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Data Needs to Reliably Hindcast a Spill's Impact: The PEPCO Pipeline Spill of April 2000 as Case Example

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Abstract

The April 2000 PEPCO pipeline spill of 525 m^3 (138,600 gal) of a combination of No. 2 and No. 6 fuels into Swanson Creek at Chalk Point, MD, was modeled with SIMAP to recreate the movement and behavior of the oil, and evaluate the model-predicted impact to shorelines and birds. As there were considerable field-based data collection efforts performed evaluating oiling of shorelines and birds injured, and the spill was in a river near populated areas with good access and logistics such that the field data were more accurate than in many cases, the case history provides an opportunity to develop improved model input assumptions. Shoreline impacts were compared to the field-based data to calibrate the physical fates model, varying the most uncertain model inputs: horizontal dispersion rate, shore holding capacity, and wind drift speed and angle. Pre-spill abundance data based on field observations during the spill greatly improved the bird impact estimates, as opposed to using seasonal mean densities for the upper Chesapeake Bay region. The results and analysis indicate appropriate model input assumptions and the primary data collection needs for spill cases where modeling might be used to assess impacts.

1 Introduction

Applied Science Associates' SIMAP oil fates and biological effects model (French McCay, 2003, 2004) was used to simulate the PEPCO oil spill of 7 April 2000 into Swanson Creek, a tributary of the Patuxent River, MD. The objectives were to provide (1) an assessment of the trajectory and fate of the oil, and thus estimate exposure to the water surface, shoreline and other habitats; (2) an estimate of impacts to birds; (3) a sensitivity analysis to determine and evaluate the most influential input assumptions; (4) an analysis of primary data collection needs for modeling such spills in the future; (5) an estimate of natural resource damages for the spill if purely habitat restoration were used for compensation of bird injuries (for use in a cost analysis); and (6) a base case for evaluating efficacy and costs of various mechanical containment and recovery-based response strategies for this and other inland waterway spills. Comparison of impacts and costs of the spill under alternative response scenarios are described in the companion paper in these proceedings by Etkin et al. (2006). The focus of this paper is on the base case, i.e., the spill with the actual response as it occurred.

When simulating a spill in an estuarine system, such as for this case, current data are best generated using a hydrodynamic model, and other environmental variables are typically easily measured or estimated from literature publications (i.e., winds, temperature, salinity, suspended sediment concentration). The horizontal and vertical diffusion (randomized mixing) coefficients are typically not measured in the field (such as with a dye study, for example), and so were varied within typical ranges for similar environments and conditions. In addition, the wind drift speed and angle for floating oil was varied among a range observed in previous spills.

Another uncertainty was the oil retention (holding) capacity of shorelines, which in the model varies by shore type and oil viscosity range. A range of oil banding widths combined with oil thickness values (and so maximum holding capacity volumes of oil) was examined in the sensitivity analysis.

The chronology of and response to the spill are described in NOAA et al., 2002; NTSB, 2001; and US EPA, 2000. Federal and state trustees, with the cooperation of the responsible party, performed a Natural Resource Damage Assessment (NRDA) for the spill. The spill resulted in lost recreational use, damage to wetlands and beach shorelines, and injuries to birds, fish and shellfish, diamondback terrapins, and benthic (mostly intertidal) communities. In the present modeling exercise, we have analyzed impacts to wetlands and shorelines, as well as bird injuries, as indices for comparison with alternate response scenarios. We have not attempted to model impacts to mammals, reptiles, subtidal fish and invertebrates, or intertidal biota (plants, fish and invertebrates) as part of this analysis.

Based on the settlement, restoration projects for the biological injuries include:

- creating a 6-acre marsh combined with a beach enhancement project;
- creating and seeding an oyster sanctuary; and
- acquiring, restoring, and maintaining a nesting habitat for ruddy ducks in the Prairie Pothole Region of the midwestern United States.

Other restoration projects compensated for the lost recreational use of resources due to the spill. For the purposes of the analysis here, we have simplified the NRDA costs to only include restoration costs for bird injuries based on a compensatory scale of habitat restoration. We used the habitat restoration model developed in French McCay and Rowe (2003) for the calculations. This analysis was applied to the base case of the actual response (described here) and the alternative response scenarios to provide estimates of NRDA costs included in the cost-benefit analysis (described in Etkin et al., 2006).

2 Model Description

SIMAP contains physical fate and biological effects models, which estimate exposure and impact on each habitat and species (or species group) in the area of the spill. The physical fate model uses wind data, current data, and transport and weathering algorithms to calculate the mass of oil components in various environmental compartments (water surface, shoreline, water column, atmosphere, sediments, etc.), oil pathway over time (trajectory), surface oil distribution, and concentrations of the oil components in water and sediments. The biological effects model simulates movements of organisms, their exposure to oil, acute toxic effects of that exposure, and population-level impacts of the lost individuals. A hydrodynamic model is used to calculate currents that transport oil components and organisms. A tactical response model allows the user to simulate booming, mechanical cleanup, burning, and dispersant usage. Environmental, geographical, physical-chemical, and biological databases supply required information to the model for computation of fates and effects. SIMAP has been validated with more than 20 case histories, including the Exxon Valdez and other large spills (French McCay, 2003, 2004; French McCay and Rowe, 2004), as well as test spills designed to verify the model's transport algorithms (French et al., 1997).

SIMAP was derived from the physical fates and biological effects submodels in the Natural Resource Damage Assessment Model for Coastal and Marine Environments (NRDAM/CME), which were developed for the US Department of the Interior (USDOI) as the basis of Comprehensive Environmental Response, Compensation and Liability Act of 1980 (CERCLA) NRDA regulations for Type A assessments (French et al., 1996). The technical documentation for the model is in French McCay (2003, 2004). Below is a brief summary of the model algorithms and assumptions.

2.1 Physical Fates Model

The physical fate model estimates the distribution of oil (as mass and concentrations) on the water surface, on shorelines, in the water column and in the sediments. Processes simulated include slick spreading, evaporation of volatiles from surface oil, transport on the water surface and in the water column, randomized dispersion, emulsification, entrainment of oil as droplets into the water column, resurfacing of larger droplets, dissolution of soluble components, volatilization from the water column, partitioning, sedimentation, stranding on shorelines, and degradation. Oil mass is tracked separately for lower-molecular-weight aromatics (1 to 3-ring aromatics), which are soluble and cause toxicity to aquatic organisms (French McCay, 2002), other volatiles, and non-volatiles. The lower molecular weight aromatics dissolve from both from the surface oil slick and whole oil droplets in the water column, and they are partitioned in the water column and sediments according to equilibrium partitioning theory (French et al., 1996; French McCay, 2003, 2004).

"Whole" oil (containing non-volatiles and volatile components not yet volatilized or dissolved from the oil) is simulated as floating slicks, emulsions and/or tarballs, or as dispersed oil droplets of varying diameter (some of which may resurface). Sublots of the spilled oil are represented by Lagrangian elements ("spillets"), each characterized by mass of hydrocarbon components and water content, location, thickness, diameter, density, and viscosity. Spreading (gravitational and by transport processes), emulsification, weathering (volatilization and dissolution loss), entrainment, resurfacing, and transport processes determine the thickness, dimensions, and locations of floating oil over time. The output of the fate model includes the location, dimensions, and physical-chemical characteristics over time of each spillet representing the oil (French McCay, 2003, 2004).

2.2 Biological Effects Model

The biological exposure model in SIMAP estimates the area, volume or portion of a stock or population affected by surface oil, concentrations of oil components in the water, or sediment contamination (French McCay, 2003, 2004). For wildlife (birds, mammals and turtles), the number or fraction of a population suffering oil-induced effects is proportional to the water-surface area swept by oil of sufficient quantity to provide a lethal or sublethal dose to an exposed animal (greater than about

10 g/m^2 oil). The probability of exposure is related to behavior: i.e., the habitats used and percentage of the time an animal spends on the water or shoreline surface of the habitat (as opposed to flying or swimming underwater). The probability of mortality once oiled (Table 1) is estimated to be nearly 100% for birds and fur-covered mammals (assuming they are not rescued and successfully treated) and much lower for other wildlife. The products of the two probabilities for various wildlife groups were derived from information on behavior and field observations of mortality after spills (reviewed in French et al., 1996).

Table 1. Probability of encounter and mortality once oiled assumed in the model if species is present in the area swept by oil exceeding a threshold thickness.

Wildlife Group	**Probability**
Surface seabirds and waterfowl	99%
Nearshore aerial divers	35%
Aerial seabirds	5%
Wetland wildlife (waders and shorebirds)	35%
Cetaceans	0.1%
Furbearing marine mammals	75%
Pinnipeds (non-furbearing), manatee, sea turtles (except hatchlings)	1%

Area swept is calculated for the habitat(s) occupied by each group of wildlife, and species or species groups are assigned to behavior groups to evaluate their loss based on similarity in habitat usage, percentage of the time the species spends on the water surface, and vulnerability to oil (i.e., having feathers or fur versus not). Wildlife mortality is directly proportional to the species' densities and the percent mortalities assumed. The wildlife mortality model was previously validated with wildlife impact data for the *Exxon Valdez* (French McCay, 2004), the *North Cape* (French McCay, 2003), and 12 other oil spills (French McCay and Rowe, 2004), verifying that these values are reasonable.

2.3 Natural Resource Damages Based on Habitat Restoration Costs

Under the US Oil Pollution Act of 1990 NRDA regulations published in January of 1996 by the US Department of Commerce (National Oceanic and Atmospheric Administration, NOAA), the approach to NRDA has to use compensatory restoration costs rather than the older approach using economic valuation. In the modeling herein, the scaling of compensatory restoration employs methods currently practiced by NOAA and other trustees, i.e., Habitat Equivalency Analysis (HEA). The scaling methods were initially developed for use in the *North Cape* case, as described in French McCay and Rowe (2003). These methods have also been used in several other cases, as well as in successful claims for 23 cases submitted by the Florida Department of Environmental Protection to the US Coast Guard, National Pollution Fund Center (French McCay et al., 2003).

Restoration should provide equivalent quality fish, invertebrate and wildlife biomass to compensate for the lost production (i.e., due to the injury caused by the spill). Equivalent quality implies same or similar species with equivalent ecological role and value for human uses. The equivalent production or replacement should be discounted to present-day values to account for the interim loss between the time of the injury and the time when restoration provides equivalent ecological and human services.

Habitat creation or preservation projects have been used to compensate for injuries of wildlife, fish and invertebrates. The concept is that the restored habitat leads to a net gain in wildlife, fish and invertebrate production over and above that produced by the location before the restoration. The size of the habitat (acreage) is scaled to just compensate for the injury (interim loss).

In the model, the habitat may be seagrass bed, saltmarsh, oyster reef, freshwater or brackish wetland, or other structural habitats that provide such ecological services as food, shelter, and nursery habitat and are more productive than open bottom habitats. The injuries are scaled to the new primary (plant) or secondary (e.g., benthic) production produced by the created habitat, as the entire food web benefits from this production. A preservation project that would avoid the loss of habitat could also be scaled to the production preserved. The latter method would only be of net gain if the habitat is otherwise destined to be destroyed. In this analysis we assume only habitat creation projects would be undertaken.

The approach to scaling the size of the needed project is to use primary production to measure the benefits of the restoration. The total injuries in kg are translated into equivalent plant (angiosperm) production as follows. Plant biomass passes primarily through the detrital food web via detritivores consuming the plant material and attached microbial communities. When macrophytes are consumed by detritivores, the ecological efficiency is low because of the high percentage of structural material produced by the plant, which must be broken down by microorganisms before it can be used by the detritivore. Each species group is assigned a trophic level relative to that of the detritivores. If the species group is at the same trophic level as detritivores, it is assumed 100% equivalent, as the resource injured would presumably have the same ecological value in the food web as the detritivores. If the injured resource preys on detritivores or that trophic level occupied by the detritivores, the ecological efficiency is that for trophic transfer from the prey to the predator. Values for production of predator per unit production of prey (i.e., ecological efficiency) are taken from the ecological literature, as reviewed by French McCay and Rowe (2003). The ecological efficiencies assumed are in Table 2.

The equivalent compensatory amount of angiosperm (plant) biomass of the restored resource is calculated as kg of injury divided by ecological efficiency. The ecological efficiency is the product of the efficiency of transfer from angiosperm to invertebrate detritivore and efficiency from detritivore to the injured resource, accounting for each step up the food chain from detritivore to the trophic level of concern. Table 3 lists the composite ecological efficiency relative to benthic invertebrate production for each trophic group in the model.

Table 2. Assumed ecological efficiencies for one trophic step.

Consumer	**Prey/food**	**% Efficiency**
Invertebrate detritivore	Angiosperm	6.6

Invertebrate	Microalgae	10
Invertebrate	Microorganisms	20
Invertebrate or fish bottom feeder	Detritivores, microalgae	10
Invertebrate or fish	Invertebrate	20
Invertebrate or fish filter feeder	Plankton	20
Invertebrate or fish piscivore	Finfish	20
Sea turtles	Macrophytes, invertebrates	4
Birds, mammals	Invertebrate	2
Birds, mammals, piscivores	Finfish	2

Table 3. Composite ecological efficiency relative to benthic invertebrate production by trophic group.

Species Category	**Trophic Level**	**Ecological Efficiency Relative to Benthic Detritivores (%)**
Fish and Invertebrates:		
Small pelagic fish	planktivorous	20
Large pelagic fish	Piscivores/predators	0.8
Demersal fish	bottom feeders	10
Mollusks	filter/bottom feeder	20
Benthic invertebrates (non-molluscan)	filter/bottom feeder	20
Demersal macroinvertebrate predators	predate bottom feeders	4
Birds:		
Waterfowl	bottom feeders	2
Seabirds	piscivores	0.4
Waders	piscivores	0.4
Shorebirds	bottom feeders	2
Raptors	piscivores	0.4

The productivity gained by the created habitat is corrected for less than full functionality during recovery using a sigmoidal recovery curve. Discounting at 3% per year is included for delays in production because of development of the habitat, and delays between the time of the injury and when the production is realized in the restored habitat. The equations and assumptions may be found in French McCay and Rowe (2003).

The needed data for the scaling calculations are:

- number of years for development of full function in the restored habitat;
- annual primary production rate per unit area (P) of restored habitat at full function (which may be less than that of natural habitats);
- delay before restoration project begins; and
- project lifetime (years the restored habitat will provide services).

In most locations, as in this case, it is likely that saltmarsh restoration would be undertaken as restoration for wildlife, fish and invertebrate injuries. Seagrass (eelgrass) bed restoration is also an option. However, this requires good water quality and appropriate environmental conditions to be successful. The calculations of NRDA costs made here are based on (saltmarsh) wetland restoration, as this is most likely to be pursued.

HEA calculations for saltmarsh were performed following the methods in French McCay and Rowe (2003), as summarized above. The calculations are based on estimated aboveground primary production rates in saltmarshes. It is assumed that the created saltmarsh requires 15 years to reach 99% of full function (based on PERL, 1990; Zedler, 1992; Seneca and Broome, 1992; French et al., 1996), ultimately reaching 80% of natural habitat productivity, the restoration begins 3 years after the spill, and the project lifetime is 50 years. For the injured resources, all weights are as wet weight and dry weight is assumed 22% of wet weight. For birds, the body mass per animal (from French et al., 1996) is used to estimate injury in kg (multiplying by number killed and summing each species category).

Roman and Daiber (1984) estimated total production rates (above and below ground) of *Spartina alterniflora* in Delaware Bay as 6580 g dry weight m^{-2} yr^{-1}. Valiela et al. (1976) estimated aboveground production in New England saltmarshes as 420 g dry weight m^{-2} yr^{-1} in low marsh and 630 g dry weight m^{-2} yr^{-1} in high marsh. Valiela et al. (1976) also estimated belowground production at 3410 g dry weight m^{-2} yr^{-1} in low marsh and 2520 g dry weight m^{-2} yr^{-1} in high marsh. Based on these data, 15% of total production is above ground. Applying this ratio, aboveground *Spartina* production in Delaware marshes is estimated as 990 g dry weight m^{-2} yr^{-1}. In addition, benthic microalgal production provides 106 g dry weight m^{-2} (Van Raalte et al., 1976). Saltmarsh creation cost, corrected to 2006$ assuming 3% inflation per year, is about \$50.07/m^2 (French et al., 1996).

3 Model Inputs

3.1 Geographical and Environmental

For geographical reference, SIMAP uses a rectilinear grid to designate the location of the shoreline, the water depth (bathymetry), and the shore or habitat type. The grid was generated from a digital coastline using the ESRI Arc/Info compatible Spatial Analyst program. The cells were then coded for depth and habitat type. The model identifies the shoreline using this grid. The digital shoreline, shore type, and habitat mapping for the Chesapeake were obtained from Environmental Sensitivity Index (ESI) Atlas database compiled for the area by Research Planning, Inc. (RPI). These data are distributed by NOAA Hazmat (Seattle, WA). The gridded habitat type data are shown (in simplified categories) in Figure 1. The grid scale resolution is a cell size of 15.8 m E-W by 20.2 m N-S.

Depth data were obtained from Hydrographic Survey Data supplied on CD-ROM by the U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Geophysical Data Center. Hydrographic survey data consist of large numbers of individual depth soundings. The depth soundings were gridded using the ESRI Arc/Info compatible Spatial Analyst program. The gridded depth data are shown in Figure 2.

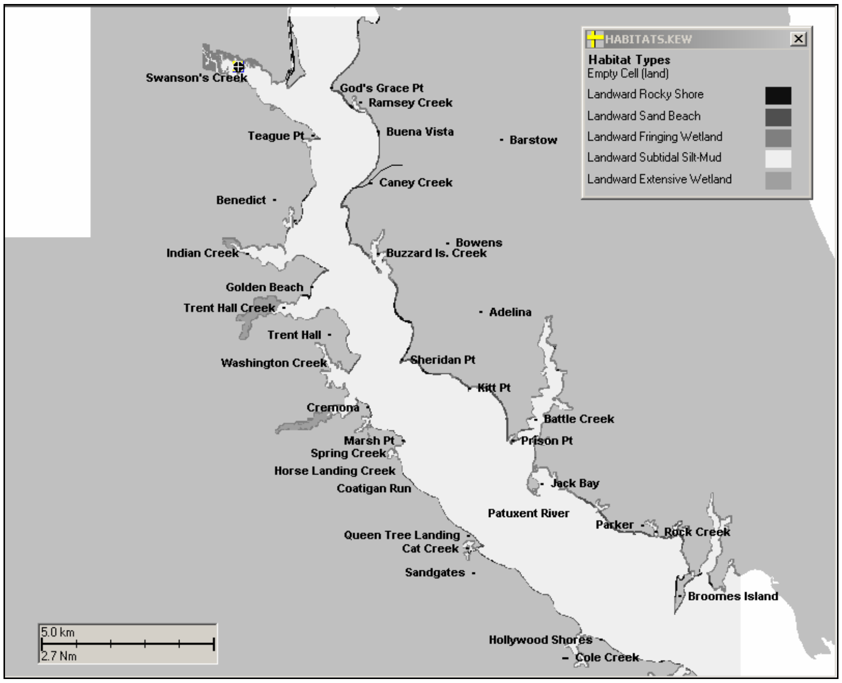

Figure 1. Gridded habitat type data used in modeling and landmark places and water bodies in the vicinity of the spill.

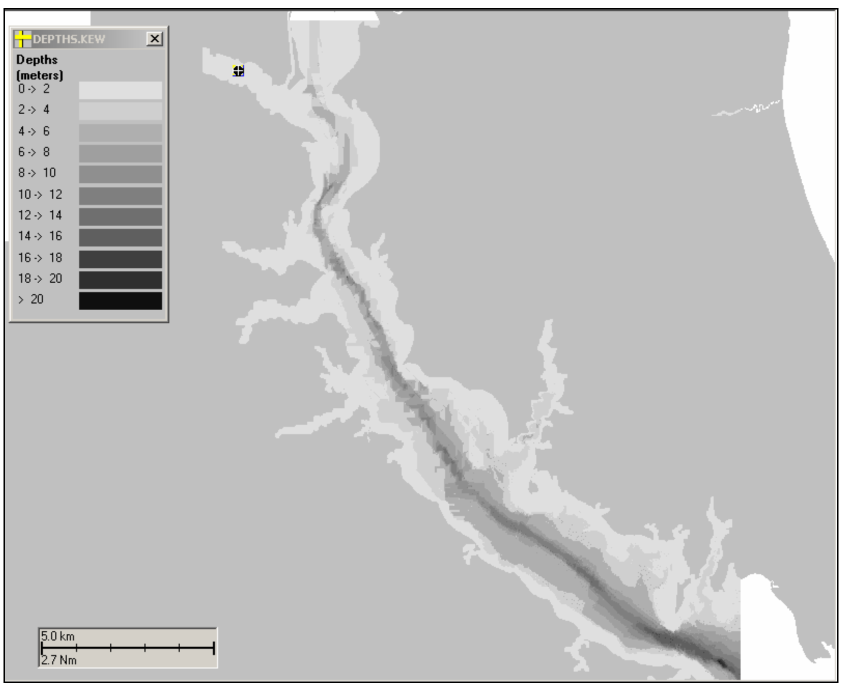

Figure 2. Gridded depth data used in modeling.

Hourly wind speed and direction data for the month following the spill were obtained from the NOAA National Data Buoy Center Internet site for the nearest station to the spill site (station TPLM2). Wind driven surface drift was specified as a percentage of wind speed and angle (to the right of downwind), or calculated within the oil spill model (based on Youssef, 1993; Youssef and Spaulding, 1993, 1994).

Surface wind drift of oil has been observed in the field to be 1-6% (average 3-4%) of wind speed in a direction 0-30 degrees to the right (in the northern hemisphere) of the down-wind direction (ASCE, 1996). These ranges were varied in model runs.

Temperature was 14^{o}C and salinity was 8 ppt, based on measurements in the Patuxent River on Apr. 4, 2000, east of mouth of Indian Creek (data provided by the Trustees). The air immediately above the water was assumed to have the same temperature as the water surface, this being the best estimate of air temperature in contact with the water and floating oil. Thus, water temperature was used in the model for calculation of evaporation rates of volatile and semi-volatile components. Thus, temperature affects evaporation rate, and so surface oil volume, but not the trajectory of the spill. The effect of water temperature within the range of a few degrees Celsius was not significant, and so not pursued in the sensitivity analysis. Salinity had little influence on the fate of the oil.

Suspended sediment is assumed 10 mg/l, a typical value for coastal waters (French et al., 1996). The sedimentation rate is set at 1 m/day. These default values have no significant affect on the model trajectory, so their values are not explored further.

The horizontal diffusion (randomized mixing) coefficient was assumed to be in the range 1-10 m^2/s. The vertical diffusion (randomized mixing) coefficient was assumed 0.0001 m^2/s. These are reasonable values based on Okubo (1971), Okubo and Ozmidov (1970) and modeling experience. The vertical diffusion coefficient used, as well as values plus or minus an order of magnitude, kept the water column well mixed, and so variation of this parameter had no significant impact on the results.

3.2 Currents

A three-dimensional tidal hydrodynamic model application to the Patuxent River estuary was performed using Applied Science Associate’s time-dependent generalized non-orthogonal boundary-fitted model in spherical coordinates developed by Muin and Spaulding (1997a). The coded application of this model, called BFHYDRO (Boundary Fitted HYDROdynamic model), has been successfully applied to coastal and estuarine waters. Some recent applications where the model was validated include Mount Hope Bay (Swanson, et al., 2005), Providence River (Muin and Spaulding, 1997b), San Francisco Bay (Sankaranarayanan and French-McCay, 2003a) and Bay of Fundy (Sankaranarayanan and French-McCay, 2003b). The boundary-fitted model technique matches the model coordinates with the shoreline boundaries and allows the user to adjust the model grid resolution as desired. Thus, the system allowed us to use a very fine grid resolution in Swanson Creek (15m), with the grid closely fitting the river boundaries, and a coarse grid resolution (150m) in the lower Patuxent River (Figure 3). The boundary-fitted grid shown in Figure 3 consisted of 357 x 187 segments with 12900 water cells in the horizontal plane, and 11 sigma levels in the vertical plane.

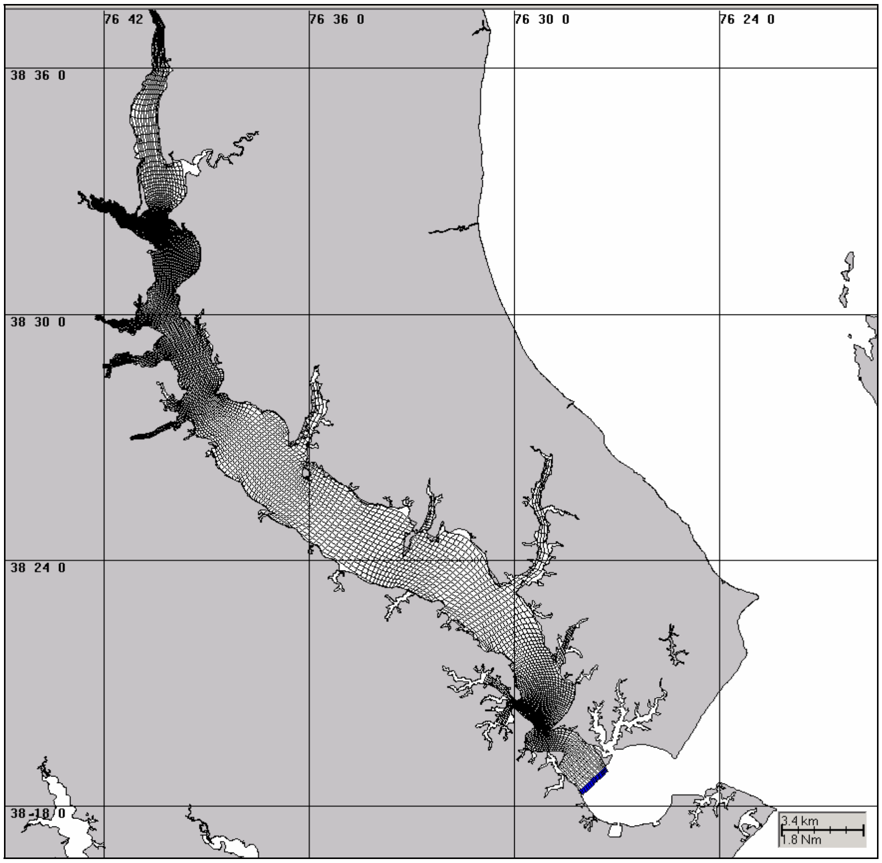

Figure 3. Boundary-fitted grid for the Patuxent River study area.

The model forcing functions consisted of time series of sea levels along the open boundary and freshwater flows at the head of the estuary. For the downstream boundary, observed 6-minute water levels recorded by National Oceanic and Atmospheric Administration's tide gauge at Solomons Island, Maryland on the Patuxent River were obtained from the National Ocean Service (NOS, 2000) at http://co-ops.nos.noaa.gov. Hourly stream flows at Bowie, Maryland for the study period, obtained from United States Geological Survey were used as freshwater flows at the upstream boundary (Figure 4).

The model-predicted elevations at two locations in the Patuxent River (Figure 5) showed good correlation with predictions from Tides and Currents (Tides & Currents Pro for Windows, Version 3.0. Nautical Software Inc.), with correlation coefficients exceeding 0.875 (Table 4). Tides and Currents, developed by Nobeltech Corporation, predicts water level and currents at selected locations from a harmonic composition tidal constituents for surface elevations and currents, published by NOAA (2000). Published harmonic tidal constituents of surface elevations and currents based long term observations are not available for locations in Patuxent River. Hence, the predictions of tides for locations inside the Patuxent River from Tides and Currents are based on long term observations at Fort McHeury, Maryland, while the predicted currents from Tides and Currents are based on long term observations at Baltimore Harbor Approach.

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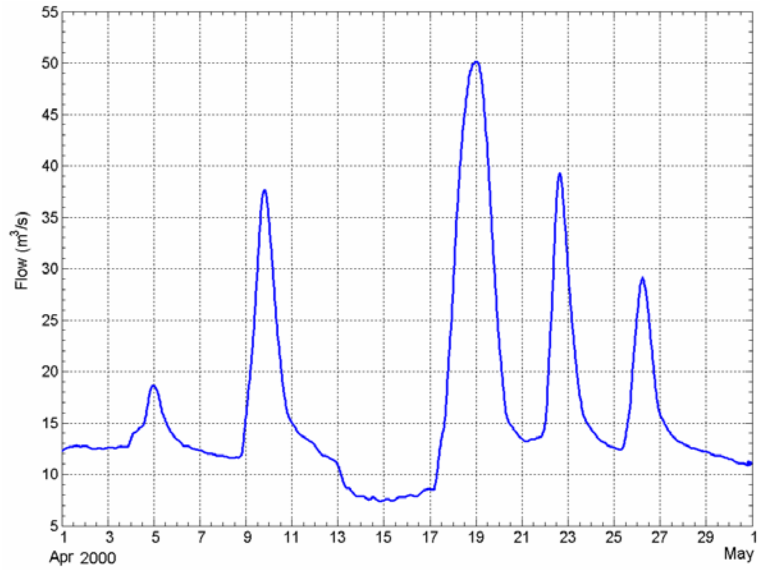

Figure 4. Time series of fresh water flow (m^3/s) into the Patuxent River at Bowie, Maryland.

The observed currents reported in Kincaid and Olson (1988) are the only such data available to date for the Patuxent River estuary. Unfortunately, only the range of current speeds, calculated as maximum flood plus maximum ebb speed, for a few stations were reported by the authors of that study. A recent paper by Lung and Bai (2003) also relied on the observations from Kincaid and Olson (1988) for validating their water quality model for the Patuxent River estuary, noting that no other data source was available. In fact, Kincaid and Olson (1988) validated their own laterally averaged hydrodynamic model for the Patuxent River using the observed maximum current range reported in Table 5. Our model-predicted maximum tidal current ranges at four locations (indicated on the map in Figure 5) compared well with maximum observed tidal current ranges (Table 5). Kincaid and Olson (1988) noted that observed that tidal velocities along the estuary ranged from 20 to 50 cm/s.

Table 4. Statistical comparison of surface elevation predictions from Tides and Currents and hydrodynamic model predictions.

Station	RMS difference (cm)	Correlation coefficient
Broomes Island[1]	6.0	0.917
Benedict[1]	8.6	0.875

[1]The surface elevation predictions at Broomes Island and Benedict from Tides and Currents were based on observations at Fort McHeury, Maryland.

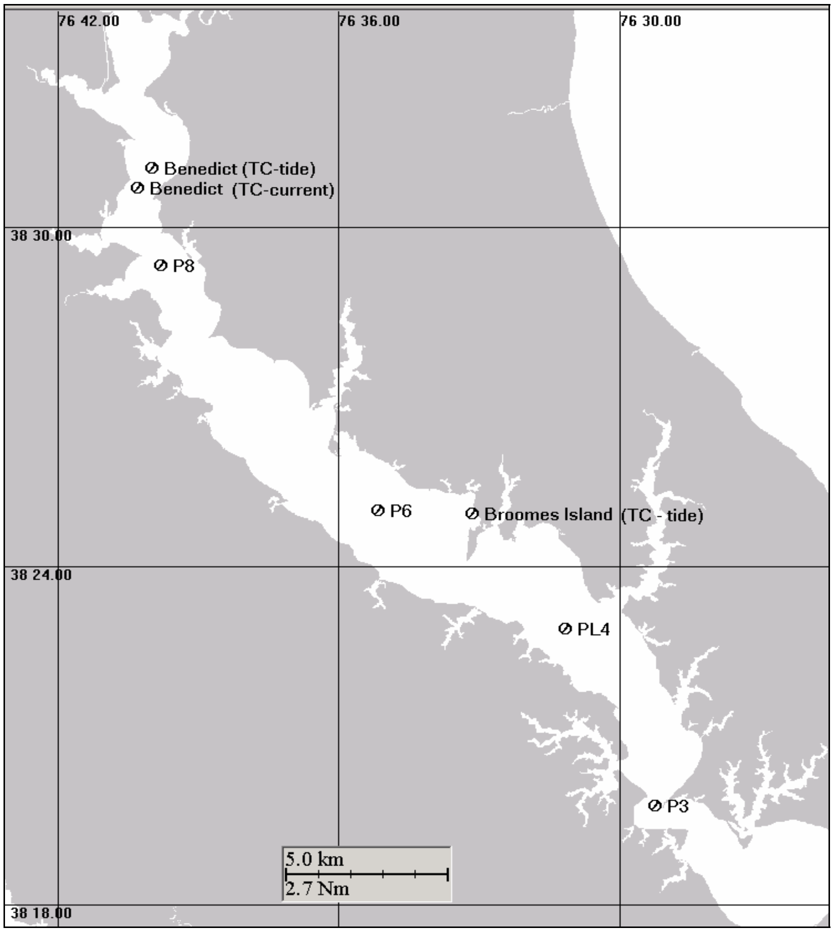

Figure 5. Locations at which the model predicted surface elevations and currents were compared with predictions from Tides and Currents (TC). Observed tidal current ranges were available at P3, PL4, P6, and P8 from Kincaid and Olson (1988).

Table 5. Comparison of observed and predicted tidal current ranges[1].

Station	Depth	Observed maximum range[2] (m/s)	Model maximum range (m/s)	Model Maximum flood (m/s)	Model Maximum Ebb (m/s)
P3	Surface	0.67	0.60	0.25	-0.35
	Bottom	0.65	0.48	0.23	-0.25
PL4	Surface	0.30	0.42	0.20	-0.22
	Bottom	0.28	0.34	0.15	-0.19
P6	Surface	0.32	0.33	0.15	-0.18
P8	Surface	0.60	0.45	0.20	-0.25
	Bottom	0.45	0.40	0.18	-0.22

[1] Range is given as the difference between maximum flood and maximum ebb. The negative ebb currents indicates southward direction of ebb currents

[2] The observed current ranges were from Kincaid and Olson (1988).

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3.3 Oil Characteristics and Toxicity

The spill was a mix of No. 6 and No. 2 fuel (so the surface tension and viscosity would be as No. 4 or No. 5 fuel oil). The key properties for the oils are in Table 6. Based on the oil density and spill volume, 481 metric tones of oil was released.

Table 6. Oil properties used in the simulations.

Property	So.LA	Reference
Density @ 25 deg. C (g/cm3)	0.91675	Source oil sample measured by Pepco
Viscosity @ 25 deg. C (cp)	123.0	French et al. (1996), as No. 5 fuel oil
Surface Tension (dyne/cm)	30.0	French et al. (1996), as No. 4 fuel oil
Fraction monoaromatic hydrocarbons (MAHs)	0.0	Assumed negligible impact
Fraction 2-ring aromatics (2-ring PAHs)	0.012992	Source oil sample measured by ENTRIX for Pepco
Fraction 3-ring aromatics (3-ring PAHs)	0.020487	Source oil sample measured by ENTRIX for Pepco
Fraction Aliphatic Volatiles[1]: boiling point < 180°C	0.0294	Jokuty et al. (1999)
Fraction Aliphatic Volatiles[1]: boiling point 180-264°C	0.224608	Jokuty et al. (1999)
Fraction Aliphatic Volatiles[1]: boiling point 264-380°C	0.293513	Jokuty et al. (1999)
Minimum Oil Thickness (mm)	0. 1	McAuliffe (1987)
Maximum Mousse Water Content (%)	0.0	(observations indicated it did not emulsify)

[1] – Environment Canada's Oil Property Catalogue (Jokuty et al., 1999) provided total hydrocarbon data for volatile fractions of unweathered oil. The aromatic hydrocarbon fraction was subtracted from the total hydrocarbon fraction to obtain the aliphatic fraction of unweathered oil.

3.4 Shoreline Oil Retention

Retention of oil on a shoreline depends on the shoreline type, width and angle of the shoreline, viscosity of the oil, the tidal amplitude, and the wave energy. In the NRDAM/CME (French et al., 1996), maximum shore holding capacity per unit area was based on observations from the *Amoco Cadiz* spill in France and the *Exxon Valdez* spill in Alaska [based on Gundlach (1987)] and later work summarized in French et al., 1996). These data are used here.

The width of the oil band on the shore varies by the slope of the shoreline, the timing of the tide relative to when oil comes ashore, and other factors. In the spill case considered here, the tide range is small and river level varied more by river flow and weather conditions. In addition, oil on shorelines is typically patchy. However, in the model it is the volume holding capacity per length of shore that is used to retain or release oil at a given segment (i.e., patchiness and the width is actually not resolved in the model—it is volume per shore length that is actually used). Thus, a range of oil banding widths, combined with the oil maximum thickness values (Table

7) from Gundlach et al. (1987), was varied in the sensitivity analysis. The shore width (intertidal zone width) was varied from 1 to 100 cm in alternate model runs.

Table 7. Maximum oil thickness and natural removal rates by shore type.

Shore Type	**Maximum Oil Thickness (mm)**	**Natural Removal (Erosion) Half Time (days)**
Rocky	2	1
Gravel	9	10
Sand	17	5
Tidal flat	6	1
Wetland	30	500

3.5 Scenario

The following conditions were assumed. The spill site was at the Pepco pipeline in Swanson Creek at near $38^{o}32.66'N$, $76^{o}42.10'W$. The release occurred at the water surface. The spill was on April 7, 2000 between 9:30AM and 2:30PM EDT. The volume released was estimated at 138,600 gallons.

Mechanical removal (i.e., by skimming and vacuum trucks) occurred after hour 24 and during daylight only (assumed 6:45AM-7:30PM based on sunrise/sunset times). Hourly mechanical recovery rates and locations deployed were based on the equipment on scene, as described in Etkin et al. (2006). Removal occurred if the oil thickness exceeded 13 μm (0.0005 inch), current speed did not exceed 35 cm/s (0.7 kt), and wave height did not exceed 1 m (3.5 ft, based on guidance in API et al., 2001).

3.6 Biological Abundances

The model uses average number or biomass per unit area (#/km^2) in appropriate habitats. In the "type A" model database (French et al., 1996), the closest biological province to the spill site is the upper Chesapeake Bay. Table 8 lists these densities, as well as those based on estimates of bird abundance in the Patuxent River along the path of the oil (i.e., in a river area of 24 km^2) as observed at the time of the spill (Wildlife Injury Workgroup, Chalk Point NRDA Council, 2001). As the model results are proportional to the assumed pre-spill density, it is clear that the results are highly sensitive to the assumed densities. (Thus, actual model runs varying density were not needed, and the results of that sensitivity analysis can be inferred.) For the spill location and time, the generalized regional and seasonal mean density data from French et al. (1996) for the upper Chesapeake Bay was not accurate enough for a spill assessment, as site- and event-specific data were quite different (Table 8).

Table 8. Field-based and model estimates of birds oiled: Pepco pipeline spill.

Species	Mean Spring Density in Upper Chesapeake Bay from French et al. (1996)	Observed Density in the Patuxent River at the Time of the Pepco Spill (# km^{-2})
Mallard	21.5	1.2
Green-winged teal	-	2.1
American coot	-	1.7
Canada goose	-	0.5
Mute swan	-	-
Ruddy duck	-	35.5
Black duck	0.8	-
Greater scaup	-	1.7
Loons	-	-
Double-crested cormorant	-	8.3
Laughing gull	5.6	-
Herring gull	2.0	-
Terns	0.6	-
Kingfisher	-	-
Virginia rail	-	-
Great blue heron	5.2	-
Osprey	.4	0.6
Total birds	36.0	51.6

4 Results

4.1 Trajectory and Fate

The fates model results of surface oil were visually compared to observed slick locations (e.g., from over-flights), SCAT reports, shoreline oiling maps, and other field data, as available (ENTRIX, 2000). Figure 6 shows the observed locations of shore oiling, based on the SCAT reports and data.

Modeling of the trajectory and fate of the oil was performed using SIMAP, varying the horizontal turbulent diffusion coefficient (1 to 10 m^2/s) to evaluate sensitivity to this assumption. It was found that the model was sensitive to this assumption, and that 5 m^2/s provided the best fit to the observations. If the horizontal turbulent diffusion coefficient was lower than this value, the oil did not spread down river quickly enough. If the horizontal turbulent diffusion coefficient was higher than this value, the oiling was too great down river, especially on the east side of the Patuxent River.

Wind drift was varied in the range 1-6% of wind speed and 0-30 degrees to the right of down wind. The best fit to the observed movements of the oil was 3% of wind speed and 0 angle relative to down-wind. Higher angles (to the right on down-wind) moved the oil too quickly towards the west bank of the river. Varying the percent of wind speed either slowed or increased the speed of oil movement out of Swanson Creek, with the best match being 3% of wind speed. The Youssef and Spaulding wind drift model, which was developed for open water situations, was found not to fit the observations for this spill. The Youssef and Spaulding model predicts an angle of about 20 degrees at the water surface. As the water body is

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shallow and narrow, and the winds were primarily from the west and northwest, a drift angle would not be expected in this situation.

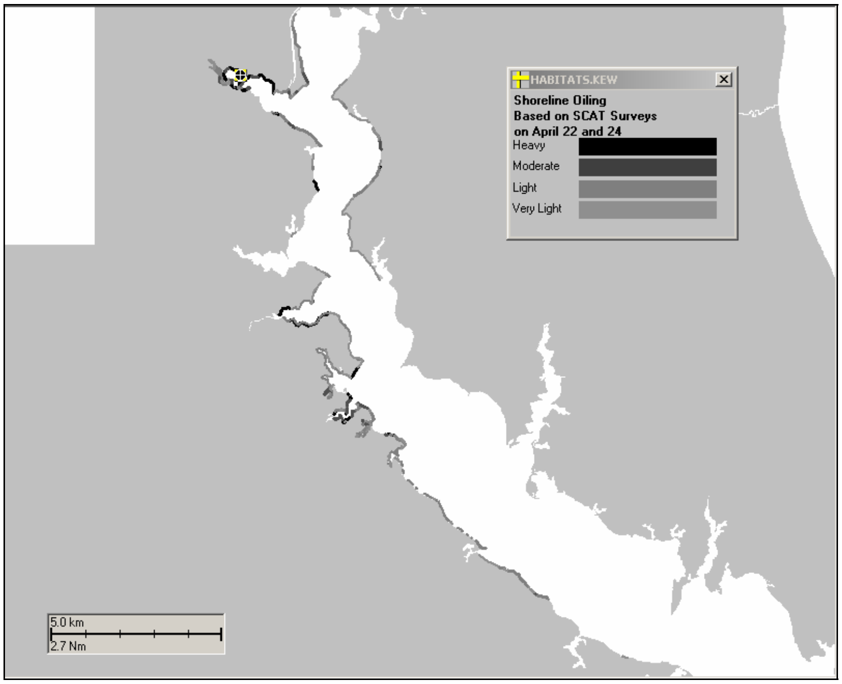

Figure 6. Observed locations of shore oiling, based on SCAT reports.

The width of shore oiling was varied from 1 to 100 cm. The width was multiplied by the oil holding capacity of the shore type to calculate the maximum volume the shore would hold per unit length of shore. The model was sensitive to the assumed width, and 20 cm provided the best fit to the observations. Figures 7-10 shore the simulated shore oiling, which may be compared with Figure 6. The modeled shore oiled is summarized in Table 9. If the width was higher than this value, the oil did not spread down river, as it was absorbed in Swanson Creek and on the opposite shore (in Buena Vista area, Figure 1). If the width was lower (thinner) than 20 cm, the oiling was too great down river.

In the sensitivity analysis, all of these parameters (horizontal turbulent diffusion coefficient, wind drift speed, wind drift angle, and width of shore oiling) were varied together in a matrix of permutations. However, in this spill case, the parameters did not interact to a significant degree. Variation of one parameter modified the results independent of the other parameters examined in the patterns described above. This might not be the case in other spill hindcasts under different circumstances.

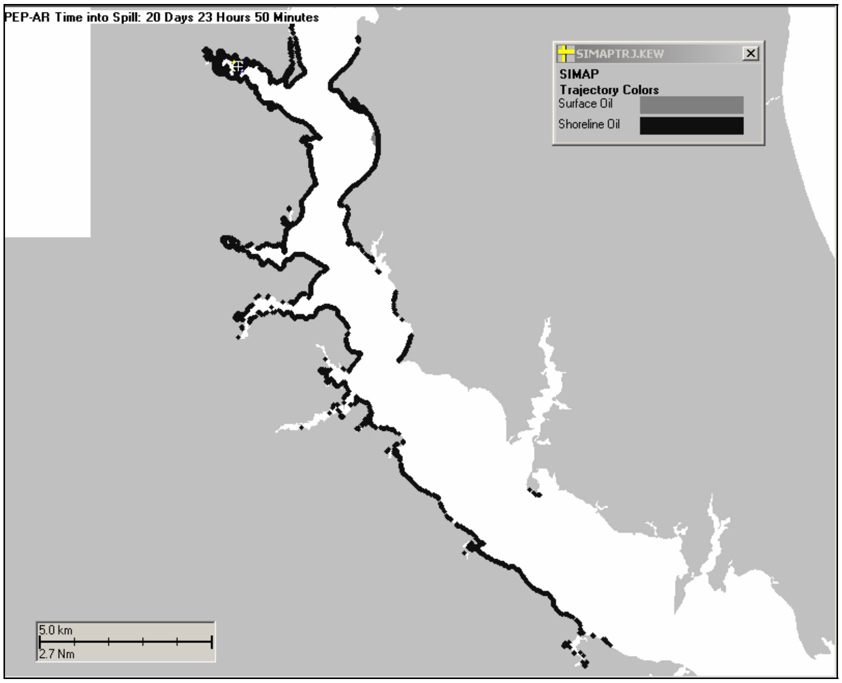

Figure 7. Shoreline oiling (to any degree > 1 g/m2) after 21 days (no shoreline cleanup simulated).

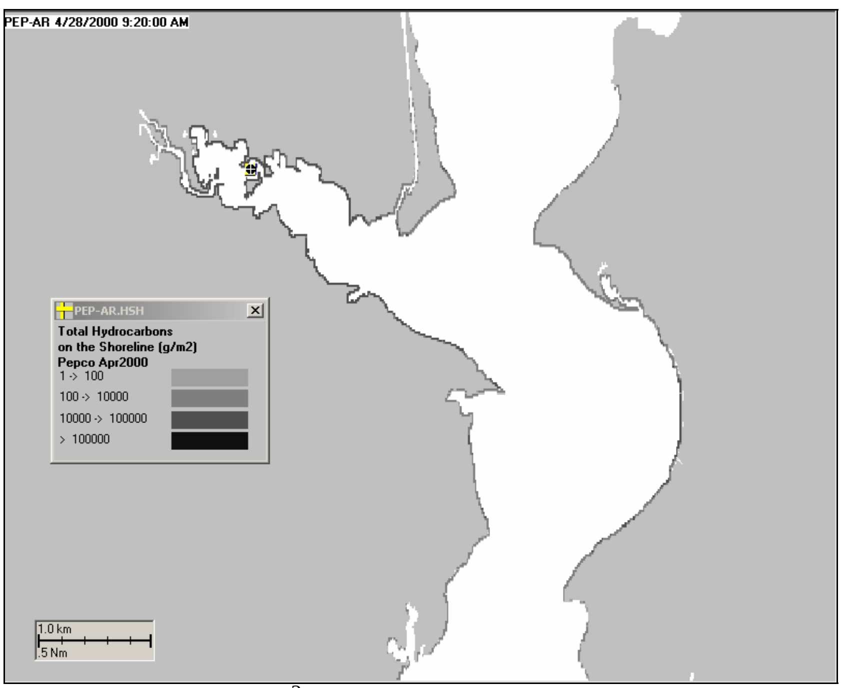

Figure 8. Total hydrocarbons (g/m^2) on the shore (no shoreline cleanup simulated) after 21 days (view 1).

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Figure 9. Total hydrocarbons (g/m^2) on the shore (no shoreline cleanup simulated) after 21 days (view 2).

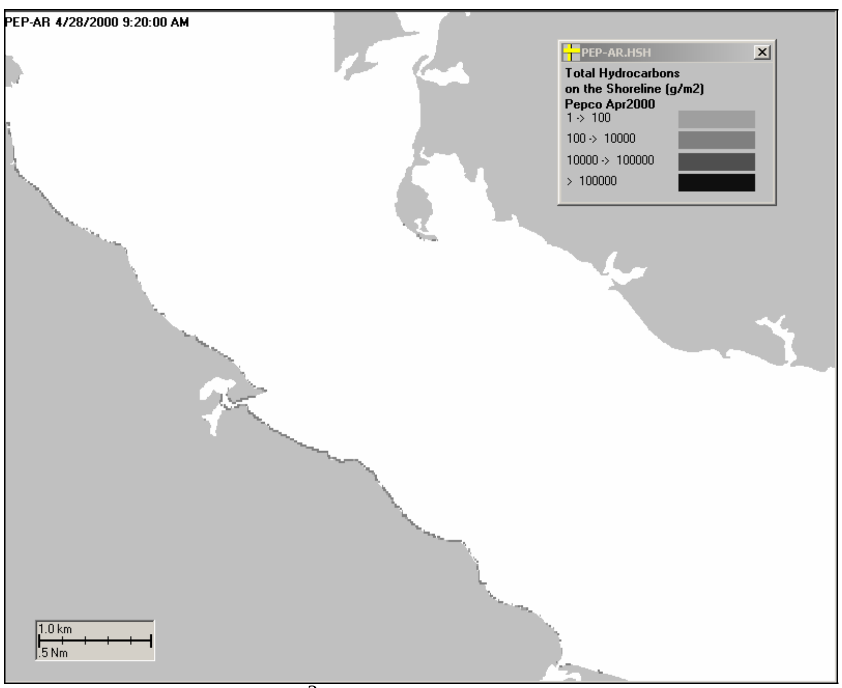

Figure 10. Total hydrocarbons (g/m^2) on the shore (no shoreline cleanup simulated) after 21 days (view 3).

Table 9. Shoreline oiled (m^2) with average thickness greater than a threshold (1mm ~ 1kg/m2).

Habitat type	**Thickness threshold**				
	>1 mm	**>0.1 mm**	**>0.01 mm**	**>0.001 mm**	**>0.0001 mm**
Rocky shoreline	504	873	1,048	1,048	1,048
Sand beach	2,247	4,028	4,572	4,640	4,640
Wetland	4,690	6,883	7,451	7,466	7,466
Total shoreline	7,441	11,784	13,071	13,154	13,154

Figures 11-14 show the trajectory of the floating oil. In general, the oil was held by southeast winds in the saltmarsh in Swanson Creek near the spill site until 3PM on April 8 (1.25 days into the spill) when a front passed through followed by strong northwest winds. The oil was blown out of Swanson Creek to against the eastern shore at Buena Vista (Figures 12-13) until late on April 10 (3.5 days into the spill), and then was moved down river by northeast and north winds (Figure 14, summarized in Figure 11). The trajectory and timing of the oil movements agree with observations made after the spill. Table 10 gives approximate thickness ranges for surface oil of varying appearance, as mapped in Figures 12-13. Black oil is >1000 g/m^2 thick. Brown sheens are 1-1000 g/m2. Rainbow sheen is about 200-800 mg/m^2 and silver sheens are 50-800 mg/m^2 thick (NAS, 1985).

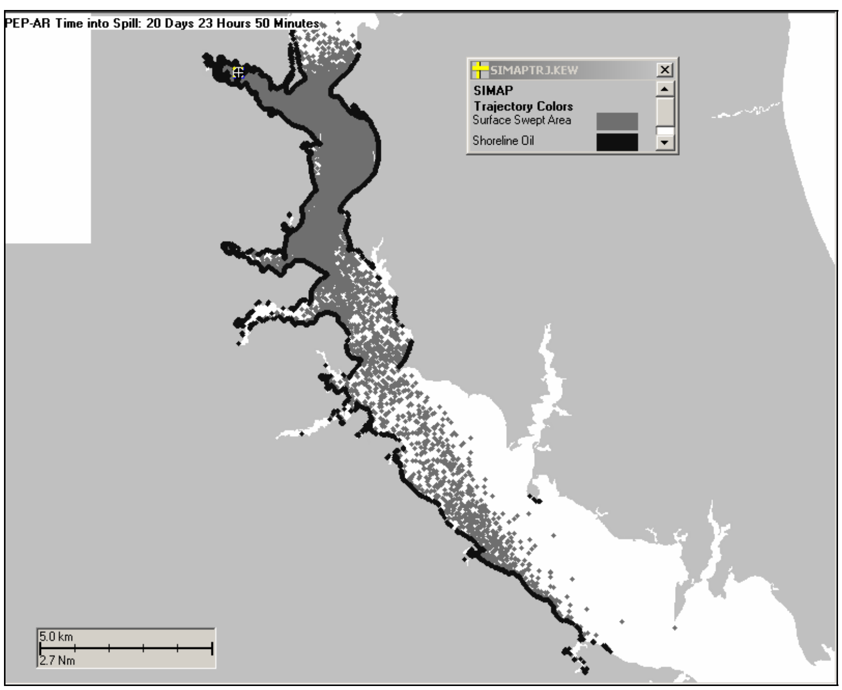

Figure 11. Trajectory of spill, cumulative exposure over 21 days.

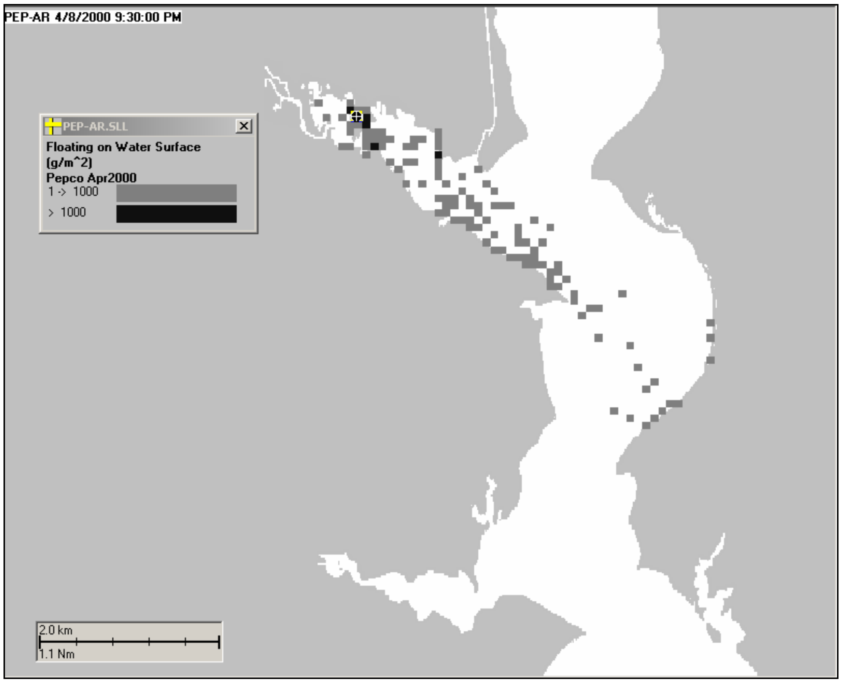

Figure 12. Thickness of oil floating on the surface April 8 at 21:30 hrs (1.5 days after the start of the spill).

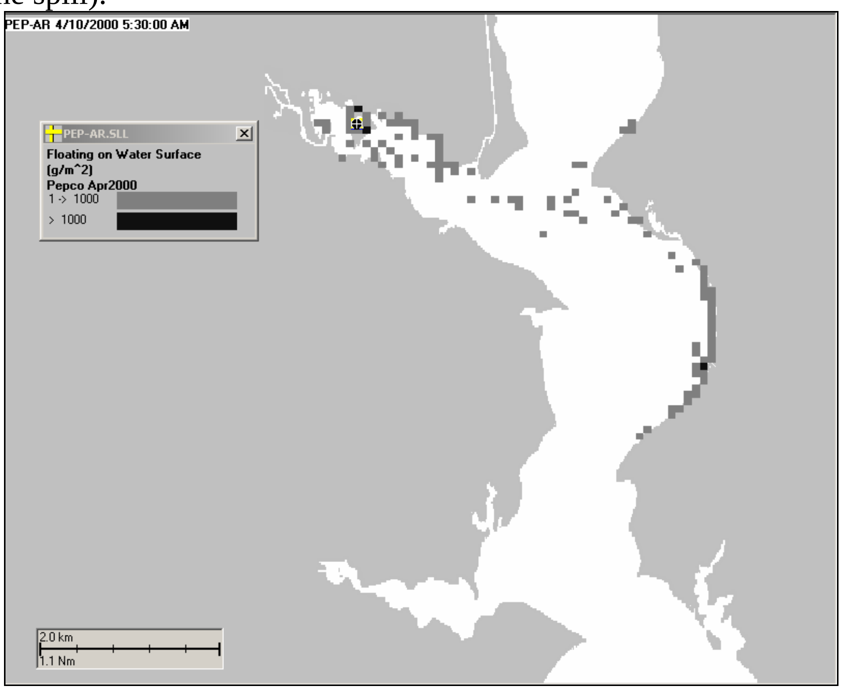

Figure 13. Thickness of oil floating on the surface April 10 at 05:30 hrs (2.8 days) after the start of the spill.

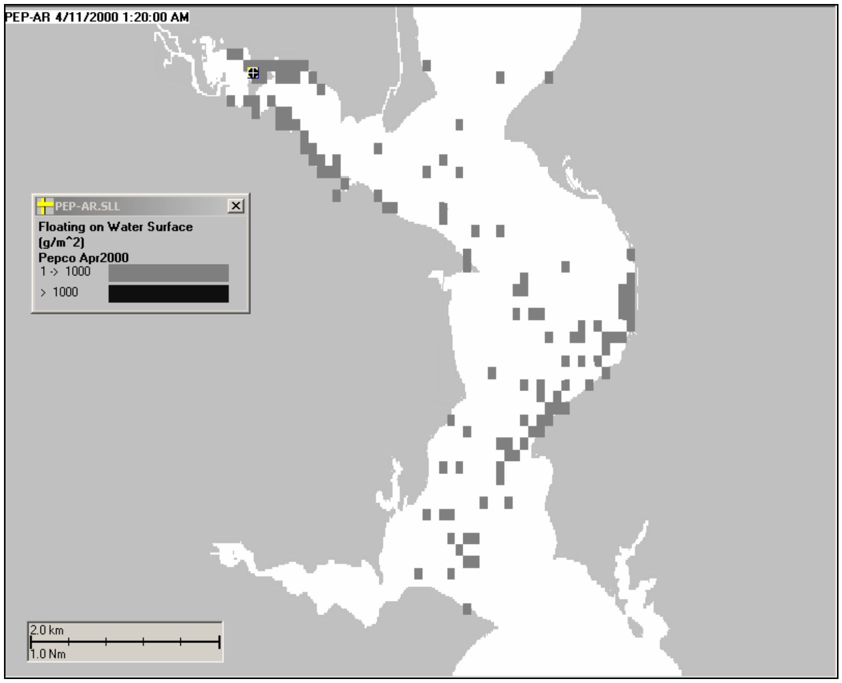

Figure 14. Thickness of oil floating on the surface April 11 at 01:20 hrs (3.7 days after the start of the spill).

Table 10. Oil thickness (microns ~ g/m^2) and appearance on water (NAS, 1985).

Minimum	Maximum	Appearance
0.05	0.2	Colorless and silver sheen
0.2	0.8	Rainbow sheen
1	4	Dull brown sheen
10	100	Dark brown sheen
1000	10000	Black oil

4.2 Biological Impacts

Wildlife rescue efforts resulted in the collection of 109 live birds and 55 dead birds (Wildlife Injury Workgroup, Chalk Point NRDA Council, 2001); however, these represent only a fraction of the actual number of birds affected by the spill. The Wildlife Injury Workgroup, Chalk Point NRDA Council (2001) estimated the actual mortality based on estimates of bird abundance observed in the area along the path of the oil (i.e., in a river area of 24 km^2) and that fraction of the 24-km^2 area where oiling likely occurred. Estimates of total mortality are shown in Table 11, along with model estimates using bird density data from French et al. (1996, NRDAM/CME, Province 16, upper Chesapeake Bay), and that data updated with the observed bird density at the time of the spill. The agreement of the model estimate, updated with the observed bird density from Wildlife Injury Workgroup, Chalk Point NRDA Council (2001), is within a factor of 2-4.

Table 11. Field-based and model estimates of birds oiled: Pepco pipeline spill.

Species	**Estimated (Wildlife Injury Workgroup, 2001)**	**Model: Densities from French et al., (1996)**	**Model: Based on Observed Densities****
Mallard	45	282	21
Green-winged teal	11	0	36
American coot	35	0	29
Canada goose	9	0	9
Mute swan	7	*	*
Ruddy duck	371	0	295
Black duck	0	11	0
Greater scaup	9	0	14
Loons	1	*	*
Double-crested cormorant	43	0	12
Ring-billed gull	2	*	*
Herring gull	1	1.6	2.8
Terns	2	0	0.8
Kingfisher	1	*	*
Virginia rail	1	*	*
Great blue heron	3	1.3	2.3
Osprey	6	2.0	5
Total birds	547	297	673

* Pre-spill density estimates unavailable, so no model estimate was made
** Best model estimates

The biomass losses are directly proportional to the pre-spill abundance assumed in the model inputs. Thus, a change in abundance is directly translated to a proportional change in the quantified injury (and natural resource damages calculated using the HEA model), and the uncertainty of the bird impact estimate is proportional to uncertainty in pre-spill density. This analysis demonstrates the importance of making on-scene estimates of bird densities at the time of a spill.

5 Discussion

The model was capable of hind-casting the oil trajectory and shoreline oiling, given accurate wind and surface current data, as well as specific details of the timing and amounts released. As winds and currents are the primary forcing variables on oil fate, obtaining accurate data at all significant spatial and temporal scales is very important to the accuracy of any simulation. Wind data are generally available for many locations and times of interest. However, deployment of a meteorological station near the spill site would improve accuracy depicting the local winds.

In most cases, the most reliable and representative current data are those generated by a calibrated hydrodynamic model. It is typically not feasible to measure site- and event-specific currents in enough spatial detail for direct input to an oil spill model, except perhaps in locations where high-frequency radars are available and can provide a surface current velocity field for the duration of the event. However, current meter measurements at specific locations after a spill would provide valuable calibration data for a hydrodynamic model.

The specific details of the timing and amounts of oil released are often difficult to obtain or estimate. Yet, these model inputs have a large influence on the results. In the PEPCO spill, the details of the release were not available until long after the spill. However, the inputs used here were determined based on witness accounts and other information on the event. Obtaining detailed information on the release is important information that is most efficiently obtained during the event. When the events are uncertain, sensitivity analysis varying the assumed release amount and timing becomes necessary.

When accurate wind, current, and release data are quantified, the remaining uncertainty is most related to the variables examined in the sensitivity analysis described above: horizontal turbulent diffusion coefficient, wind drift speed and angle, and shore holding capacity per length of shore (by shore type). Typically, there will be a need for a fate model calibration step varying these model inputs and comparing the results to field observations. Shoreline surveys, such as SCAT, provide important data on the ultimate locations where oil came ashore, as well on oil amounts per length of shore if such detail is recorded. However, observations on locations and movements of estimated quantities of oil at specific times are more useful for calibration of the trajectory. Aerial photographs from overflights, combined with observers' interpretation and descriptions, can provide valuable information for this purpose.

The biological model results for bird mortality are sensitive to the length and width of the swept path by the trajectory. Thus, the more accurate the trajectory and physical fate simulation, the better the estimate of swept area that might oil wildlife. More accurate field observations of bird densities lower the uncertainty in the model estimates, as model uncertainty is proportional to the uncertainty in bird density. Given the significant effort put into the field estimates of bird injuries and the contained water body making field observation feasible, the uncertainties in the field estimates of birds oiled are likely much lower in this case than for many other spills. Thus, the agreement of the model to the field estimates verifies that the model algorithms are realistic, and model estimates can be accurate given accurate field density estimates for the time of the spill.

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