Final Report

Chalk Point Oil Spill of April 7, 2000 in Patuxent River, MD: Modeling of the Fates and Acute Biological Effects of the Spilled Oil on the Water Column

by

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SUMMARY

Applied Science Associates' SIMAP oil fates and biological effects model was used to simulate the Chalk Point 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 pathways and fate of the oil, and thus estimate exposure to the water surface, shoreline and other habitats, water column, and sediments; and (2) an estimate of injuries to aquatic organisms that can be used to scale compensatory restoration.

The model uses incident specific wind data, current data, and transport and weathering algorithms to calculate mass balance in various environmental compartments (water surface, shoreline, water column, atmosphere, sediments, etc.), oil slick distribution over time (trajectory), and concentrations of the oil components in water and sediments. Wind data were hourly speed and direction from a nearby recording station. The tidal current data used were sketched in vectors based on published (digital) tidal charts (Tides & Currents ®). Output maps of oil slicks, water concentrations, shoreline oiling, and sediment contamination may be viewed in an animated time-varying Windows interface or as snapshots (figures).

Using the model input data and average species sensitivity to PAHs, and including the field collected mummichogs killed in the marsh, the best estimate of total injury to fish and invertebrates is 2,464 kg. This total injury includes: (1) the biomass equivalent of the direct kill, equal to 1,485 kg; and (2) future growth of the killed animals, had there not been a spill, totaling 979 kg (the production foregone).

The tidal current data used for the simulation is approximated as a rectilinear twice-daily high tide of equal tide height. Wind-driven currents are calculated by the model based on the local winds. Freshwater river flow was not included in the simulations. With more detail in the current data, the simulated fate of the oil would be more accurate spatially and temporally (in the detailed distribution). However, the overall distribution and concentrations would not vary much from those provided here.

The accuracy of the biological injury assessment depends primarily on the accuracy of (1) the fates model results and (2) the biological abundance data input to the model. With more accurate current data, the fates model and percent mortality results would be more accurate, but the percent losses would change by less than an order of magnitude. Abundance data for fish and invertebrates assumed in the modeling were as follows: (1) age classes > one year of age were based on data compiled by ENTRIX (2002); and (2) young-of-the-year (< one year of age) were based on the type A model database (French et al, 1996c). 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.

1. INTRODUCTION

April 7, 2000, a mixture of No. 2 and No. 6 fuel oils were spilled into Swanson Creek, a tributary of the Patuxent River at Chalk Point, from a pipeline break at the PEPCO power plant (responsible parties are PEPCO and ST Services). For the first 24 hours, a strong southeast wind held the oil in the saltmarsh at the spill site. Late afternoon on April 8 the winds became strong from the northwest, blowing the oil out into the Patuxent River. Winds remained strong from the northwest and west for two days, entraining the oil into the water and moving it downstream. Oil moved further downstream on subsequent days.

Modeling allows estimation of the fates of the oil to be completed in three-dimensional space and over time. Modeling uses incident specific wind and current data, and well-tested transport and weathering algorithms, to calculate mass balance in various environmental compartments (water surface, shoreline, water column, atmosphere, sediments, etc.), oil slick distribution over time (trajectory), and concentrations of the oil components in water and sediments.

The modeling analysis was performed using a model system developed by Applied Science Associates (ASA) called SIMAP (Spill Impact Model Analysis Package). SIMAP was developed from the oil fates and biological effects submodels in the Natural Resource Damage Assessment Model for Coastal and Marine Environments (NRDAM/CME). The NRDAM/CME was developed by ASA for the U.S. Department of the Interior over the past 12 years. The NRDAM/CME (Version 2.4, April 1996) was published as part of the CERCLA type A Natural Resource Damage Assessment (NRDA) Final Rule (Federal Register, May 7, 1996, Vol. 61, No. 89, p. 20559-20614; French et al, 1996a).

This report describes modeling of the fate and biological effects of the oil from the Chalk Point spill. Model results are displayed by a Windows graphical user interface that animates the trajectory and concentrations over time. The figures included here (appendices) are snapshots taken from that output.

It should be noted that the model runs were made using approximate current data sketched in from tidal charts. Freshwater flow in the river was not included (and so the downstream movement will be slightly underestimated). The wind data used were hourly wind speed and direction from a nearby measurement station, and so quite accurate. Wind-driven currents were estimated in the model based on this wind data. Other data inputs were also accurate such that results are not sensitive to the uncertainty of these parameters. Thus, the uncertainty and detail of the model simulations is a function of the current data used.

Refinement of the current data would provide a more accurate simulation in the detail of the oil movements. The most accurate way to refine the current data would be to use a hydrodynamic model that interpolates between measurement points and external forcing factors (tides, stream flow, etc.). ASA's hydrodynamic model is capable of using coastline and bathymetry data already set up for the SIMAP oil fates model, and it

outputs a current data file that may be read into the SIMAP model directly. However, the quantified injury resulting from this analysis would not be substantially different from the results reported herein.

Section 2 describes the physical fates and biological effects models used for this analysis. Section 3 describes the model input data and assumptions. Results of the physical fates model are described in Section 4. Section 5 describes the biological impacts and injury quantification results. Section 6 summarizes the injury quantification. References cited are in Section 7. Appendices provide input data and model results, in tables, maps and other figures.

2. MODEL ALGORITHMS

2.1 Physical Fates Model

The modeling analysis was performed using a model system developed by Applied Science Associates (ASA) called SIMAP (Spill Impact Model Analysis Package). SIMAP includes (1) an oil physical fates model, (2) a hydrodynamics model for simulation of currents, (3) a biological effects model, (4) an oil physical, chemical and toxicological database, (5) environmental databases (winds, currents, salinity, temperature), (6) geographical data (in a GIS), (7) a biological database, (8) a response module to analyze effects of response activities, (9) graphical visualization tools for outputs, and (10) exporting tools to produce text format output.

SIMAP was developed from the oil fates and biological effects submodels in the Natural Resource Damage Assessment Model for Coastal and Marine Environments (NRDAM/CME). The NRDAM/CME (Version 2.4, April 1996) was published as part of the CERCLA type A Natural Resource Damage Assessment (NRDA) Final Rule (Federal Register, May 7, 1996, Vol. 61, No. 89, p. 20559-20614). The technical documentation for the NRDAM/CME is in French et al. (1996a,b,c). The model algorithms will only be briefly summarized here.

The physical fates model estimates the distribution of oil (as mass and concentrations) on the water surface, on shorelines, in the water column and in the sediments. It is three-dimensional, using a latitude-longitude grid for environmental and geographical data. Algorithms based on state-of-the-art published research include spreading, evaporation, transport, dispersion, emulsification, entrainment, dissolution, volatilization, partitioning, sedimentation, and degradation. Oil mass is tracked separately for lower molecular weight aromatics (1 to 3-ring aromatics) which cause toxicity in the model, other volatiles, and non-volatiles. The lower molecular weight aromatics dissolve from the whole oil and are partitioned in the water column and sediments according to equilibrium partitioning theory (French et al., 1996a, 1999).

SIMAP includes the physical fates model in the NRDAM/CME, with several changes and additions, summarized below. A complete description of the fates model algorithms is in French et al. (1996a) and Appendix A of French (1998a), while French et al. (1999) contains documentation of the design and sources of the algorithms and data. Most of the additions to prepare SIMAP from the NRDAM/CME were made to increase model resolution, allow modification and site-specificity of input data, allow incorporation of temporally varying two- or three-dimensional current data, and facilitate analysis of results. Thus, most of the additions are to enable changes to the input data (from that provided with the NRDAM/CME) rather than to model algorithms.

Differences between the SIMAP model algorithms and those in the NRDAM/CME are:

- Evaporation algorithm: The NRDAM/CME is based on Mackay et al. (1980) and Payne et al. (1984), while SIMAP uses the empirical formulation in Stiver and

Mackay (1984) and Mackay et al. (1982), with the most volatile fractions evaporated first. The results are similar with these two algorithms.

- Entrained oil droplets: In the NRDAM/CME they are assumed to move horizontally with the same surface currents as surface slicks. Surface slicks are transported by tidal and background currents from a current data file, plus the added vectors of surface wind drift. In SIMAP, the surface wind drift is only applied to entrained droplets if they are in the surface wind-mixed layer. SIMAP tracks entrained droplets separately from surface slicks. Wind drift may be added as a user-specified, constant percentage of wind speed (such as the accepted 3-4% of wind speed, ASCE, 1996), with the option of including a drift angle clockwise (in the Northern Hemisphere) of the down wind direction. Alternatively, the algorithm developed by Youssef and Spaulding (Youssef, 1993; Youssef and Spaulding, 1993, 1994) is used for wind transport in the surface wave-mixed layer.
- Sedimentation of entrained oil droplets: This is not included in the NRDAM/CME (which assumes it is zero), but is included in SIMAP.
- Degradation rates: The NRDAM/CME uses two rates, one for water column oil and one for sedimented oil. The rates are in the database. SIMAP uses user input for these rates, and allows different rates for low molecular weight aromatics and whole oil.
- Entrainment: In both models, the same entrainment algorithm of Delvigne and Sweeney (1988) is used to simulate mixing of oil into the water by wind-driven waves breaking on the water surface. The NRDAM/CME assumes the spill occurs on the water surface. However, a subsurface release and surf entrainment algorithm has been added to SIMAP, also based on the algorithm and data of Delvigne and Sweeney (1988). The energy and particle size distribution for various assumed levels of turbulence during the release are taken from Delvigne and Sweeney (1988). The subsurface release is initialized by the model in a user-defined volume and location in the water column. The user also sets the turbulence level of the release. When applicable, highly turbulent surf entrainment is specified by the user for a window of time after the spill.
- Calculation of water column concentrations: Both models use Lagrangian particles to track the center of mass of sublots of entrained or dissolved oil. Each particle has an inferred Gausian (normal) spatial distribution of mass around it, calculated from the horizontal turbulent diffusion (randomized mixing) coefficient and the time since the mass entered the water column. In the NRDAM/CME, when concentrations are calculated, the mass is assumed evenly distributed out to one standard deviation from the center in the two horizontal directions. In SIMAP, this distribution is truly Gausian (a more accurate representation). In both models, the contributions of mass from all submerged particles is summed into a "plume grid", which is a three-dimensional concentration grid quantifying the concentrations in the water at any given time.
- Turbulent diffusion (mixing) coefficients: The NRDAM/CME uses the model of Okubo (1971) to relate the horizontal turbulent diffusion coefficient to the length scale of the grids used. Because of the large sizes of the grids in the NRDAM/CME, the horizontal coefficient defaults to a maximum of 100 m^2/sec. The vertical turbulent diffusion coefficient is assumed 1 cm^2/sec, except in the surface wind-

mixed layer (1.5 times wave height), where it is a function of wind speed. SIMAP uses user inputs for the horizontal and vertical coefficients to be used below the wind-mixed layer, and the same surface wind-mixed layer algorithm.

- Numbers of Lagrangian particles: The NRDAM/CME uses default numbers of particles to represent the oil and biota exposed to oil. In SIMAP the user may set these values.

Thus, the fates models are not significantly different in SIMAP and the NRDAM/CME, except as noted above regarding subsurface and surf entrainment, turbulent diffusion coefficients, and concentration field calculations. Additional model output formats were also added to SIMAP. SIMAP may use site-specific data inputs, whereas in the NRDAM/CME, default databases are used. The changes in the databases and model defaults, from the NRDAM/CME to SIMAP, cause the largest changes in the results produced by the two models.

2.2 Biological Effects Model

The biological effects model uses habitat-specific and seasonally-varying estimates of fish and invertebrate abundances, and productivity of plant and animal communities at the base of the food chain, to determine biological effects resulting from the spill. The model performs these calculations by first estimating the portion of a stock or population affected. The fractional loss is multiplied by abundance or biomass per unit area to quantify an injury as number or kg of biomass lost.

A rectangular grid of habitats represents the area potentially affected by the spill, with each grid cell coded for habitat type. Habitats include various water and shoreline environments. The habitat grid matches the grid set up for the physical fates model using the GIS database. A contiguous grouping of habitat grid cells with the same habitat code represents an ecosystem in the biological submodel. Fish and rates of lower trophic level productivity are assumed constant and evenly distributed across an ecosystem within each of four seasons. Animals are assumed to move at random within each ecosystem during a single season (after that they may move on). Fish eggs, larvae, and juveniles (i.e., young-of-the-year) are assumed constant and evenly distributed across each ecosystem within each month of an annual cycle. Planktonic stages (eggs and larvae in the water column) are moved with the currents.

Fish and their eggs and larvae are affected by dissolved contaminant concentration (in the water or sediment). Mortality is calculated using laboratory acute toxicity test data (LC50, concentration lethal to 50% of test individuals) corrected for temperature and time of exposure, and assuming a log-normal relationship between percent mortality and dissolved concentration. LC50s for the most toxic component of oil, dissolved aromatics, are used to define the center of that log-normal function. (See next section.) Movements of biota, either active or by current transport, are accounted for in determining time and concentration of exposure. Organisms killed are integrated over space and time by habitat type to calculate a total percentage killed.

The map of percent mortality is multiplied times abundance to estimate fish and invertebrates killed as numbers or biomass (kg). Each species and stage is assigned a behavior group: planktonic (move with currents), demersal and stationary (on the bottom exposed to near bottom water), benthic (in the sediments and stationary), demersal fish (on the bottom exposed to near bottom water and moving slowly), small pelagic fish (moving randomly and slowly in the water column), or large pelagic fish (moving randomly and rapidly in the water column). The percent mortality of the exposure group is multiplied times abundance at the time exposed and in the habitat type to calculate the species' mortality. (See French et al., 1996a for details.)

Lost production of plants and animals at the base of the food chain is also integrated in space and over time using EC50s, the effective concentration to reduce growth to 50% of normal, to parameterize a log-normal function of the same form as the mortality function. Lost production of fish and shellfish due to reduction or contamination of food supply is estimated using a simple food web model. In practice, the direct losses from acute mortality are much greater than the food web losses, which typically account for less than 1% of the estimated injury.

The biological effects model computes reduction of fish and shellfish population size and catch in the present and future years using standard fisheries models. The injury includes losses due to mortality of adults, juveniles and young-of-the-year due to the spill. Relatively high natural mortality rates of fish eggs and larvae are considered in the model, since a high number killed at the time of the spill would have died anyway. Young-of-the-year (eggs, larvae, and juveniles less than one year old) of each fish species category are tracked as percents of the one-year-old population. Young-of-the-year and older age classes are not assumed to necessarily inhabit the same environment concurrently, and their losses are calculated separately.

The biological effects model in SIMAP is the same as for the NRDAM/CME, with the following updates:

- The SIMAP biological effects model accepts site- and event-specific geographical and environmental data. Habitat and depth grids may be of fine resolution, as needed to accurately map the area of the spill.
- Current data used to transport planktonic forms is updated as for the fates model, i.e., the ability to use hydrodynamic model output in two or three dimensions, constant or time-varying.
- The toxicity parameters (i.e., LC50s) and algorithm is updated to that described in French McCay (2001), as summarized in Section 2.3.
- The biological database may be updated from the default available in the NRDAM/CME to reflect site- and event-specific abundances.
- The injury is quantified as total production lost because of the spill. (See Section 2.4)

2.3 Oil Components and Toxicity

Oil is a mixture of hydrocarbons of varying physical, chemical, and toxicological characteristics, and therefore, varying fates and impacts on organisms. In a model, oil needs to be represented by component categories, and the fate of each tracked separately. The "pseudo-component" approach (Kirstein et al. 1987; Payne et al. 1984; French et al. 1996; Jones 1997) is used, where chemicals in the oil mixture are grouped by physical-chemical properties and the resulting component category's behavior is modeled as if it were a single chemical with characteristics typical of the chemical group.

The following summarizes the oil toxicity model, OilToxEx, used in SIMAP. The full development of OilToxEx and data upon which it is based are in French McCay (2001).

The most toxic components of oil to water-column and benthic organisms are lower-molecular-weight compounds, which are both volatile and soluble in water, especially the aromatic compounds (French et al. 1996; French 1998a, 2000; French McCay 2001). It has been shown that toxicity of narcotic organic compounds, such as these lower-molecular-weight aromatics in oil (MAHs and PAHs), is related to the octanol-water partition coefficient (K_{ow}), a measure of hydrophobicity (Nirmalakhandan and Speece 1988; Hodson et al. 1988; Blum and Speece 1990; McCarty 1986; McCarty et al. 1992; Mackay et al. 1992a; McCarty and Mackay 1993; Varhaar et al. 1992; Swartz et al. 1995; French et al. 1996; French 1998, 2000, French McCay 2001). Chemicals that have a narcotic mode of action impact organisms by accumulating in lipids (such as in the cell membranes) and disrupting cellular and tissue function. The more hydrophobic the compound, the more accumulation in the tissues and the more severe the impact. However, the more hydrophobic the compound, the less soluble it is in water, and so the less available it is to aquatic organisms. Compounds of $\log(K_{ow})>5.6$ are considered insoluble and so unavailable to aquatic biota (French 2000; French McCay 2001). Thus, impact is the result of a balance between bioavailability (dissolved-component exposure) and toxicity once exposed.

The acute toxic effects of narcotic chemicals, including lower molecular weight aromatics, is additive (Swartz et al. 1995; French et al. 1996; French 1998, 2000, DiToro et al., 2000; DiToro and McGrath, 2000; French McCay 2001). The Toxic Unit (TU) model is used to estimate the toxicity of a mixture of narcotic chemicals. A TU is defined as the exposure concentration divided by the LC50 (lethal concentration to 50% of exposed organisms). For a mixture, the toxic units are additive. When Σ TU = 1, the mixture is lethal to 50% of exposed organisms.

It may be shown (French et al., 1996a, French 2000; French McCay 2001) that the LC50 of the mixture ($LC50_{mix}$) is related to the LC50 of each chemical i in the mixture and the fractional concentration of chemical i in the total mixture, $F_i = C_{w,i} / (\Sigma C_{w,i})$, where $C_{w,i}$ is the dissolved concentration of chemical i in the water.

(1) $$LC50_{mix} = 1 / \Sigma (F_i / LC50_i)$$

The values of F_i may be measured in the field, or if field samples are not available, F_i may be estimated from the source oil composition. It has been shown that for surface waters, where turbulent entrainment of oil has occurred, the values of F_i are nearly proportional to the source oil aromatic composition. The values of $LC50_i$ can be estimated using regression models relating LC50 to K_{ow} (French McCay, 2001). The 95% confidence range of this regression provides LC50s for average (50^{th} percentile), sensitive (2.5^{th} percentile), and insensitive (97.5^{th} percentile) species. This oil toxicity model is used to estimate the LC50 for the dissolved aromatic mixture originating from the spilled oil. Only compounds of $\log(K_{ow}) \leq 5.6$ are included in the additive toxicity model.

Toxicity varies with time of exposure, the LC50 decreasing as exposure time increases (Sprague, 1969; Kooijman, 1981; McAuliffe, 1987; Anderson et al., 1987; French and French, 1989; French, 1991; McCarty et al., 1989, 1992a,b; Mackay et al., 1992a; French et al.1996). This is due to the accumulation of toxicant over time up to a critical body residue (tissue concentration) that causes mortality. The accumulation is more rapid at higher temperature, such that LC50 at a given (short) exposure time decreases with increasing temperature.

The LC50 of an aromatic in the oil mixture varies with exposure time and temperature according to:

(2) $$LC50_\infty = LC50_t \, (1 - e^{-\varepsilon t})$$

(3) $$\log_{10}(\varepsilon) = \varepsilon_1 - \varepsilon_2 \log_{10}(K_{ow})$$

(4) $$d\varepsilon / dT = \tau \, T$$

where t is time of exposure, $LC50_t$ is LC50 at time t, $LC50_\infty$ is LC50 at infinite time of exposure, K_{ow} is the octanol-water partition coefficient, $\varepsilon_1 = 1.47$ and $\varepsilon_2 = 0.414$, T = temperature (C), and $\tau = 0.11$ (French McCay, 2001).

LC50s for monoaromatics and PAHs from the literature were corrected for time and temperature of exposure to calculate $LC50_\infty$. The QSAR (Quantitative Structure Activity Relationship) regression for narcotic chemicals, specifically the aromatics in oil, was developed:

(5) $$\log_{10}(LC50_\infty) = \log_{10}(\phi) + \gamma \log_{10}(K_{ow})$$

For 278 bioassays on individual aromatics, the slope and intercept of the regression are: $\log_{10}(\phi) = 4.8926$ and $\gamma = -1.0878$. This QSAR describes the mean response for all species (i.e., the response of the average species). The slope of this relationship is constant for all species (see DiToro et al., 2000 for theory). The intercept varies by species, with 95% of species falling within the range $\log_{10}(\phi) = 3.9704$ (sensitive species) and $\log_{10}(\phi) = 5.8147$ (insensitive species). Equation (5) may be used to

estimate $LC50_\infty$ for any aromatic, assuming an appropriate intercept for the species of concern.

The SIMAP model takes into account the time and temperature of exposure, using the rearrangement of equation (2): $LC50_t = LC50_\infty / (1- e^{-\varepsilon t})$ to correct the LC50. Time of exposure is evaluated by tracking movements of organisms relative to toxic concentrations (greater than the concentration lethal to 1% of exposed organisms, LC1, approximated as 1% of $LC50_\infty$). Stationary or moving Lagrangian tracers that represent organisms record the concentrations of exposure over time. Exposure time is the total time concentration exceeds LC1. The concentration is the average over that time. The percent mortality is then calculated using the log-normal function centered on $LC50_t$.

In the SIMAP fates model (as in the NRDAM/CME, French et al. 1996, 1999), crude oils and petroleum products are represented by five components:

1. MAHs: BTEX (benzene, toluene, ethylbenzene, and xylene) and some of the substituted benzenes, which are volatile and soluble in water;
2. PAHs: 2-4 ring PAHs (and the more hydrophobic substituted benzenes), semi-volatile and moderately soluble in water;
3. non-aromatic volatiles (boiling points $<180^oC = 356^oF$);
4. non-aromatic semi-volatiles (boiling points 180-380^oC); and
5. a residual fraction, which is neither volatile nor soluble in water (boiling points $>380^oC = 716^oF$).

BTEX is very soluble in water, and so exposure concentrations in water can be high. However, BTEX is only moderately hydrophobic and so relatively low in toxicity. It is also very volatile. Thus, the BTEX rapidly volatilizes reducing exposure concentrations. For these reasons, the impact of BTEX after a spill is typically low and of short duration.

PAHs and many of the alkyl-substituted benzenes are less soluble than BTEX, but do dissolve in significant quantities into the water. Thus, they are bioavailable. Because they are more hydrophobic than BTEX, they more strongly partition into the lipids in membranes and tissues. Thus, they are more toxic and can have significant impacts on aquatic organisms.

Lower-molecular-weight aliphatic hydrocarbons (e.g., alkanes and cycloalkanes with boiling points less than about 380^oC) may also contribute to toxicity after an oil spill. However, the aliphatics are more volatile (have higher vapor pressure) and less soluble than aromatics of the same molecular weight (Mackay et al. 1992b,c,d) and would be more readily lost to the atmosphere from surface waters. They are also less toxic than the aromatics of similar molecular weight (French 2000; French McCay, 2001).

The residual fraction in the model is composed on non-volatile and insoluble compounds that remain in the "whole oil" that spreads, is transported on the water surface, strands on shorelines, and disperses into the water column as oil droplets or remains on the surface as tar balls. This is the fraction that comprises black oil, mousse, and sheen.

The $LC50_{mix}$ of the aromatic mixture is calculated using equation (1), including those aromatics that are measured in the oil and dissolved in the water (with $\log(K_{ow}) \leq 5.6$) for long enough for exposure to aquatic organisms to be significant. Typically, only the PAHs are dissolved in sufficient quantity and remain in the water long enough for their TU values to be significant. The biological effects model uses the calculated ΣPAH (or ΣBTEX + ΣPAH if BTEX is significant) and the estimated $LC50_{mix}$, corrected for time and temperature of exposure, to estimate mortality to aquatic biota. Typically, the appropriate $LC50_{mix}$ is for average sensitivity for most species, as specific data are not available for all species. However, for certain sensitive species the 2.5^{th} or 97.5^{th} percentile $LC50_{mix}$ is more appropriate. Categorization of species as sensitive, average or insensitive is based on bioassay data reviewed in French McCay (2001).

2.4 Quantification of Fish and Invertebrate Injury as Lost Production

The biomass (kg) of animals killed represents biomass that had been produced before the spill. In addition to this injury, if the spill had not occurred, the killed organisms would have continued to grow until they died naturally or to fishing. This lost future (somatic) production is estimated and added to the direct kill injury. The total is the total production lost.

The loss is expressed in present day (i.e., present year) values using a 3% annual discount rate for future losses. Restoration should compensate for this loss. The scale of restoration needed is equivalent to production lost when both are expressed in values indexed to the same year (i.e., the present year).

The injury to be compensated is the interim loss. Interim losses are injuries sustained in future years (pending recovery to baseline abundance) resulting from the direct kill at the time of the spill. Interim losses potentially include the following:
Lost future uses (ecological and human services) of the killed organisms themselves;
Lost future (somatic) growth of the killed organisms (i.e., production foregone, which provides additional services);Lost future reproduction, which would otherwise recruit to the next generation.

The approach here is that the injury includes the direct kill, plus the lost somatic growth of the killed organisms, both of which would have provided services. Because the impact on each species, while locally significant, is relatively small compared to the scale of the total population in the area (e.g., as estimated in French et al, 1996c), it is assumed that density-dependent changes in survival rate are negligible, i.e., changes in natural and fishing mortality of surviving animals are assumed not to compensate for the killed animals during the natural life span of the animals killed.

It is also assumed that the injuries were not large enough to significantly affect future reproduction and recruitment in the long term. It is assumed that sufficient eggs will be produced to replace the lost animals in the next generation. The numbers of organisms affected, while locally significant, are assumed to be relatively small portions of the total reproductive stock. Given the reproductive strategy of the species involved to produce

large numbers of eggs, of which only a few survive, it is assumed that density-dependent compensation for lost reproduction occurs naturally.

The services provided by the injured organisms are measured in terms of production, i.e., biomass (kg wet weight) directly lost or not produced. Among other factors, services of biological systems are related to the productivity of the resources, i.e., to the amount of food produced, the usage of other resources (as food and nutrients), the production and recycling of wastes, etc. Particularly in aquatic ecosystems, the rate of turnover (production) is a better measure of ecological services than standing biomass (Odum, 1971). Thus, the sum of the standing stock killed (which resulted from production previous to the spill) plus lost future production is a more appropriate scaler, as opposed to standing stock alone (as number or kg), for measuring ecological services.

This injury estimation method was developed and used previously in the injury quantification for the North Cape spill of January 1996 (French, 1998a,b,c; NOAA, 1998). The method makes use of the population model in the NRDAM/CME and SIMAP. Injuries are calculated in three steps:

1. The direct kill is quantified by age class using a standard population model used by fisheries scientists.
2. The net (somatic) growth normally to be expected of the killed organisms is computed and summed over the remainder of their life spans (termed lifetime production).
3. Future interim losses are calculated in present day values using discounting at a 3% annual rate.

The normal (natural in local waters) survival rates per year and length-weight by age relationships are used to construct a life table of numbers and kg for each annual age class. Lifetime production is estimated as the sum of the net (somatic) growth normally to be expected of the killed individual over the remainder of its life span. The age-class specific weight gain per year times percent expected to be left alive by the end of that year is summed over all years to calculate total lifetime production. Growth in future years is discounted 3% annually.

More specifically, lifetime production is calculated as follows. The number of animals that would have lived the following year (percent survival times initial number killed) is multiplied times average weight gain for that age class to the next annual age class; those surviving to the following year are multiplied times that next year's weight gain; and so on, until the end of the species life span. In addition, discounting of future losses is applied at a 3% annual discount rate. Thus, next year's values are 97% of present-year value, the following year's value is 94% of present value, etc.

The equations to calculate lifetime production are as follows. The lifetime production expected from a single age class i is:

$$(6) \qquad \sum N_{i,y} S_i (W_{i+1} - W_i) / (1+d)^y$$

$$\sum_{y}$$

and for all age classes is

(7) $$\sum_{i} \sum_{y} N_{i,y} S_i (W_{i+1} - W_i) / (1+d)^y$$

with number by age being calculated as

(8) $$N_{i+1,y+1} = N_{i,y} S_i = N_{i,y} e^{-(M_i+F_i)}$$

$N_{i,y}$ is the number of age class i remaining alive at the beginning of year y, S_i is the portion of age class i surviving to class i+1, W_i is the weight per individual for age class i, and d is the discount rate (=0.03).

Survival rates include accounting for natural and fishing mortality. Fishing mortality is applied (at 100%) beginning at the age of recruitment to the fishery. Annual survival is calculated as $e^{-(M+F)}$, where M = instantaneous annual natural mortality rate and F = instantaneous annual fishing mortality rate. Before the age of recruitment, fishing mortality F=0.

The relationships of weight and length to age are based on standard fisheries models, as used in French et al. (1996a). For growth by individuals, the von Bertalanfy equation (a variation of the Brody growth equation, Ricker, 1975) is used:

(9) $$L_t = L_\infty [1 - \exp(-K (t- t_o))]$$

where L_t is length (cm) at age t, L_∞ is length at ∞ (at an infinite age, i.e., the asymptotic maximum length), K is a constant called the Brody growth coefficient, and t_o is a constant representing the age at zero length. To calculate weight from length,

(10) $$W_t = a L_t^b$$

where W_t is weight (g, wet) at age t years and a and b are constants.

The result of these calculations is the present-day value of the direct kill plus future interim losses. Total injuries are measured as kg (wet tissue weight) of kill and production lost. It should be noted that compensation is needed for lost production of each of the individual species injured, and that losses are additive.

Discounting at 3% per year is included to translate losses in future years (interim loss) to present-day values. The discounting multiplier for translating value n years after the spill to present value is calculated as $(1+d)^{-n} = 1/(1+d)^n$, where d=0.03. Thus, the losses in future years have a discounted value in the present. In this report, all discounting is calculated based on the number of years from the year of the spill. The present day is considered the year of the spill.

3. INPUT DATA

3.1 Geographical and Model Grid

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 is generated from a digital coastline using the ESRI Arc/Info compatible Spatial Analyst program. The cells are then coded for depth and habitat type. Note that the model identifies the shoreline using this grid. Thus, in model outputs, the coastline map is only used for visual reference; it is the habitat grid that defines the actual location of the shoreline in the model.

The digital shoreline, shore type, and habitat mapping was obtained from the ArcView Environmental Sensitivity Index (ESI) coverage for Patuxent River area. The gridded habitat type data are shown in Appendix A-2. The grid scale resolution is a cell size of 33 m E-W by 20 m N-S.

Within a grid, habitats are designated as landward or seaward. Landward portions are the inlets or shallower portions of the grid. The seaward portion is the main part of the water body. This designation allows different biological abundances to be simulated in landward and seaward zones of the same habitat type (e.g., open water with sand bottom). The biological database is coded to landward or seaward by species (see French et al., 1996a, c). (See Section 3.8 below for explanation of assignment of species abundances to seaward = open water and landward = shoal waters.)

Ecological habitat types (Table 3-1) are broadly categorized into two zones: intertidal and subtidal. Intertidal habitats are those above spring low water tide level, with subtidal being all water areas below that level. Intertidal areas may be extensive, such that they are wide enough to be represented by an entire grid cell at the resolution of the grid. These are typically either mud flats or wetlands, and are coded 20 (seaward mudflat), 21 (seaward wetland), 50 (landward mudflat) or 51 (landward wetland). All other intertidal habitats are typically much narrower than the size of a grid cell. Thus, these fringing intertidal types (indicated by F in Table 3-1) have typical (for the region, French et al., 1996a) widths associated with them in the model. Boundaries between land and water are fringing intertidal habitat types. On the waterside of fringing intertidal grid cells, there may be extensive intertidal grid cells if the intertidal zone is extensive. Otherwise, subtidal habitats border the fringing intertidal.

Table 3-1. Classification of habitats. Seaward (Sw) and landward (Lw) system codes are listed. (Fringing types indicated by (F) are only as wide as intertidal zone in that province. Others (W = water) are a full grid cell wide and must have a fringing type on the land side.)

Habitat Code (Sw,lw)	Zone	Ecological Habitat	F or W
1,31	Intertidal	Rocky Shore	F
2,32		Gravel Beach	F
3,33		Sand Beach	F
4,34		Fringing Mud Flat	F
5,35		Fringing Wetland (Saltmarsh)	F
12,42	Subtidal	Sand, silt, and/or mud bottom	W
18,48	Intertidal	Man-made, Artificial	F
20,50		Extensive Mud Flat	W
21,51		Extensive Wetland (Saltmarsh)	W

The intertidal habitats were assigned based on the shore types in digital Environmental Sensitivity Index (ESI) maps distributed by NOAA HAZMAT (CD-ROM) and supplied to ASA by Research Planning Inc. (RPI). This data was gridded using the ESRI Arc/Info compatible Spatial Analyst program. Open water areas were defaulted to general soft bottom, as open water bottom type has no influence on the model results. Where data are missing, shore types are defaulted as in Table 3-2. Habitats in the shallower portions of the river and creeks are designated as landward.

Table 3-2. Default fringing intertidal habitat type, given adjacent subtidal or extensive intertidal habitat type.

Subtidal or Extensive Intertidal Habitat	Fringing Intertidal Habitat
Seagrass Bed (47)	Sand Beach (33)
Subtidal Sand Bottom (41)	Sand Beach (33)
Extensive Mudflat (50)	Fringing Mudflat (34)
Extensive Wetland (51)	Fringing wetland (35)

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 Appendix A-3.

3.2 Environmental Data

The model uses hourly wind speed and direction for the time of the spill and simulation. Wind data (hourly speed and direction) are from Thomas Pt. MD near Annapolis (NOAA station TPLM2). Wind speed and direction data are listed in Appendix C.

Temperature is 14^{o}C and salinity is 8 ppt, based on measurements in the Patuxent River on Apr. 4, 2000, east of mouth of Indian Creek (data provided by the Trustees). Water 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 is insignificant. Salinity has little influence on the fate of the oil.

Suspended sediment is assumed 10 mg/l, a typical value for coastal waters (French et al., 1996a). 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 is assumed 1 m^2/sec. The vertical diffusion (randomized mixing) coefficient is assumed 0.0001 m^2/sec. These are reasonable values based on Okubo (1971) and modeling experience. The vertical diffusion coefficient used keeps the water column well mixed, and so variation of this parameter has no significant impact on the results.

3.3 Currents

3.3.1 Tidal and Other Currents

Currents have significant influence on the trajectory, and are important data inputs. Tidal currents were manually interpolated from tidal chart vectors in a commercial (digital) tidal data program (Tides & Currents ®). They are assumed rectilinear (in and out equal and opposite) and with two equal high tides per 25 hrs (M2). Downstream freshwater-driven river flow was not included. Appendix B contains maps of the resulting current data used in the oil model simulations.

3.3.2 Wind-driven Surface Currents

Wind-driven surface currents are calculated within the SIMAP fates model, based on local wind speed and direction. 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).

Wind drift speed and angle were studied in detail by Youssef and Spaulding (Youssef, 1993; Youssef and Spaulding, 1993, 1994). Wind drift speed is a percentage of wind

speed over the water, highest at low wind speed and decreasing as wind speed increases. The range of drift speed for winds up to 20 kts (averaged over time) is 2-4% of wind speed. At 10 kts or less, which prevailed during the spill event, the percent of wind speed is about 3.5-4% at the water surface, decreasing to 2% at 0.1m below the surface. The angle to the right of down wind is highest at low wind speed, on the water surface ranging from about 20^o-30^o at 10 kts or less. The drift speed decreases, and the drift angle increases, deeper into the water column.

Youssef and Spaulding (Youssef, 1993; Youssef and Spaulding, 1993, 1994) developed a set of equations to describe the percent of wind speed and angle as functions of wind speed and depth in the water. This algorithm has been incorporated into SIMAP. The wind drift is applied to the upper 5 meters of the water column. This Youssef and Spaulding algorithm was used in model runs for surface wind drift.

3.4 Oil Characteristics and Toxicity

The spill was of a mix of No. 6 and No. 2 fuel. Oil characteristics were specified using measurements of the source oil, as available. Other characteristics were assumed to be as for defaults from the NRDAM/CME database (French et al., 1996b). As a mix of No. 6 and No. 2 fuel would be closest to No. 5 fuel (of those available in the NRDAM/CME database), characteristics of No. 5 fuel were used as default data. Only the percentage of soluble aromatics in the fuel has a significant impact on the model results.

The needed evaporation constants are not from the NRDAM/CME database (French et al., 1996b) (nor were they measured on the source oil). The A, B, T_G and T_o constants for the Stiver and Mackay (1984) evaporation algorithm (equations 10-11 therein) were calculated from boiling curve data for the most similar fuel (bunker) obtained from Environment Canada's oil catalog (Whiticar et al., 1992). T_G is the gradient of the distillation curve; T_o is the initial boiling point of the distillation curve; and A and B are constants (see Stiver and Mackay, 1984).

The percentage of aromatic volatiles has a significant influence on the model results. The volatiles evaporate and dissolve, decreasing the surface oil volume over time and inducing water column contamination. The aromatic content was as measured and reported by ENTRIX for the source oil sample: 3.348% PAH with log(kow)<5.6 (which is bioavailable and so the toxic fraction). BTEX was not measured in the source oil, but is assumed to have negligible impact on water column organisms because of its high volatility. (See discussion in Section 2).

To estimate $LC50_{mix}$ values for dissolved PAHs in the water, equation (1) is used with LC50s calculated from QSARs (given in Section 2.3) and F_i values calculated from dissolved concentrations of aromatics (with $\log(K_{ow} \leq 5.6)$ in water samples taken in the field after the spill. Table 3-3 summarizes the measurements of parent and alkyl-substituted PAHs in the water samples. The $LC50_{mix}$ values based on the water samples are similar to those in Table 3-3 where dissolved aromatics are assumed in the same proportions as in the source oil. PAH concentrations in the source oil are based on an

April 19, 2000 sample from the pipeline, as reported by ENTRIX (2000, Appendix D, Table D-2). If dissolved concentrations are assumed to be at equilibrium with the source oil, the calculated LC50s (223 ppb for average and 27 ppb for sensitive species) are not in agreement with those based on the measured dissolved samples (Appendix D, Table D-3). Thus, equilibrium is not indicated, rather the dissolved concentrations are in similar proportions to the source oil concentrations.

The geometric mean of the $LC50_{mix}$ values for species of average sensitivity is 75 ppb. The lowest LC50 for sensitive species (2.5^{th} percentile species) is about 5 ppb. Thus, two LC50s were run in the biological effects model: 5 ppb and 75 ppb to estimate injuries for species of average sensitivity to PAHs and for sensitive species.

3.5 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., 1996a,b), shore holding capacity 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., 1996a). These data are used here. The shore width (intertidal zone width) was assumed 1 m.

3.6 Scenario

The following conditions were assumed. The spill site was at the Pepco pipeline in Swansons Creek at 76^{o} 42.097' W, 38^{o} 32.6555' N. The release occurred at the water surface. The spill was instantaneous on April 7, 2000 at 17:30 EDT. The volume released is estimated at 120,000 gallons.

3.7 Summary of Fates Model Inputs

Appendix D contains a list of model inputs for the SIMAP physical fates model. Note that some of the inputs are either not applicable to this case or are not used for the trajectory and fate analysis of this spill.

Table 3-3. Measured PAH concentrations in water samples [PAH], calculated LC50s using QSARs, and TU = toxic units = PAH concentration/LC50. Only PAHs with log(Kow) < 5.6 are included in [PAH] and in calculation of LC50 (for infinite exposure time) using QSARs. The range of species sensitivities is shown: ave.spp. is the mean QSAR including data for all species. Sensitive and insensitive are the extremes for 95% of species. LC50s predicted from the source oil are also given, assuming high entrainment such that oil-in-water dispersions were formed (as indicated by the high winds after the release).

Sample	Date	Time	Location	[PAH] µg/L	LC50 (µg/L) Ave. sp.	LC50 (µg/L) Sensitive	LC50 (µg/L) Insensitive	TU Ave. sp.	TU Sensitive	TU Insensitive
SW-2	4/10/00	1400	Swanson Creek	167.007	54	6	453	3.078	25.733	0.368
SW-17	4/10/00	1120	Swanson Creek	0.087	110	13	915	0.001	0.007	0.000
SW-18	4/10/00	1745	Swanson Creek	1.003	84	10	700	0.012	0.100	0.001
SW-19	4/10/00	1040	Swanson Creek	458.376	43	5	357	10.742	89.802	1.285
SW-20	4/10/00	1530	Swanson Creek	1.851	57	7	476	0.033	0.272	0.004
SW-21	4/10/00	1610	Swanson Creek	16.935	92	11	772	0.183	1.533	0.022
SW-24-3	4/12/00	1420	Cat Creek	0.187	87	10	724	0.002	0.018	0.000
SW-25-6	4/12/00	1436	Cat Creek	0.221	101	12	842	0.002	0.018	0.000
SW-26-9	4/12/00	1453	Cat Creek	1.146	135	16	1128	0.008	0.071	0.001
SW-27-12	4/12/00	1557	Persimmon Creek	6.436	52	6	437	0.123	1.029	0.015
SW-28-15	4/12/00	1614	Persimmon Creek	7.096	67	8	557	0.106	0.890	0.013
SW-29-18	4/12/00	1643	Persimmon Creek	3.897	75	9	629	0.052	0.433	0.006
SW-30-21	4/12/00	1710	Washington Creek	0.612	90	11	753	0.007	0.057	0.001
SW-31-24	4/12/00	1731	Washington Creek	0.612	90	11	753	0.007	0.057	0.001
SW-32-27	4/12/00	1746	Washington Creek	2.628	69	8	574	0.038	0.320	0.005
SW-33-30	4/12/00	1756	Washington Creek	2.278	67	8	557	0.034	0.286	0.004
SW-34-33	4/12/00	1814	Washington Creek	2.973	69	8	573	0.043	0.363	0.005
PR-SC-01-A	4/13/00	1610	Patuxent R. for bioassay	25.616	60	7	502	0.427	3.567	0.051
All			Geometric mean	2.958	75	9	624			
Source oil	4/19/00		Pepco Pipeline µg/kg =	33,479,400	45	5	378			

3.8 Biological Abundances

The model uses average number or biomass per unit area (#/km^2) in appropriate habitats (French et al., 1996a,c). The species is assigned to a behavior category according to its location and movements in the water column (i.e., small pelagic, large pelagic or demersal). The species is assumed uniformly distributed across its preferred habitats and vertically in the zone it occupies. Thus, the habitat grid defines the habitat map, and so the abundance of each species. Demersal species remain within 1 m of the bottom, while pelagic species move throughout the water column at slow (small pelagics) or fast (large pelagics) swimming speeds.

In the model, fish and invertebrates abundance varies by landward open water, seaward open water, and structured habitat (i.e., wetlands, reefs, and macroalgal beds, Table 3-1). In the biological database input to the model, the abundances are for fished stocks and the biomass includes those animals greater than the age of recruitment to fishing (or one year of age for small fishes such as killifish and silversides). Within the biological effects model, the age/size distribution is computed from fishery modeling parameters (natural and fishing instantaneous mortality rates, length as a function of age, and weight-length relationships), such that the mortality is calculated for all age classes from age 1 year up (and assuming the various age classes live in the same habitat in that age structure). Young-of-the-year mortality is quantified separately. The biological database includes number of age 1-year (365 day old) individuals per km^2. Thus, young-of-the-year mortality is for only those that would have survived their first year if not for the spill.

For this assessment, the species abundances for April 2000 (Table 3-4) were assumed to be as described in ENTRIX (2002). Open water abundances apply to >1.6 m water, designated as "seaward" in the model. Shoreline/shoal abundances apply to <1.6 m water, designated as "landward" or "structured" (=wetland) in the model.

The life history parameters (growth, mortality) for these species were assumed to be as included in the NRDAM/CME (French et al., 1996c). The life history parameters are specified by species category (generally family or similar taxa, identified by codes with the same value in hundreds), using data for one representative species in the group. Thus, for example, life history parameters for white perch are assumed to be as for striped bass, and Atlantic silverside life history parameters are used for all species in their group (inland silverside, striped killifish, mummichog, spottail shiner, and less common finfish).

Table 3-4. Abundance estimates of fish and invertebrates for April 2000 (ENTRIX, 2002).

Model Code	Species	Open Water Standing Stock kg/km^2	Shoreline/Shoal Standing Stock kg/km^2
102	Bay anchovy	.04	.04
302	Blueback herring	.01	.01
405	Atlantic menhaden	470	470
1001	Atlantic silverside	5.4	5.4
1009	Striped killifish	-	1.2
1010	Mummichog	-	4.5
1011	Spottail shiner	-	.07
1012	Inland silverside	-	.02
1013	Less common finfish species	54	6.7
3101	Striped Bass (adult)	269	269
3101	Striped Bass (subadult)	1.8	1.8
3102	White Perch	2,271	341
3303	Atlantic Croaker	1,369	-
3401	American eel	432	7.4
3530	Hogchoker	1,932	32
4504	Brown bullhead	11	11
4601	Blue crab	351	351
4822	Horseshoe crab	163	163
5901	Oysters (ash free dry weight)	437	-

Young-of-the-year abundance estimates were assumed to be as those derived in the NRDAM/CME database (French et al., 1996c). The NRDAM/CME contains mean monthly abundances for 77 biological provinces in US coastal and marine waters. The data for the upper Chesapeake Bay province (16) in April were used. Presence of young-of-the-year in the Patuxent River in April was confirmed based on Stone et al. (1994). Table 3-5 shows the species that were used in the young-of-year calculations, and the stages in which these species have been found to occur in April in the Patuxent River. Those species noted by Stone et al. (1994) as present as juveniles are assumed to include individuals less than 1 year in age, based on the duration of the larval stages for the species of concern (French et al, 1996c). Abundance and weight per individual are in Table 3-6. Note that the young-of-the-year abundances are for those animals that would survive to 365 days of age, but the model treats them as behaving according to their age and stage in April (see French et al., 1996a for further explanation).

White perch young-of-the-year abundance estimates were not included in the NRDAM/CME database (French et al., 1996c). Thus, white perch young-of-the-year

were estimated from striped bass young-of-the-year, using the ratio of abundance for the two species in the ichthyoplankton surveys performed in the Patuxent River in 2000 (ENTRIX, 2002). The ratio was 4.481 based on 16,310 white perch collected versus 3,640 striped bass young-of-the-year.

Table 3-5. Young-of-the-year species used in the SIMAP biological model, and the stages in which these species occur in April in the Patuxent River (Stone et al., 1994).

Species	Present as Eggs	Present as Larvae	Present as Juveniles
River herring (blueback)	*	*	
Striped bass	*	*	*
White perch	*	*	*
Atlantic croaker			*
Hogchoker	*	*	*
Blue crab			*

Table 3-6. Young-of-the-year abundance (as age 365-day equivalents) and weight per individual at 365 days of age (French et al., 1996c; province 16 therein) used as input to the model runs.

Species	Abundance in Water <1.6m (#/km2)	Abundance in Water >1.6m (#/km2)	g per individual
River herring (blueback)	3.36	0	47
Striped bass	14.27	7.84	12
White perch	63.94	35.11	12
Atlantic croaker	1265	1265	120
Hogchoker	2.51	2.43	50
Blue crab	34,470	0	215

4. OIL FATES MODEL RESULTS

Modeling of the trajectory and fate of the oil was performed using SIMAP, varying the horizontal turbulent diffusion coefficient (1, or 5 m^2/sec) to evaluate sensitivity to this assumption. The fates model results of surface oil were visually compared to observed slick locations (e.g., from overflights), scat reports, shoreline oiling maps, and other field data, as available (ENTRIX, 2000). It was found that the model was not sensitive to this assumption, but that 1 m^2/sec provided a better fit to the observations.

Appendix F contains trajectory plots. The oil trajectory snapshots show the modeled "spillet" centers of mass for surface oil (black), subsurface entrained oil droplets (blue), and dissolved aromatics (PAHs, green). These figures do not weigh the dots for mass, and the dots are of varying mass.

Total hydrocarbon concentrations, including the slicks on the water surface, are plotted in Appendix G. The maps show total hydrocarbons on and in the water after the spill. Concentrations are calculated for a grid (50 X 50 cells horizontally, 5 layers vertically) sized to just cover the plume at the time of the output. The surface layer of that grid contains the surface slicks and entrained oil immediately under the slick (near the water surface). The maps show the vertical maximum concentration at each (latitude-longitude) location, and thus the surface slicks. The cross-section inserts on the maps show subsurface concentrations along the dotted line in the planar view map.

In this case, each of the 5 vertical layers is 0.2m thick. Thus, 1 ppb = 1 mg/m^3 = 0.2 mg/m^2 for slicks in the surface layer. Table 4-1 gives approximate thickness ranges for surface oil of varying appearance. Dull brown sheens are about 1000 mg/m^2 thick. Rainbow sheen is about 200-800 mg/m^2 and silver sheens are 50-800 mg/m^2 thick (NAS, 1985). Thus, only >1000 ppb would be visible.

Table 4-1. 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

Appendix H shows the amount of oil accumulated on shorelines and sediments for the simulation, as mass of total hydrocarbons per unit area (averaged in each habitat grid cell). No shoreline cleanup was simulated in the model. Thus, oil simply accumulates and remains on the shore.

The oil trajectory (Appendix F) agrees with the field observations, as described in several documents and indicated by the shoreline oiling maps (ENTRIX, 2000). The following description of the fate of the oil is based on the model trajectory and driving forces (wind). From April 7 at 17:30 until April 8 at 15:30, the wind was strong and from the southeast, holding the oil in the marsh near the pipeline break. From April 8 at 15:30 until April 10 at dusk, strong northwest and west blew the oil out of Swanson Creek across the Patuxent River to Buena Vista. The strong winds also entrained oil into the water (based on the entrainment data from Delvigne and Sweeney, 1988), increasing the aromatic concentrations in the water. The aromatics (PAHs, Appendix I) were swept downstream and up and down the river with the tides. Overnight April 10-11 and through April 11, the oil was blown to the west bank and downstream by north and northeast winds. The trajectory shows more oil in Indian Creek than Trent Hall Creek. However, the addition of some freshwater flow to the simulation would move the oil downstream slightly and more into Trent Hall Creek and southward. By the afternoon of April 12, the simulation shows oil reaching Cats Creek and Jack's Bay.

Dissolved PAH concentrations in the water are shown in Appendices I and J. The model-predicted PAH concentrations (Appendix I) generally agree with the data in Table 3-3 in magnitude of concentration. The only concentrations >10 ppb are in Swanson's Creek (before April 10) and spottily in the Patuxent River (10-100 ppb). Persimmon and Washington Creek areas are 1-10 ppb on April 12. The details of the PAH peaks in the simulation will not line up with sample measurements exactly, as the current data used in the modeling are not of sufficient accuracy to predict concentrations in fine resolution.

Note that the magnitude of concentrations and water volumes contaminated at each level are the influential factors in the injury quantification. The exact locations of PAH peaks do not influence the results, as the aquatic biota are assumed uniformly distributed across the habitats they occupy. Thus, the distribution of contamination between habitat categories where abundance varies is the important factor to the resulting quantification of injuries.

Appendix E shows the mass balance of oil in the simulation. The graph shows, as a function of time since the release start, percent of total mass spilled on the water surface, in the water column, on shorelines, in the sediment, in the atmosphere, and degraded. Quantitative measurements of oil mass cleaned up (not including water and debris) are not available. Thus, cleanup was not included in the model simulations.

5. BIOLOGICAL EFFECTS MODEL RESULTS

Fish and invertebrate injuries were estimated using the biological effects model. The percent loss figures (Appendix K) show the model's percent mortality estimates for organisms stationary on the bottom and exposed to bottom water. Two LC50s were used: (1) that for a species of average sensitivity (75 μg/L) and (2) that for a sensitive species (2.5 percentile) for the sample with the most toxic mixture (5 μg/L).

The points on the maps are locations of Lagrangian tracers (or "particles" or elements) used to record the exposure concentrations over time for the animals they represent (see Section 2.3). For the average species, impacts are slight and restricted to shallow waters. However, for the sensitive species impacts are estimated as covering a significant area. Most of the impact is in Swanson Creek, Indian Creek and Trent Hall Creek.

Tables 5-1 to 5-3 list the estimated kills of fish and invertebrates, assuming the abundances described in Section 3.8 and using the two assumptions for LC50: (1) species of average sensitivity where $LC50_{\infty}$ for the oil PAH mixture = 75 μg/L and (2) sensitive species where $LC50_{\infty}$ for the oil PAH mixture = 5 μg/L. As none of the species of concern have been shown to be sensitive to PAHs (French McCay, 2001), the results for the LC50 = 75 mg/L are the best estimates. Table 5-1 is for young-of-the-year, as number of animals that would have survived to 365 days (1 year) of age otherwise. Young-of-the-year kills are included in the kills of Tables 5-2 and 5-3. The biomass equivalent of the direct kill of YOY is 757 kg.

Tables 5-2 and 5-3 also list the calculated production foregone and total injury (= direct kill plus production foregone) in kg. Assuming the model input data and average species sensitivity to PAHs, future growth of the killed animals, had there not been a spill, would total 979 kg (the production foregone). Thus, the model-estimated total injury to fish and invertebrates (assuming average species sensitivity, Table 5-2) is 2,460 kg.

Table 5-1. Model estimates of kills of eggs, larvae and juveniles (young-of-the-year) as age one-year equivalents (#, kg), assuming average (75 μg/L) or sensitive (5 μg/L) species LC50s.

	LC50 = 75 μg/L		**LC50 = 5 μg/L**	
Species	**#**	**kg**	**#**	**kg**
Blueback herring	0.4	0.02	5.9	0.28
Striped bass	2.2	0.03	46	0.54
White perch	10	0.12	205	2.43
Atlantic croaker	1,844	221	4,104	492
Hogchoker	0.4	0.02	7.2	0.36
Blue crab	2,494	535	40,396	8671
Total	**4,351**	**757**	**44,764**	**9167**

Table 5-2. Model estimates of fish and invertebrate losses totaled for all age classes, assuming average species LC50 = 75 μg/L.

Fishery species	Kill (kg)	Production Forgone (kg)	Total Injury (kg)
Bay anchovy	0.01	0.0	0.01
Blueback herring	0.02	0.10	0.12
Atlantic menhaden	120	50	170
Silversides, etc.	4.6	0.7	5.2
Striped bass	60	81	141
White perch	252	343	595
Atlantic croaker	329	317	645
American eel	17	20	38
Hogchoker	84	70	154
Brown bullhead	1.7	0.7	2.4
Blue crab	579	44	623
Horseshoe crabs	32	51	83
Oysters, dry weight	2.13	0.75	2.87
Total	**1,481**	**979**	**2,460**

Table 5-3. Model estimates of fish and invertebrate losses totaled for all age classes, assuming sensitive species LC50 = 5 μg/L for all species (a worst case assumption).

Fishery species	Kill (kg)	Production Forgone (kg)	Total Injury (kg)
Bay anchovy	0.09	0.01	0.1
Blueback herring	0.3	1.6	1.9
Atlantic menhaden	1,845	765	2,610
Silversides, etc.	83	12	95
Striped bass	990	1,360	2,350
White perch	5,148	7,002	12,150
Atlantic croaker	2,476	1,365	3,841
American eel	303	359	661
Hogchoker	1,470	1,226	2,696
Brown bullhead	29	11	40
Blue crab	9,364	713	10,077
Horseshoe crabs	503	818	1,321
Oysters, dry weight	4.8	1.7	6.4
Total	**22,216**	**13,635**	**35,851**

6. SUMMARY OF INJURY QUANTIFICATION

Table 6-1 summarizes the total injuries to fish and invertebrates, with the "silversides, etc." category broken down to individual species. This includes the data in Table 5-2 calculated by the model (assuming average sensitivity to PAHs for all species) and an additional 200 mummichogs collected in the marsh after the spill (USFWS, 2000, Animal Mortality Log). The 200 mummichogs were assumed killed in the marsh in areas where exposure was not evaluated by the model. Thus, these animals are in addition to the model injury. They are assumed to be of the same size and result in the same injury per animal as the modeled injury of mummichogs.

Table 6-1 summary of the total injuries to fish and invertebrates.

Fishery species	**Kill (kg)**	**Production Forgone (kg)**	**Total Injury (kg)**
Bay anchovy	0.01	0.0	0.01
Blueback herring	0.02	0.1	0.12
Atlantic menhaden	120	50	170
Atlantic silverside	1.39	0.21	1.6
Striped killifish	0.30	0.05	0.35
Mummichog (model)	1.1	0.2	1.3
Mummichog (marsh)	3.3	0.5	3.8
Spottail shiner	0.02	0.00	0.02
Inland silverside	0.01	0.00	0.01
Less common finfish	1.7	0.3	2.0
Striped bass	60	81	141
White perch	252	343	595
Atlantic croaker	329	317	645
American eel	17	20	38
Hogchoker	84	70	154
Brown bullhead	1.7	0.7	2.4
Blue crab	579	44	623
Horseshoe crabs	32	51	83
Oysters, dry weight	2.1	0.8	2.9
Total	**1,485**	**979**	**2,464**

Assuming the model input data and average species sensitivity to PAHs, and including the field collected mummichogs killed in the marsh, the best estimate of total injury to fish and invertebrates is 2,464 kg. This total injury includes: (1) the biomass equivalent of the direct kill, equal to 1,485 kg; and (2) future growth of the killed animals, had there not been a spill, totaling 979 kg (the production foregone).

Restoration should provide 2, 464 kg of equivalent quality fish and invertebrate biomass to compensate for the lost fish and invertebrate production. Equivalent quality implies same or similar species with equivalent ecological role and value for human uses. The equivalent production should be discounted to present-day values to account for the interim loss between the time of the injury and the time restoration provides equivalent ecological and human services.

The accuracy of the biological injury assessment depends primarily on the accuracy of (1) the fates model results and (2) the biological abundance data input to the model. With more accurate current data and details of the fates model simulations, the percent losses would change by less than an order of magnitude. Since most of the injuries were in shoal waters (<1.6m) where abundance was assumed uniformly distributed, the quantified injury would not change significantly if the details of the oil hydrocarbon movements were more closely simulated.

The injuries quantified are sensitive to the assumed LC50. However, none of the species involved have been shown to be particularly sensitive to PAHs. Thus, the injuries are unlikely to be underestimated by this analysis based on toxicity to average species.

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.

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