from a dredging operation (near-field source strength). DREDGE allows the user to estimate this value using Nakai's TGU method or dredge-specific correlation models. Additionally, DREDGE allows user selected source strength values to be entered for any dredge type. Nakai's TGU method can be used for most dredge types. Correlation models are available only for cutterhead and bucket dredges. There are a number of limitations associated with the models used in DREDGE. The sediment resuspension models are only applicable to dredging operations similar to those used in the development of the empirical equations. The models generally produce reasonable estimates for normal operating characteristics, but unusual operating parameters may yield unreasonable results.

The far-field transport models used assume a dominant, unidirectional current that exists sufficiently long for suspended sediment concentrations to reach steady state, assuming a steady source from a specific location and settling by Stoke's Law. Although the dredge is moving continuously, the movement is usually slow compared to transport in the water column. Transport models solved analytically for plume geometries characteristic of cutterhead and bucket dredges are used to estimate downstream (far-field) transport of suspended sediments under steady-state conditions. Considerable simplifications are necessary to solve the fundamental transport equation analytically. While these simplifications limit the applicability of the resulting models, the analytical solutions allow for rapid calculation of suspended sediment concentrations with accuracy compatible with the source strength models.

TBP

Theoretical Bioaccumulation Potential (TBP), an equilibrium partitioning based screening model, is commonly used to evaluate dredged sediments for open water disposal. Bioaccumulation is a measure used to predict exposure effects for characterizing ecological risk. The TBP model estimates the steady-state concentration of a neutral organic chemical that would ultimately accumulate in an organism from continuous exposure to contaminated sediment. TBP is calculated from chemical concentration and organic carbon content of the sediment, lipid content of the target organism, and the relative affinity of the chemical for sediment organic carbon and animal lipid. TBP is an estimate of the maximum bioaccumulation of contaminants in aquatic organisms.

The assumptions of the TBP model derive from thermodynamics. The system, consisting of sediment, organism, and water, is modeled as being closed. A neutral organic chemical in the system is given free movement and will distribute throughout the phases until equilibrium is established. The concentrations at equilibrium are determined by the chemical potentials in each phase. Organic carbon in the sediment and lipids in the organism are assumed to be the primary compartments that account for partitioning of neutral chemicals. Thus, the expected equilibrium concentration in an exposed organism of a given lipid content is a function of the concentration of a chemical in the sediment (normalized on the basis of its organic carbon content) and a partitioning coefficient between the sediment and the lipids (McFarland 1984, McFarland and Clarke 1987). The model equation is $TBP = BSAF (C_s / foc) f_L$ where the partitioning coefficient is the biota/sediment accumulation factor (BSAF), C_s is the concentration of neutral organic chemical in sediment, foc is the decimal fraction total organic carbon content of the sediment, and f_L is the decimal fraction lipid content of the target organism.

TBP was incorporated into the RECOVERY model to assess the effect of contaminated sediments on biota (Ruiz and Gerald 2001). The model uses the organic carbon (foc) of the sediments, the estimated contaminant sediment concentration (C_s), a BSAF, and biota lipid content to estimate the body burden of biota exposed to contaminated sediments. If site-specific data are not available, the BSAF database has lipid and BSAFs for a number of contaminants.

USACE Databases

BSAF

The BSAF database was constructed from numerous field and laboratory observations. Empirically derived BSAFs were calculated as $BSAF = (C_t / f_L) / (C_s / f_{oc})$ where C_t / f_L is the lipid-normalized contaminant concentration in the tissues of the exposed organism and C_s / f_{oc} is the organic carbon-normalized contaminant concentration in the sediment to which the organism has been exposed. The database contains BSAFs for contaminants of concern and lipid fractions for a number of organisms.

ERED

The U.S. Army Corps of Engineers/U.S. Environmental Protection Agency Environmental Residue-Effects Database (ERED) is a compilation of data, taken from the literature, where biological effects (e.g., reduced survival, growth, etc.) and tissue contaminant concentrations were simultaneously measured in the organism. Currently, the web-based database is limited to those instances where biological effects observed in an organism are linked to a specific contaminant within its tissues (Bridges and Lutz 1999). Currently, the system contains data from 736 studies published between 1964 and 2000. From these studies 3,463 distinct observations have been included on-line. The ERED includes data on 222 contaminants, 188 species, 13 effect classes, and 126 endpoints. Updates to the central database will occur periodically as new data sources and citations are discovered. Most papers involving mixtures of contaminants were excluded from the database because these effects could not be linked to a specific contaminant.

CURRENT DEVELOPMENTS

RECOVERY

Description

RECOVERY (version 4.3) is the U.S. Army Corps of Engineers screening-level water quality model for assessing the long-term impacts of contaminated bottom sediments on surface waters. It is an extension of previous versions of the RECOVERY model developed by Boyer et al. (1994) and has been upgraded to run on a PC in a Windows environment using a graphical user interface (GUI) written in Visual Basic. The revisions to the RECOVERY model and its GUI are being completed this year. The system is idealized as a well-mixed surface water layer underlain by a vertically-stratified sediment column of variable contamination. The discretized sediment layer configuration is useful for assessing capping scenarios and sites where contamination occurred over a long time; thus, contamination appears layered. The specification of a mixed surface layer is included because an unconsolidated layer is often observed at the surface of sediments due to a number of processes, including bioturbation and mechanical mixing.

The model couples contaminant interaction between the water column and the bottom sediment, as well as between contaminated and clean bottom sediments. The model formulations are intended primarily for organic contaminants. The contaminant is assumed to follow linear, reversible, equilibrium sorption and first-order decay kinetics. As shown in Figure 7, the system is physically represented as a well-mixed water column (i.e., zero-dimensional) underlain by a vertically-stratified sediment column (i.e., one-dimensional). The sediment is well-mixed horizontally but segmented vertically into a well-mixed surface (active) layer and deep sediment. The deep sediment is segmented into variably contaminated and clean sediment regions with varying thicknesses, porosities, and contaminant concentrations. Processes incorporated in the model are sorption, decay, volatilization, burial, resuspension, settling, bioturbation, and pore-water diffusion. The solution couples contaminant mass balance in the water column and in the mixed sediment layer along with diffusion in the deep sediment layers. The model was verified against laboratory and field data, as well as against an analytical solution for the water and mixed sediment layers (Ruiz et al. 2001, Aziz et al. 2001).

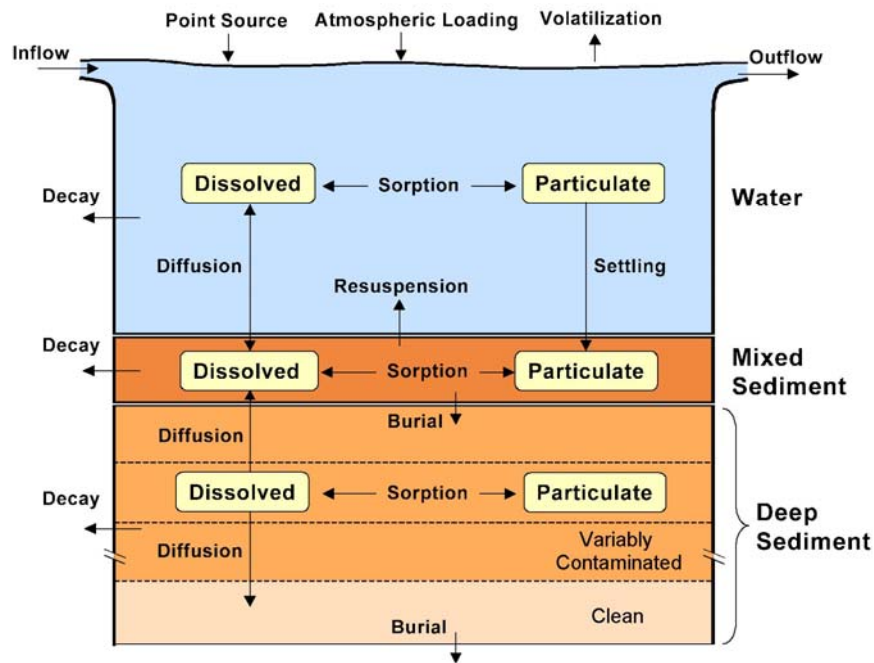


Figure 7. Schematic of the sediment-water system as modeled in RECOVERY.

These comparisons indicate that the model can be used as an assessment tool for evaluating remediation alternatives for contaminated bottom sediments. The model has been employed for numerous projects including environmental dredging (Thibodeaux and Duckworth 2001), capping design (Bailey et al. 2004), and environmental assessments (Ruiz et al. 2002).

Model Formulation

Contaminants Mass Balance. For a well-mixed water column, the mass balance for a single contaminant in the water column can be written as

$$V_w \frac{dc_w}{dt} = Q c_i - Q c_w - k_w V_w c_w - k_v V_w c_w - v_s A_w F_{pw} c_w + v_r A_m c_m + v_d A_m (F_{dp} c_m - F_{dw} c_w) + W \quad (1)$$

where V_w = volume of water body, m^3 ; c_w and c_m = concentrations of toxicant in water and mixed sediments, respectively, mg/m^3 ; c_i = inflow concentration, mg/m^3 , which reflects both direct and tributary loadings; t = time, years; Q = flushing flow rate, m^3/yr ; k_w = decay rate constant of the contaminant in the water, yr^{-1} ; k_v = volatilization rate of contaminant, yr^{-1} ; v_s = settling velocity of particulate matter, m/yr ; A_w and A_m = surface areas of water and mixed sediment, respectively, m^2 ; F_{pw} = fraction of contaminant in particulate form in the water; v_r = resuspension velocity of sediments, m/yr ; v_d = diffusion mass-transfer coefficient at the sediment-water interface, m/yr ; F_{dp} = ratio of contaminant concentration in the sediment pore water to contaminant concentration in total sediment; F_{dw} = fraction of contaminant in the dissolved form in the water; and W = external loads, kg/yr .

The model assumes that the movement of one contaminant is independent of the presence of other contaminants. The term on the left-hand side of Equation 1 represents the rate of change of contaminant mass in the water column. The first term after the equal sign represents the rate at which the contaminant is introduced into the water body. The second term represents the outflow rate of the contaminant by flushing flow. The third term is rate of contaminant decay, and the fourth term is the rate of contaminant mass volatilization. The next two terms are rate of contaminant transfer between the sediments in the water column and the mixed layer as a result of deposition and

resuspension, respectively. The next term represents the net rates of contaminant transfer by diffusion between the water column and the mixed sediment. The last term in the equation is the rate of point source input.

To complete the representation of the interaction between contaminants in the water column and the mixed sediment layer, the contaminant mass balance in the mixed sediment is used and is written as

$$V_m \frac{dc_m}{dt} = -k_m V_m c_m + v_s A_w F_{pw} c_w - v_r A_m c_m - v_b A_m c_m + v_d A_m (F_{dw} c_w - F_{dp} c_m) + v_d A_m (F_{dp} c_s(0) - F_{dp} c_m) \quad (2)$$

where V_m = volume of mixed layer, m^3 ; k_m = decay rate constant of the contaminant in the mixed layer, yr^{-1} ; v_b = burial velocity, m/yr ; and $c_s(0)$ = contaminant concentration at the top of the deep contaminated layer, mg/m^3 .

In Equation 2, the term on the left-hand side of the equation represents the rate of contaminant mass accumulation in the mixed layer. The first term after the equal sign is the decay rate of the contaminant in the mixed layer. The next three terms represent the transfer of contaminants between the mixed layer and the water column and the deep sediment by settling, resuspension and burial of sediment particles. The last two terms represent the interaction between the mixed layer and the water column and the deep sediment by diffusive transfer.

Coupling between the contaminants in the mixed sediment layer and the deep sediment takes place by diffusive transfer as well as by the burial of contaminated particles from the mixed sediment layer into the deep layer. Both the deep contaminated and clean sediments can be modeled with one-dimensional advection-diffusion-decay equations of the form

$$\frac{\partial c_s}{\partial t} = \phi F_{dp} D_s \frac{\partial^2 c_s}{\partial z^2} - v_b \frac{\partial c_s}{\partial z} - k_s c_s \quad (3)$$

where c_s = contaminant concentration in the deep sediments, mg/m^3 , ϕ = sediment porosity; D_s = diffusion rate in the sediment pore water, m^2/yr ; z = depth into the sediment, m , where $z = 0$ at the top of the deep sediments; and k_s = decay rate constant of the contaminant in the deep sediments, yr^{-1} . The term on the left-hand side of the equation is the rate of contaminant concentration change in the sediment. The first term after the equal sign is the rate of contaminant transfer by diffusion in the sediment pore water. The second term represents the rate of contaminant transfer due to sediment particle burial, and the last term represents the decay rate of contaminants in the pore water of the deep sediment.

Reactions. The decay rate constants, k_w , k_m and k_s , represent all mechanisms for decay except volatilization, which is accounted for separately. These mechanisms include photolysis, hydrolysis, and bacterial degradation.

Solids Mass Balance. In addition to the contaminant mass, the sediment mass must be conserved. The velocity terms v_s , v_r and v_b in Equations 1 and 2 are computed according to a steady-state mass balance for mixed sediment layer solids as described by Chapra and Reckhow (1983). The mass balance is

$$0 = v_s A_w s_w - (v_r + v_b) A_m (1 - \phi) \rho_p \quad (4)$$

where ρ_p is the density of the sediment solids, gm/m^3 . The framework assumes that suspended solids concentration, s_w , is given. Therefore, if two of the three velocities are specified, the equation can be employed to calculate the third.

In addition to these transfer velocities, the other facet of the physical system considered in the model is the sediment porosity. Different porosities may occur in the mixed layer and the deep sediments. Within the layers of the deep sediment region, porosity can vary from layer to layer, but is assumed constant over time. This means that compaction is not included. The same argument is assumed for particle density and partitioning.

Partitioning. The contaminant partitioning coefficients can be specified directly by the user or computed using a default relationship for hydrophobic organic contaminants (Karickhoff et al. 1979). The model allows different partitioning coefficients to be employed in the mixed layer and the vertically segmented sediments. RECOVERY allows for different organic carbon contents (f_{oc}) to be specified for the water column, mixed layer, and the deep sediments. Analogous to other physico-chemical characteristics of the sediments, the f_{oc} can vary with depth (layers) in the deep sediments.

In addition to organic contaminants the user can address the impact of contaminated sediments with heavy metals with the user defined partition coefficient. RECOVERY uses the leachable metal equilibrium concept to address the partition between the aqueous phase metal concentration and the solid phase concentration (ERM Hong Kong 1996, Palermo et al. 1993). For a single metal, the distribution or partition coefficient is

$$K_d = \frac{C_s}{C} \quad (5)$$

where K_d = equilibrium distribution coefficient; C_s = leachable metal solid phase concentration; and C = aqueous phase metal concentration.

Bioturbation Formulations. To address the impact of bioturbation in natural systems, two mechanisms are included in RECOVERY to simulate the process (see Figure 8). Particle mixing is simulated by the use of the mixed layer, and bio-pumping or enhanced biological mixing of pore water is simulated by biodiffusion. Biodiffusion is implemented by a depth of enhanced transport and an enhanced mass transfer coefficient (typically about an order of magnitude greater than the molecular diffusion coefficient, Berner 1980 and Drake et al. 1994).

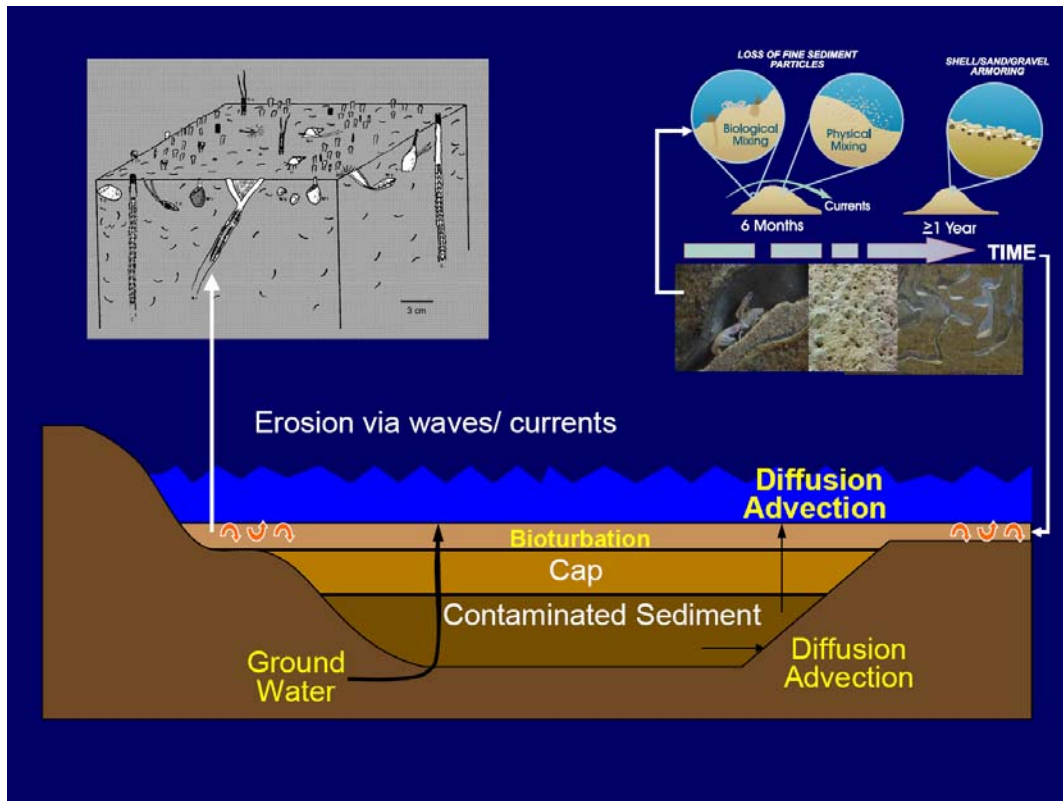


Figure 8. Schematic of surficial mixing processes.

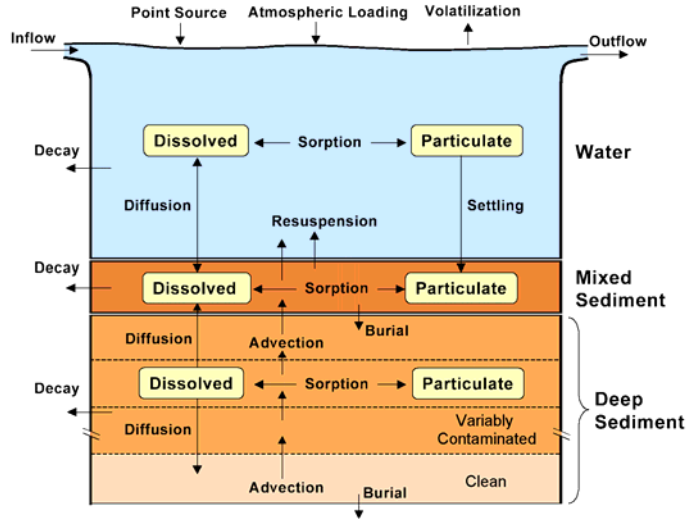


Figure 9. Schematic of CAP model processes with the incorporation of advection.

CAP Model

Description

The Capping Analysis Program (CAP) was developed for the New York Dredged Material Management Plan CAD facility evaluations. The model is an extension of frameworks developed previously (Ruiz and Gerald 2001, Boyer et al. 1994, Chapra 1982, 1986, and Chapra and Reckhow 1983). As in the RECOVERY model (the basis for the CAP model), the system is idealized as a well-mixed surface water layer underlain by a vertically stratified sediment column that allows for groundwater advection and sediment consolidation (Figure 9).

The specification of a mixed surface layer is included because an unconsolidated layer is often observed at the surface of sediments due to a number of processes, including bioturbation and mechanical mixing. The contaminant is assumed to follow linear, reversible, equilibrium sorption and first-order decay kinetics. Pathways incorporated in the CAP model, in addition to sorption and decay, are volatilization, burial, resuspension, settling, advection, and pore-water diffusion. The advection represents groundwater flow through the sediment profile or expulsion of pore water due to consolidation of the sediment profile. The temporal and spatial advection rates are specified as input. The consolidation induced rates as well as porosities and thicknesses are generated from the predictions of the Primary Consolidation, Secondary Compression, and Desiccation of Dredged Fill (PSDDF) model (Stark 1996).

Model Formulation

Contaminants Mass Balance. A mass balance for the contaminant in the water and mixed sediment layer can be solved analogous to the RECOVERY model. The deep contaminated and clean sediments can be modeled with one-dimensional advection-diffusion-decay equations of the form

$$\frac{\partial c_s}{\partial t} = \frac{\partial}{\partial z} \left(\phi F_{dp} D_s \frac{\partial c_s}{\partial z} \right) - \frac{\partial}{\partial z} (\phi F_{dp} c_s v_g) - \frac{\partial}{\partial z} (v_b c_s) - k_s c_s \quad (6)$$

where c_s = contaminant concentration in the sediments, $\mu\text{g}/\text{m}^3$; ϕ = sediment porosity; D_s = diffusion rate in the sediment pore water, m^2/yr ; v_g = advective velocity due to consolidation and groundwater flow, m/yr ; z = depth in the sediment layer where $z = 0$ at the top of the deep sediments, m ; k_s = decay rate constant of the contaminant in the deep sediments, $1/\text{yr}$; t = time, years; v_b = burial velocity of sediments, m/yr ; F_{dp} = ratio of contaminant concentration in the sediment pore water to contaminant concentration in total sediment.

The term on the left-hand side of the equation is the rate of contaminant concentration change in the sediment. The first term after the equal sign is the rate of contaminant transfer by diffusion in the sediment pore water. The second term represents the rate of contaminant transfer due to advection of the sediment pore water. The third term represents the rate of contaminant transfer due to sediment particle burial, and the last term represents the decay rate of contaminants in the pore water of the deep sediment.

Reactions and Reactive Capping. In addition to the first order decay (rates) implemented in RECOVERY, a Reaction Package option is being developed over the next 18 months for future use in the CAP model that includes aerobic/anaerobic biodegradation for the oxidation of organic carbon that follows the conceptual model by Jorgensen (1989) as illustrated in Figure 10, a cometabolism package, and a reductive dechlorination package. This reaction package will be implemented by coupling CAP with the SEAM3D model.

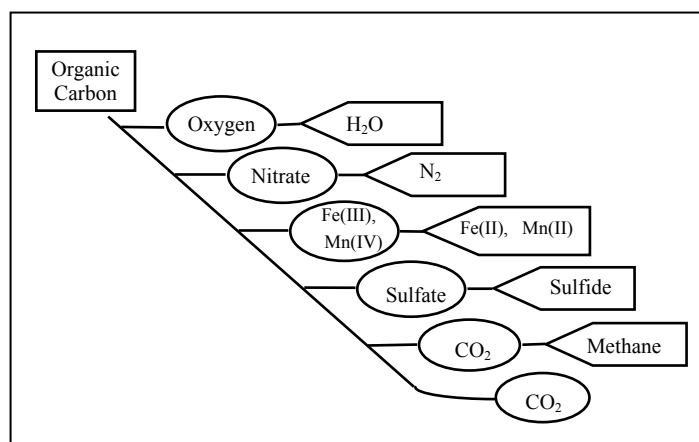


Figure 10. Conceptual model for oxidation of organic carbon.

SEAM3D (Sequential Electron Acceptor Model, 3D transport) is a code designed to simulate the spatial distribution of aquifer redox conditions resulting from the introduction or depletion of an electron donor/carbon source and the subsequent transport and biodegradation of contaminants (Widdowson et al. 1997, Waddill and Widdowson 1998, Waddill and Widdowson 2000). The SEAM3D Biodegradation Package simulates the complete range of terminal electron accepting processes (TEAP) (oxygen-, nitrate-, Fe(III)-, Mn(IV)- sulfate-reduction and methanogenesis) subject to electron acceptor availability. In SEAM3D, the TEAP distribution is solved simultaneously with the chlorinated ethane transport equations so that the rate of each biotransformation process depends on the model-simulated redox condition in each cell of the model grid (Widdowson and Waddill 1998, Widdowson 2004). The rate and extent of biodegradation of natural organic matter (diagenesis) and COC (e.g., PCB congeners) in the sediment would then be linked to the prevailing redox condition and availability of oxygen (in the case of direct oxidation) or hydrogen (in the case of reductive dechlorination). The CAP model with the SEAM3D reaction package will provide a computational tool based on first principles for (1) simulating redox conditions, sediment diagenesis and contaminant biotransformations in capped sediments using a mass and energy balance approach, and (2) quantifying multi-component contaminant mass flux under various capping and amended scenarios.

Particle Tracking Model (PTM)

Description

US Army Corps of Engineers have been developing the Particle Tracking Model (PTM) to aid in evaluation of environmental windows and impacts of dredging operations on environmental resources from particulates (MacDonald and Davies 2005). Being an object-oriented model, PTM is to be interfaced with SMS (Surface Modeling System) which provides a GUI to many other hydrodynamic models such as ADCIRC, CH3D, and RMA10, to name a few, and wave models such as STWAVE. This would enable PTM to utilize many up-to-date hydrodynamics and wave models once they are put in SMS independently of PTM. Model development effort can

be focused on dredged material fate dynamics instead of diverting efforts to work on interface to other hydrodynamic and wave models.

The PTM employs both Eulerian and Lagrangian frameworks. Eulerian calculations are carried out over the entire domain and, thus, are dependent on the grid sizes. Eulerian calculations are made mostly at the sediment-water interface through bedform, bed shear and mobility, transport potential, and transport rates. In Lagrangian framework, the waterborne constituent being modeled is represented as a finite number of discrete particles that are tracked as they are transported by the flow. Major advantages of a Lagrangian approach over the traditional Eulerian approach are computational efficiency and visualization of the plume transport. The Lagrangian calculations include local flows, mobility of a particle, and trajectory calculations. Thus the Lagrangian approaches provide diffusion and advection processes accurately and efficiently and calculates particle pathways so that sources and destinations of particles are easily identified.

Dredge plumes affect the environment in a variety of ways and on multiple temporal and spatial scales. Solids associated with plumes increase light attenuation and can transport sorbed toxicants. To date, the PTM was developed to predict transport of particles, not contaminants or dissolved constituents. Oxygen-demanding substances potentially deplete the plume of oxygen. High ammonia or sulfide concentrations associated with dredged sediment can be toxic to aquatic organisms. Adsorbed and dissolved toxicants can have sub-lethal effects on organisms in the plume vicinity. Although the total release of solids, oxygen demand, and toxicants may be small from a whole-system perspective, local effects within the plume can adversely affect the habitat of valuable living resources including fish, shellfish, and aquatic vegetation. Therefore, the modeling of contaminant fate and transport is being added to PTM over the next 18 months.

Model Formulation

Typically, the water quality model is solved on Eulerian framework. A set of partial differential equations (PDE) describes the advective and diffusive transport and reaction kinetics of a water quality in a fluid. Advection-dominated processes such as dredge plume transport are difficult to resolve with such a scheme. Numerically induced oscillation of concentration does not guarantee positivity and a point source will immediately attain the size of the grid cell containing the point source. Diffusion processes are not also accurately described by this approach using eddy diffusion. A semi-Lagrangian approach can mitigate the flattening of the concentration profile (numerical diffusion) and the concentration oscillation by calculating the concentration at each fixed point in a spatial grid over a time step. However, in such an approach, mass conservation is not strictly imposed. Mass conservation is the key in any water quality model. The problem with fully-Lagrangian approach is the computational cost, notwithstanding the complicated kinetics calculations. In air quality model, a few models have been developed as providing high accuracy, positivity sign preservation, and mass conservation (Stohl et al. 2005). We have been adopting this trajectory-grid approach to build a water quality model for dredging plumes. Lagrangian transport is fully dependent on PTM which provides the trajectory. The trajectory describes the advection of a water quality parameter accurately according to a given mean velocity field. PTM also determines the diffusive part of the particle trajectories. A coupling with kinetics will provide a complete Lagrangian water quality model.

To take advantage of PTM to determine Lagrangian particle motions, we assume that (1) the particle is actually a small water parcel which contains the mass of its chemical constituents, and (2) all parcels occupying the same grid cell partake in kinetics within the cell. The first assumption removes the need of defining a different set of particles for different water quality variables. Instead each parcel is tagged with a given set of masses of different constituents. The concentration of a given constituent in a grid cell is represented by sum of the given constituent masses of all parcels in the grid cell. In the second assumption, kinetics reactions in a grid cell determine resulting concentration which is redistributed among parcels. Then, the transport for the next time step follows and repeats the kinetics.

PTM estimates diffusion through turbulent diffusion coefficient, E_t ,

$$E_t = K_{E_t} d_w u_n^* \quad (7)$$

Here, K_{Et} is an empirical coefficient which relates E_t to water depth, d_w , and shear velocity, u^* . In PTM, Lagrangian motion of a particle sets the trajectory after half time step

$$x_{n+1/2} = x_n + \frac{u_{A,n} dt + u_{D,n} dt}{2} \quad (8)$$

And after one time step

$$x_{n+1} = x_n + \frac{u_{A,n+1/2} dt + u_{D,n+1/2} dt}{2} \quad (9)$$

Here, u_A and u_D represent advective and diffusive transport velocities, respectively. n represents previous time step so that $n+1/2$ is the time half time step from previous time. Diffusive transport velocity, u_D , is determined from

$$u_D = \Pi \sqrt{\frac{6E_t}{dt}} \quad (10)$$

Here, Π is a random number between -1 and 1.

Mass balance equation for advection dominant process is

$$\frac{dC}{dt} = \frac{\partial C}{\partial t} + \vec{V} \cdot \nabla C = -(\nabla \cdot \vec{V})C \quad (11)$$

Here S is a source term.

Numerical solution follows the PTM time-split scheme. At half time step, concentration is updated by

$$C(x_{n+1/2}, n+1/2) = \left[1 - \frac{dt(\nabla \cdot \vec{V})_n}{2} \right] C(x_n, n) + S_n \quad (12)$$

At the next time step, it becomes

$$C(x_{n+1}, n+1) = \left[1 - \frac{dt(\nabla \cdot \vec{V})_{n+1/2}}{2} \right] C(x_{n+1/2}, n+1/2) + S_{n+1/2} \quad (13)$$

We can implement the above approach in two different ways. One way is to combine the first-order direct Eulerian computation of kinetics along the time steps on the Lagrangian trajectory, and the other way is to impose a kinetics grid on the modeling domain so that reactions (i.e., kinetics) take place only within each grid cell similar to the Eulerian model. Every half time step (or even every time step with Lagrangian calculations), kinetics are solved on grid cells and put back to the Lagrangian parcels. Chock and Winkler (1994) devised the following schemes in their trajectory-grid air quality model:

1. At time t , the masses of a given species in all parcels of a given tag in the same grid cell are summed and divided by the partial volume of the grid cell which is proportional to the given tagged parcels in the cell. The resulting concentration from all species partake in kinetics calculation for a duration of a time step, Δt .
2. The new species concentrations are then converted to masses and distributed back to the particles of the same tag
3. Steps 1 and 2 are carried out for parcels of different tags. The new masses are then transported for a time step to new positions and corresponding grid cell and yield the concentration of the species at the fixed grid cell at $t+\Delta t$.
4. Steps 1, 2, and 3 are repeated to calculate new concentration at $t+2\Delta t$.

Figure 11 shows the schematics of contaminant transport module in PTM. The transport module will inherit PTM particle structure which consists of trajectory and particle characteristics such as grain size, density, and settling velocity. The particle structure also includes fate of the particle, i.e., whether deposited or suspended. Contaminant module will inherit PTM particle structure and add a few kinetics parameters such as speed of reaction and partitioning coefficient (Figure 12). As a start, we assign two types of parcels—particulate and dissolved.

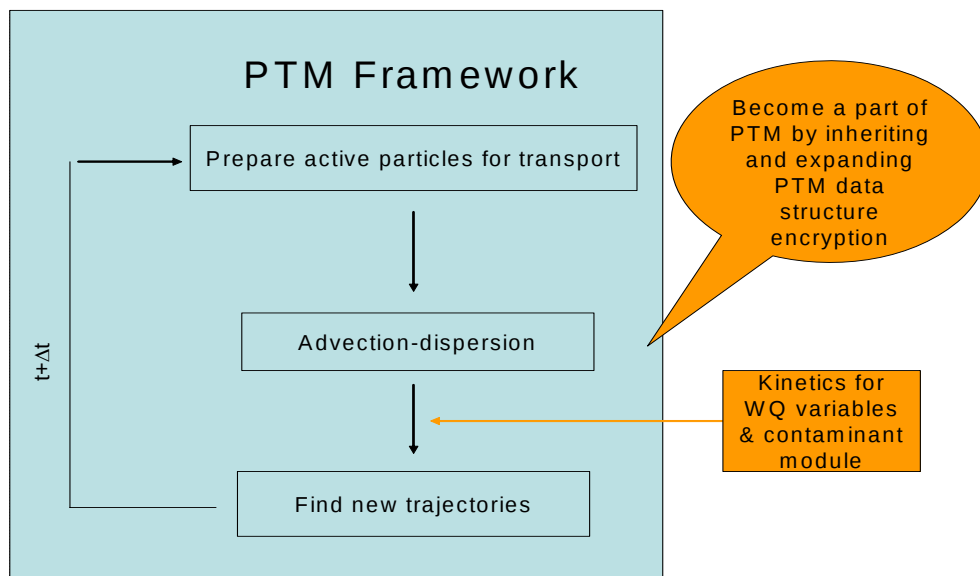


Figure 11. Schematics of contaminant transport module in PTM.

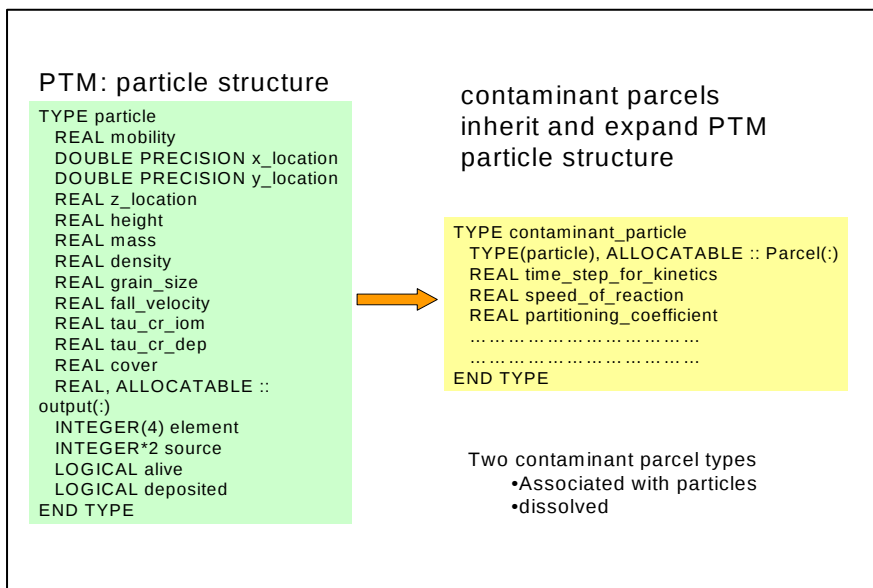


Figure 12. Modification of PTM particle structure for contaminant module.

FUTURE DEVELOPMENTS

Future developments planned to be completed during the next four years will couple the CAP/RECOVERY and PTM models with ICM, a basin scale resource management tool, which includes living resources, submerged aquatic vegetation, and diagenesis to model contaminant cycling between the sediment bed and water column for prediction of long-term risk as shown in Figure 13. The multi-dimensional, finite volume water quality model CE-QUAL-ICM (ICM) has a full suite of natural water quality state variables (Cерco and Cole 1995) and has been used extensively for Chesapeake Bay and many other aquatic systems. After coupling, the ICM model will include toxic chemical fate and transport coupled with sediment transport. This extension will provide dissolved and particulate chemical concentrations throughout the water column and at multiple depths in the sediment bed throughout the domain. Availability of these concentrations is the key to being able to perform exposure and risk assessment. Sediment transport algorithms will be adapted from the SEDZLJ sediment transport model, which is an extension of SEDZL. SEDZL model has been applied for study of contaminant cohesive sediments in many lakes, reservoirs, and coastal seas (Ziegler and Nisbet 1995).

This new version of ICM will be unique as it will have the capability to simulate the generation and fate of organic matter, which can affect contaminant mobility and exposure. The coupling of local transient contaminant sources and releases from dredging operations to the ICM model will permit estimation of bioaccumulation and ecological risk at the basin scale. The ICM model is being incorporated into the Surface water Modeling System (SMS), which provides a full suite of hydrodynamic, grid generation, data management, and visualization tools, allowing flexibility and ease of modeling for the simplest to most complex settings. The SMS contains all of the latest set of dredging tools and models that can provide contaminant sources for PTM and ICM. The SMS integration will also include development of a user guide for applying the modified PTM within SMS.

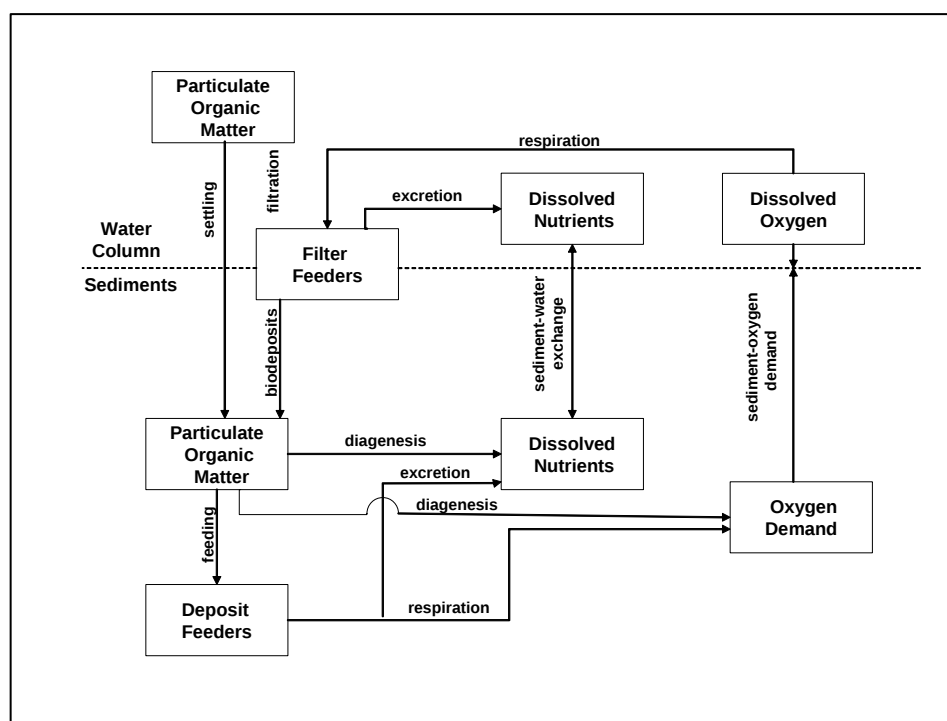


Figure 13. Schematic of CE-QUAL-ICM with sediment diagenesis.

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