FINAL REPORT

Preassessment NRDA Analysis: Preliminary Modeling of the Fates and Effects of Oil Released from the *M/V New Carissa* in February-March 1999

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Table of Contents

1. Introduction 1

2. Overview Of Model Algorithms 1

3. Input Data 4
   3.1 Geographical 4
   3.2 Environmental Data 4
   3.3 Oil Characteristics 5
   3.4 Scenario 8
   3.5 Biological Data 8

4. Results and Discussion 9

5. Conclusions 13

6. References 15

Appendices
   Appendix A: Unified Command Decision Memo
   Appendix B-1: Wind and wave data
   Appendix B-2: Wind data used for model runs for *New Carissa* spill
   Appendix C: Habitat and bathymetry data and grids used in the modeling
   Appendix D: Inputs for model runs
   Appendix E: Properties assumed for *New Carissa* oil
   Appendix F: Biological data base – NRDAM/CME PROVINCE 47, No. CA-Oregon Shelf
   Appendix G: Results of Model Runs: Physical Fates Simulations
   Appendix H: Results of Model Runs: Injuries
   Appendix I: Estimation of the Toxicity of Dissolved *New Carissa* PAHs

List of Figures

Figure 1. Effect of temperature on LC50.

Figure 2. Relationship of LC50 to dissolved PAH concentration.

List of Tables

Table 1. Summary of total PAH content of source oil samples (Payne and Driskell, 1999).

Table 2. *New Carissa*: Estimated volumes burned, spilled, and remaining on the vessel, assuming 70,000 gal. Spilled, minimum burn, and maximum remaining volumes. Volume and specific gravity estimates for each tank are based on the Unified Command Decision Memo (Appendix A).

Table 3. Temperature effect on LC50 (ppb), for 96-hour LC50s and PAHs. The LC50s are derived using QSARS based on literature data for PAHs, as developed in French (1998).

Table 4. LC50 at varying time of exposure and temperature, which is equivalent to the standardized LC50 at 25C, 96 hours exposure.

Table 5. All New Carissa water TPAH data available as of July 9, 1999 (from Payne and Driskell, 1999).

Table 6. Summary of dissolved PAH samples taken 12 February 1999 near the New Carissa, and calculated PAH 96-hour LC50 (ppb) at 25C and 9C.

Table 7. Cases run for both physical fates and biological effects.

Table 8. Summary of model estimates of fish and shellfish injuries.

Table 9. Fish and shellfish losses for scenario Caris16.

1. INTRODUCTION

The *M/V New Carissa* initially came aground just outside Coos Bay, Oregon, and began releasing oil on 8 February 1999. After weeks of response and cleanup operations, the bow section was towed out to sea on 1 March 1999. The bow section broke away and re-grounded just outside the mouth of Alsea Bay, 60 nm north of Coos Bay, on 3 March 1999. A second, smaller release of oil occurred at this grounding site. Details of the spill chronology and other data collected after the incident are available in the *New Carissa* Preassessment Data Report (RPI, 1999), to be released later this year.

This report summarizes modeling of the fates and effects of the first release, and provides an estimate of the injury caused by acute toxicity in the water column (fish and shellfish). The second release was much smaller than the first, and a decision was made not to model the fate and effects of that release.

This preliminary modeling is designed to evaluate the scope of the acute injuries, as part of the NRDA preassessment analysis. Local event-specific wind data were used in the modeling. However, the current data used were sketches of general tidal current patterns for Coos Bay and southward wave-driven flow along the outer coast. Estimates of wind and wave-driven currents were not available for this analysis, but were judged to have significant influence on the location of the oil over time. The biological abundance data used were the default data available in the CERCLA type A model, the NRDAM/CME. If needed, detailed current and local biological abundance data could be incorporated later to refine the estimates of injury. However, it is not expected that the resulting water column injuries would differ greatly from those estimates contained herein.

The modeling approach was as follows:

- Estimate fates and concentrations of oil components in space over time
- Verify the fates model predictions by comparison with observed oil distributions and concentrations.
- Estimate biological injuries caused by the oil distributions and concentrations.

The model used is a modification of the type A model, such that surf entrainment may be simulated and site-specific data may be used.

2. OVERVIEW OF MODEL ALGORITHMS

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 biological effects model, (3) input tools for oil physical, chemical and toxicological data, (4) input tools for environmental data, (5) tools to grid and enter geographical data, (6) input tools for biological data, (7) a response module to analyze effects of response strategies, (8) graphical visualization tools for outputs, and (9) 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 this model 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. The model is three-dimensional, using a latitude-longitude grid for environmental 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 low molecular weight aromatics (1 to 3-ring PAHs) which cause toxicity in the model, other volatiles, and non-volatiles.

The biological model estimates acute toxic response of marine biota to the oil. Mortality of fish, shellfish, and their eggs and larvae are computed as a function of temperature, concentration, and time of exposure. Lost productivity of lower trophic levels is assessed as resulting from acute exposure. Lost catch missing from the affected populations are estimated for present and future years, using estimated natural and harvest mortality rates which would occur in the absence of the spill.

SIMAP includes the physical fates and biological effects models in the NRDAM/CME, with several changes and additions, summarized below. A complete description of the fates model algorithms is in 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 were made to increase model resolution, allow modification and site-specificity of input data, allow incorporation of temporally-varying 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). 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 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 (about 1.5 times wave height). SIMAP tracks entrained droplets separately from surface slicks.
- 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 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. 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 dispersion (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.
- Dispersion (mixing) coefficients: The NRDAM/CME uses the model of Okubo (1971) to relate the horizontal dispersion 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 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, dispersion coefficients, and concentration field calculations. The biological models use identical algorithms. The NRDAM/CME biological model code was edited only to allow user entry of habitat grid, biological abundances, and oil parameters. Additional model output formats were also added to SIMAP. Both models in SIMAP may use site-specific data inputs, whereas in the NRDAM/CME, default databases are used. The changes in the databases and model defaults cause the largest changes in the results produced by the two models. Database changes are discussed in the next section.

3. INPUT DATA

3.1 Geographical

The digital shoreline used was obtained from the NOAA National Ocean Service database and imported into SIMAP. A model grid was generated and filled with depth and habitat mapping data, the results of which are shown in Appendix C.

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 for estuarine and near shore areas. The depth soundings were gridded using the ESRI Arc/Info compatible Spatial Analyst program. The resulting gridded depth data are shown in Appendix C.

The shoreline habitats in the affected area were assigned based on the digital Environmental Sensitivity Index (ESI) maps distributed by NOAA HAZMAT (CD-ROM for the West Coast of the US). Open water areas were defaulted to sand bottom, as open water bottom type has no influence on the model results.

3.2 Environmental Data

Standard meteorological data were acquired from the National Data Buoy Center Internet site for the Cape Arago, OR C-MAN station at 43.34 ^{o}N, 124.38 ^{o}W, and the nearest NDBC buoy, number 46050, “Yaquina Bay”, off Newport, to the north at 44.62 ^{o}N, 124.53 ^{o}W.

Winds at Cape Arago have a strong southerly component, only rarely shifting around to the north for brief periods. Typical mean hourly winds (two minute average) were on the order of 5 meters per second (10 knots) around the period that spilled oil was first reported on February 8. This condition continued over much of the following week with two periods of heavier winds of about 10 m/s, with peaks of 15 m/s, on the night of February 9 and over much of February 12. These heavier winds predominated the period of the evening of February 15 through February 18, as well as after February 20. Wind speed conditions at the more remote buoy 46050 were very similar to those at Cape Arago over most of this period, but had generally shorter periods when the winds abated to 5 m/s or less.

The C-MAN stations here do not report wave height data, so the data from the offshore buoy were relied upon for some measure of sea state. Significant wave heights over the period of February 8 to 21 were generally around 5 meters, with lower values of 2 to 4 meters occurring during the periods of lesser wind speeds at Cape Arago, and with greater values of 6 meters and up during the periods of greater wind speeds.

Wind speed and direction data used in the simulations are tabulated and plotted in Appendix B. Surface water temperature data at buoy 46050 was 9°C. This temperature is assumed for both the water surface and the air immediately above the water.

A tidal current data file for Coos Bay was developed using data available from the tidal station at the entrance to Coos Bay and using maps of the results of tidal simulations generated by NOAA HAZMAT during the response as general guidance. M2 (semidiurnal, i.e. 12.4 hrs per tidal cycle) tidal currents were used in simulations of the spill event, since that is the dominant tidal signal in Oregon waters. High tide was at 05:23 hours at Charleston (just inside Coos Bay entrance) on 8 February 1999.

Two versions of the current data file were prepared; each with southward directed background (constant) current vectors to simulate the wave-driven transport south along the shore towards the entrance of Coos Bay. Simulations were run with zero background currents (to determine the influence of the background currents on the results) and each of these versions of the background current. Tidal currents were the same in all simulations.

Suspended sediment was assumed 10 mg/l, because of the high wind and waves following the spill event. A sedimentation rate of 3 m/day was assumed, since much of the suspended sediment was sand.

Horizontal and vertical diffusion (randomized mixing) coefficients were assumed 1-10 m^2/sec in the horizontal and 0.001 m^2/sec in the vertical directions. These are reasonable values based on Okubo (1971). The vertical diffusion coefficient used keeps the water column well mixed, and so variation of this parameter has no significant impact on the results. Variation of the horizontal diffusion coefficient was examined (see below).

3.3 Oil Characteristics

The oil on board the *New Carissa* consisted of 4 distinct fuels (Payne and Driskell, 1999): 2 bunker fuels (both rated with viscosity 280 centiStokes) and 2 marine diesel oils. Payne and Driskell (1999) describe the chemical composition of the 4 fuels, as well as of the environmental samples taken after the spill. The Unified Command Decision Memo of March 12, 1999 (Appendix A) contains the density of the fuel in each tank

Physical and chemical data on bunker and diesel fuels were taken from the NRDAM/CME database (French et al., 1996b), except as noted below. The data assumed for the release are volume-weighted averages for these fuels, as listed in Appendix E. Constants for the evaporation algorithm were calculated from data obtained from Environment Canada's oil catalog (Whiticar et al., 1992).

Table 1 summarizes the total polynuclear aromatic hydrocarbon (TPAH) concentrations in each of the source oils (from Payne and Driskell, 1999). PAHs cause the most toxic impact to water column and benthic organisms of any portion of the oil, both because of

their fate (slightly soluble and semi-volatile, so as to enter and remain in the water column) and their toxicity to organisms. The mean total PAH content of the diesels was 5.5%. One of the bunker fuels (BFO 280) was 5.6% TPAH, while the other was only 0.54% TPAH. Thus, the percent composition in the spilled volume needed to be estimated. Since the 2 diesels and the BFO were all about the same in PAH content, the key was to determine what percentage of the spill was the low PAH MFO.

The Unified Command Decision Memo of March 12, 1999 (Appendix A) contains information on the volume in each fuel tank before the grounding. The memo describes which tanks were open to the sea before the burn, and estimates the amount from each tank which was burned. Based on this information, a mass balance was constructed of how much oil from each tank was released, burned, and remained on board.

Table 2 is based on the Unified Command Decision Memo (Appendix A). The Unified Command estimated the total spill volume as approximately 25,000 to 70,000 gallons, of which 5,000-10,000 gallons were released before the burn the night of 11 February. However, the SCAT surveys (see Preassessment Data Report, RPI, 1999) estimated that the amount of oil stranded on beaches was 11,413 gallons on 11 February and 20,698 gallons on 12 February. Thus, assuming the estimated volume of oil on the beaches does not include significant water or sediment, the amount of oil estimated to be on the beaches is greater than the amount estimated to have been released at that point in time. The Unified Command stated that the spill estimates were only approximate, as conditions did not allow for accurate measurement. Similarly, the SCAT survey estimates are only approximate.

Given the uncertainties in estimating spill quantities, this preliminary modeling effort was conducted using two spill scenarios, one assuming that 70,000 gallons of oil were released, and another assuming that twice as much, or 140,000 gallons were released. The model was run with the 140,000 gallon release assumption to evaluate the sensitivity of the model outputs to the spill quantity, in effect to evaluate a “worst case scenario” for the potential impacts from the spill assuming the Unified Command had significantly underestimated the spill volume.

Table 2 was constructed assuming the total spill volume was 70,000 gallons. The Unified Command estimated the volumes initially in each tank and burned. The unknowns were filled in using all the information available, including the measured density of oil in each tank before the grounding. For example, the density of the fuel in tank 4 indicates it was 55% MFO and 45% BFO. The majority of the spilled volume appears to be from tank 4. Tank 2 was also a mixture, but little appears to have been spilled from it. Most of the remaining spill volume was MFO from tanks 1 and 5, and the diesel.

The volume-weighted PAH content of the release was about 2.8% TPAH. As most of the release before the burn was apparently MFO, the TPAH content of that phase was 0.54%. After the burn, 86% of the release occurred, with a TPAH content of 3.1%.

Preliminary modeling results showed that the first phase of the release likely produced little impact on water column or benthic organisms. It was the second phase that produced most of the impact. Thus, for the model results herein, the oil was assumed to contain 3.1% PAH.

Toxicity of the oil was estimated using QSAR methods and the toxicity algorithm in the type A model (French et al. 1996a). The QSAR method was developed by French (1998a,b) as part of the analysis for the North Cape oil spill. In the North Cape analysis, the resulting 96-hour LC50 for total PAHs (with log (kow)<5.5) was 51 ppb at 25°C for species of average sensitivity. (Note the model input is the toxicity parameter normalized to 25°C. As the model runs, it corrects this input to the temperature at the spill site.) Equivalent LC50s for sensitive and insensitive species ranged from 16-359 ppb at 25°C. The model uses input data at a standard temperature of 25°C, and corrects to ambient temperature as it runs. At the temperature of the *New Carissa* event (9°C) the corresponding LC50s are 99 ppb (most sensitive species, such as amphipods), 317 ppb (average), and 2229 ppb (insensitive species) (Table 3, Figure 1). Table 4 shows how LC50 varies with time of exposure.

Table 5 contains all of the *water* sample TPAH data that could be identified as oil from the *New Carissa* spill (Payne and Driskell, 1999). These TPAH samples were taken between 11 and 16 February.

Water samples from 12 February taken along the beach near the grounded vessel (Payne and Driskell, 1999) were taken to be representative of the PAH content of oil in the water column. Appendix I contains the individual dissolved PAH concentrations for the 7 samples on that date (Table I-1). Dissolved concentrations of 2 to 4-ring PAHs with log (kow) < 5.5 are available to aquatic organisms and, thus, are toxic (see French 1998b). On average, 94% of the total PAH was 2 to 4-ring with log (kow) < 5.5 (Table 6). Tables I-2 to I-8 contain calculations of the percent composition of these PAHs for the 7 *New Carissa* dissolved PAH samples. The resulting LC50 of the PAH mixture is calculated using the QSAR model for PAH toxicity developed in French (1998b). The LC50 of the mixture is a weighed toxicity value based on the LC50 of each individual PAH and its percent composition in the mixture.

Table 6 summarizes the estimated LC50s at 25°C for the 7 *New Carissa* dissolved PAH samples. The range is 46-107 ppb for species of average sensitivity, and 14-46 ppb for sensitive species (e.g., amphipods). The LC50 decreases as the dissolved concentration of PAH decreases (Figure 2). This is because the samples with higher PAH concentrations contain relatively more of the low kow compounds, which are less toxic. With time, those low kow compounds will be preferentially removed by volatilization, leaving a more toxic mixture dissolved in the water.

Only a few dissolved PAH samples were over 10 ppb dissolved PAH (Table 5). Most samples were on the order of 1 ppb. Thus, the average LC50 for the dissolved PAH mixture after the spill was likely near the low end of the calculated LC50 range, i.e., in agreement with the LC50s estimated by French (1998b) for the North Cape spill. Thus,

the average 96-hour LC50 of 51 ppb at 25°C was used as input to the model for the studies herein.

3.4 Scenario

The Preassessment Data Report (RPI, 1999) contains a detailed chronology of the events. The spill site was at 124° 18.71'W, 43° 23.92'N. In brief, the release occurred beginning at 7:00 hrs on 8 February 1999. The release rate was slow from 8-12 February. At 17:42 hours on 11 February the vessel's tanks and holds were detonated with charges and the fuel was ignited. After the burn, a much larger release occurred as several tanks were open to the sea but still contained fuel. Most of the second larger phase of the release occurred from the afternoon of 12 February until 15 February. Very little oil was released from 16 February and after.

Given the chronology, and the ability of the model to simulate two sequential releases, the following modeling assumptions were made. The first phase of the release was assumed to be of a constant rate from 07:00 on 8 February until 13:00 on 12 February. The second phase of the release was also a constant rate from 13:00 on 12 February until 01:00 on 15 February. The end time was chosen as midway between the last observation of a large release rate on the afternoon of the15th and the morning of the 16th when the release rate was observed to be very small. Thus, the first phase occurred over 102 hours, and the second phase occurred from 102 hours to 162 hours after the spill start.

The spill volume of 70,000 gallons in Table 2 is based on the Unified Command Decision Memo (Appendix A, and see discussion above). Again, it should be noted that the Unified Command estimate was stated to be approximate, as it was not possible to directly measure the release volume, or calculate it by difference from fuel remaining in the ship. Thus, the 70,000-gallon spill estimate (10,000 gal. before the burn) may be low, and the estimated amount burned or remaining on board too high.

In model runs, the spill volume is assumed 70,000 gallons. An alternate volume of 140,000 gallons was also run to show sensitivity to this assumption. Other inputs are listed in Appendix D.

3.5 Biological Data

The biological data for the Oregon coast, province 47, in the NRDAM/CME was used as default data. Appendix F shows the biological data assumed.

The model separately quantifies injuries for animals greater than one year old (part of which is the catchable stock) and for young-of-the-year. The database contains abundance values for the catchable stock only. The younger age classes (>1 year old) are calculated from the stock abundance (using fisheries population modeling and mortality rates). The stock abundance per unit area is in the biological database (Appendix F).

The young-of-the-year abundance is the number of animals that would survive to age 365 days. This number is back calculated from the spawning stock, as that number needed to maintain the population at the current level. The total number of young-of-the-year are distributed spatially within habitats they occupy each month of the year (based on life history information) to calculate a number per unit area of habitat. That number per unit area of habitat is in the biological database (Appendix F).

The database for the Oregon coast in winter includes several species of interest. In the database, Pacific herring (both adult and young-of-the-year) are in nearshore and structured (i.e., seagrass, kelp, reef) habitats. Lingcod (both adult and young-of-the-year) are in offshore habitats. (Note that lingcod and greenlings are treated as a single category, with 99% of the standing biomass being lingcod.) Oysters are assumed only present in reefs. The map of reefs in Appendix C shows they are in Coos Bay at locations removed from the entrance.

4. RESULTS AND DISCUSSION

The differences between inputs to model runs are in Table 7. Other model inputs are listed in Appendix D. (Earlier model runs were made during the response phase to help guide field data collection. However, as they were based on incomplete data they are not reported here.)

Field observations after the spill are described in the Preassessment Data Report (RPI, 1999). Briefly, before the burn, the surface oiling was primarily near the grounded vessel. Eventually, most of the surface oil was transported to the north by southerly winds. While the dominant longshore current appeared to be to the north during much of the spill, some oil was carried southward near shore (presumably by wave-driven currents) and into Coos Bay (on incoming tides) during 10-11 February.

Based on these observations, runs were made both with tidal currents alone (Caris12) and with the addition of southward drift near the shore to transport the oil to the mouth of Coos Bay (Caris13-Caris17). The trajectories and modeled concentrations of dissolved PAHs are summarized by figures in Appendix G. Only Caris12 and Caris16 are shown in the Appendix, as the output is voluminous. Additional figures and model output can be viewed at a later time if needed.

The trajectory maps only show the pathway of the oil. The dots on the map are color-coded as to the type (total oil or just dissolved PAHs) and location (on the surface or subsurface as entrained or dissolved). The dots do not infer any mass or volume. They just represent some non-zero mass, however small. Since the dots do not infer any mass or volume, the trajectory is exactly the same whether the total spill volume is 70,000 gallons or 140,000 gallons. The trajectories are instructive in describing where the oil went.

The trajectories of Caris13 to Caris17 are all very similar, as shown by Caris16. The trajectory (Appendix G) shows that the surface oil moved faster and farther north, while the dissolved PAHs remained where they were entrained and dissolved (near and just north of the spill site) for a longer time. Eventually the PAH plume was swept northward and dispersed. This same pattern was observed by Payne and Driskell (1999).

Caris12 includes no southward drift near the spill site, including only tidal currents. The simulation shows the oil moving too fast to the north. Thus, the inclusion of tidal currents alone is not sufficient to simulate the *location* of the oil. Caris13-Caris17 (as approximated by Caris16) more correctly predicts the oil location.

The dissolved PAH concentrations estimated by run Caris16 (assuming 140,000 gal. spilled) most closely match the PAH concentrations measured in the field after the spill (Appendix G and Table 5, Payne and Driskell, 1999). Thus, the model suggests that a higher volume than 70,000 gallons may have been spilled. However, the focus of this modeling effort is not to estimate the spill quantity. There are substantial uncertainties associated with some of the model inputs, just as there were uncertainties in the derivation of the Unified Command spill volume estimates. At best, given the conditions at the time of the spill that severely hampered efforts to obtain accurate measurements from the ship and from the field, the actual spill quantity cannot be precisely determined. To provide a cautious evaluation of the impacts of the spill, it is appropriate to conduct this modeling analysis assuming the spill quantity estimated by the Unified Command and a scenario in which the amount of oil spilled was twice as great.

Table 8 and Appendix H summarize the biological injuries calculated by the model. The fishery (fish and shellfish) injuries estimated by the model were very low in all cases (Table 8, Appendix H). While the runs were apparently quite variable in result, none of the injuries quantified are large relative to the natural variation of abundance in the field, so the differences are not significant. Based on these modeling results, it is anticipated that inclusion of more detail and accuracy in the current data will not produce water column injury estimates much different than these results.

The best injury estimate of the model cases run is that for Caris16, because this case shows the closest agreement to the magnitude of PAH concentrations measured in field samples. The results indicate that the acute water column injuries resulting from the spill are low, amounting to about 1300 kg of fish and shellfish injuries.

These fish and shellfish kills are from acute exposure to dissolved PAH concentrations. The LC50 at ambient temperature ($9^{o}C$) is that concentration (317 ppb = 317,000 ng/l) which would kill 50% of those exposed for 96 hours. The concentration assumed to kill 1% of those exposed for 96 hours is 1% of the LC50, or 3.2 ppb. The LC50 increases for shorter times of exposure. Thus, modeled concentrations less than 3 ppb resulted in no acute injuries.

The acute water column injuries estimated by the model were all for organisms on the outer coast, and not for Coos Bay. However, the Coos Bay water samples showing 5-8 ppb dissolved PAHs (Table 5) may have resulted in some minor impact not predicted by the model. The oyster beds, which are well up inside Coos Bay, were not likely to have been impacted by acutely toxic PAH concentrations.

The biological model does not estimate sublethal or chronic (long-term exposure) effects on fish or shellfish. However, the physical fates model output of PAH concentrations may be evaluated to determine the potential for chronic effects.

One estimate of the threshold for chronic effects, which has been used in risk assessments, is an aqueous concentration equivalent to 1% of the LC50. This would be about 3 ppb in the *New Carissa* case. Literature reports on chronic concentration thresholds for PAH sublethal effects are few. However, some are available, as described below.

Recently, several experimental observations of lowest observable effects concentration (LOEC) for fish eggs and larvae exposed to oil PAHs have been published (Marty et al., 1999; Bue et al, 1998; Carls et al., 1999; Heintz et al. 1999; White et al., 1999; Roy et al., 1999). In these studies, aqueous PAH concentrations as low as 0.4 ppb TPAH at 5-6°C were shown to have effects. However, it appeared that this LOEC was resulting from larger PAHs in highly weathered oil. For less weathered oil, the LOECs were on the order of 9 ppb TPAH (Carls et al., 1999). The other studies reported consistent results (Marty et al., 1999; Bue et al, 1998; Heintz et al. 1999; White et al., 1999). Thus, it appears that fresh oil PAH concentrations of about 10 ppb, for a week or more of exposure, would cause chronic impacts to fish.

Tissue concentrations of PAHs that have been shown to have effects are summarized as follows. Donkin et al. (1989, 1991) estimated 50% effects levels in mussels for non-alkylated PAH tissue concentrations of about 14-31 ppm, corresponding to 80-920 ppb in the water. The highest 50% effects levels were for the lowest kow compounds, with the highest for naphthalene. Note that the LOECs, which were not reported, would be lower than these values. Gobas (1989) estimates that, in general for organics in aquatic fish and invertebrates, the chronic threshold tissue concentration for effects is 0.1-1 mmole/kg. For PAHs with molecular weight of about 200, this is 20-200 ppm. This agrees with the critical body residue (CBR, also called lethal body burden, LBR) model and associated data (McCarty), which show CBR=2-8 mmole/kg for chemicals (such as PAHs) that have a narcotic mode of action. Chronic values for narcotics are about 10-100 times lower than acute values (Giesey and Graney, 1989).

In a recent book chapter by Hellou (1996), an estimate of the threshold PAH tissue concentration for effects in bivalves is provided as on the order of 540 ppm to 8620 ppm. However, the estimates are based on rough approximations and assumptions, and the result is not consistent with the data described above. First, the author uses a sediment threshold for effects of 0.1-1ppm PAH. However, it is unclear what individual PAHs are included in that 0.1-1.0ppm, as some may be biologically unavailable or nontoxic (which

the author points out). Also, this threshold is for bulk PAH, but some is adsorbed to sediment, and some is dissolved in the pore water. The author estimates a water concentration, which may be intended as this pore water fraction, or may be intended to be overlying water. It is not clear which, but a fraction in water is assumed. Next a bioconcentration factor (BCF) is assumed to estimate a tissue concentration for effects. All of these steps are very approximate. The main problem is the PAHs represent a mixture with kows, BCFs, and toxicities that range over orders of magnitude. Thus, this kind of bulk calculation on total PAH is not appropriate or accurate.

It is better to use directly-measured water or tissue concentrations where effects are seen. The range in the literature for fish and invertebrates is about 10-200 mg/kg in tissue (10-200 ppm). Assuming a BCF on the order of 2000 (roughly typical for PAHs), the corresponding water concentration would be order of 0.5-100 ppb PAHs. Water concentrations of TPAH shown to have effects on fish embryos were observed between 0.4 and 10 ppb. Thus, the threshold water concentration for effects after the *New Carissa* spill would be on the order of 1-10 ppb for sensitive species and stages, and 10-100 ppb for less sensitive organisms (such as mussels).

These levels, combined with the field data on PAH concentration in water and tissues after the *New Carissa* spill, indicate that some sublethal effects might have occurred near the spill site and in parts of Coos Bay. However, the data do not suggest any major long-lasting impacts.

The tables in Appendix H include estimates of interim loss resulting from the direct kills from acute exposure, measured as production foregone. Production foregone is defined as the biomass that would have been produced by animals killed by the spill if they had not been killed and had been subject to the same growth and mortality as animals which had not been killed. It is calculated from numbers in each age class killed, as follows. The number of an age class which would have lived the following year (percent survival times number killed) is multiplied times average weight gain from the killed 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. The losses for each killed age class are then summed for all age classes to calculate total production foregone. In addition, discounting of future losses is applied at a 3% annual discount rate. Thus, next year's losses are 97% of present-year value, the following year's value is 94% of present value, etc.

The equations to calculate lifetime production foregone are:

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

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

where $N_{i,y}$ is the number of killed animals of age class i that would have remained 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 beginning at the age of recruitment. Annual survival is calculated as $e^{-(M+F)}$, where M = instantaneous annual natural mortality rate and F = instantaneous annual fishing mortality rate. Values for M, F, and the age of recruitment to the fishery are from French et al. (1996c). 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 Bertalanffy equation (a variation of the Brody growth equation, Ricker, 1975) is used:

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

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,

$$W_t = a L_t^b$$

where W_t is weight (g, wet) at age t years and a and b are constants. Needed length-weight by age parameters are in French et al. (1996c).

The sum of the direct kill (kg) and the interim loss measured as production foregone (kg) gives the total injury caused by the spill.

5. CONCLUSIONS

Based on these modeling results, it is anticipated that inclusion of more detail and accuracy in the current data will not produce water column injury estimates much different than these results. The best injury estimate of the model cases run is that for Caris16, because this case shows the closest agreement to the magnitude of PAH concentrations measured in field samples. The results indicate that the acute water column injuries resulting from the spill are relatively low, amounting to about 1300 kg of fish and shellfish injuries.

Based on these model results, most of the injury, in terms of biomass as kg wet weight) was for rockfish, Pacific whiting, squid and anchovies (Table H-4, Appendix H, repeated as Table 9). The Pacific herring and lingcod (99% of greenlings category) injury (which includes older classes and young-of-the-year) was small. The impacts to crabs also appear to have been relatively small. In consulting with local resource managers, concern was raised that the default data available in the CERCLA Type A model may underestimate the pre-spill water column abundance of these organisms, and by

extension, the acute injury from the spill. However, local season-specific data are not available to determine if the default data are in fact low. Also, even if the actual pre-spill abundance of crabs were substantially higher, the model would still generate relatively small impacts for these resources.

The modeling results also indicate that concentrations of PAHs that could be lethal to oysters did not occur in Coos Bay in the areas where commercial oyster beds are located. It should be noted, however, that some spatial heterogeneity occurs in the distribution of PAHs that isn't accounted for in the model. The currents used in the modeling were smoothed averages, and did not include small current features and eddies.

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Table 7. Cases run for both physical fates and biological effects.

Case	Volume (gallons)	Horizontal Dispersion (m^2/sec)	Currents
Caris12	70,000	3	Coosbay1.dir, no background current
Caris13	70,000	10	Coosbay1.dir
Caris14	140,000	10	Coosbay1.dir
Caris15	70,000	1	Coosbay1.dir
Caris16	140,000	1	Coosbay1.dir
Caris17	70,000	1	Coosbay2.dir