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Estimation of potential impacts and natural resource damages of oil

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Abstract

Methods were developed to estimate the potential impacts and natural resource damages resulting from oil spills using probabilistic modeling techniques. The oil fates model uses wind data, current data, and transport and weathering algorithms to calculate mass balance of fuel components in various environmental compartments (water surface, shoreline, water column, atmosphere, sediments, etc.), oil pathway over time (trajectory), surface distribution, shoreline oiling, and concentrations of the fuel components in water and sediments. Exposure of aquatic habitats and organisms to whole oil and toxic components is estimated in the biological model, followed by estimation of resulting acute mortality and ecological losses. Natural resource damages are based on estimated costs to restore equivalent resources and/or ecological services, using Habitat Equivalency Analysis (HEA) and Resource Equivalency Analysis (REA) methods.

Oil spill modeling was performed for two spill sites in central San Francisco Bay, three spill sizes (20th, 50th, and 95th percentile volumes from tankers and larger freight vessels, based on an analysis of likely spill volumes given a spill has occurred) and four oil types (gasoline, diesel, heavy fuel oil, and crude oil). The scenarios were run in stochastic mode to determine the frequency distribution, mean and standard deviation of fates, impacts, and damages. This work is significant as it demonstrates a statistically quantifiable method for estimating potential impacts and financial consequences that may be used in ecological risk assessment and cost-benefit analyses. The statistically-defined spill volumes and consequences provide an objective measure of the magnitude, range and variability of impacts to wildlife, aquatic organisms and shorelines for potential spills of four oil/fuel types, each having distinct environmental fates and effects.

Keywords: Oil spill model; Natural resource damage; Fates and effects; Ecological risk; Spill costs

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1. Introduction

In order to determine risks of impact to resources and potential natural resource damages, multiple scenarios and conditions need to be evaluated to develop an expectation of risk of oil impacting each resource of concern. The most influential variables determining oil fates are spill location, oil type, spill size, and environmental conditions after the release. In this study, a Monte Carlo simulation approach was used for two potential spill locations and each of 12 spill size and oil type combinations, to characterize the bio-economic consequences of spills.

The United States Army Corps of Engineers San Francisco District (ACOE) is evaluating the oil spill risks associated with the four submerged rock pinnacles (Harding, Shag, Arch and Blossom Rocks) located in central San Francisco Bay east of the Golden Gate and in or near the shipping traffic lanes (Fig. 1). The concern is the potential for a loaded oil tanker or freighter grounding on these rocks and causing an oil spill. The purpose of this study was to evaluate the ecological and financial consequences of such spills using bio-economic oil spill modeling. The present paper summarizes the results of this work, focusing on biological impacts and natural resource damage (NRD) costs. Details of the model assumptions and results may be found in French McCay et al. [1]. Estimated socioeconomic and response costs are quantified in Etkin [2,3].

The 12 spill scenarios analyzed were for a matrix of four oil types (gasoline, diesel, crude oil, and heavy fuel oil) and three spill sizes (small, medium and large). Four fuel types were selected as representative of fuels shipped through San Francisco Bay: Alaska North Slope crude oil (AK crude), heavy fuel oil (HFO), diesel and gasoline. In order to define the potential spill volumes, a probability density function

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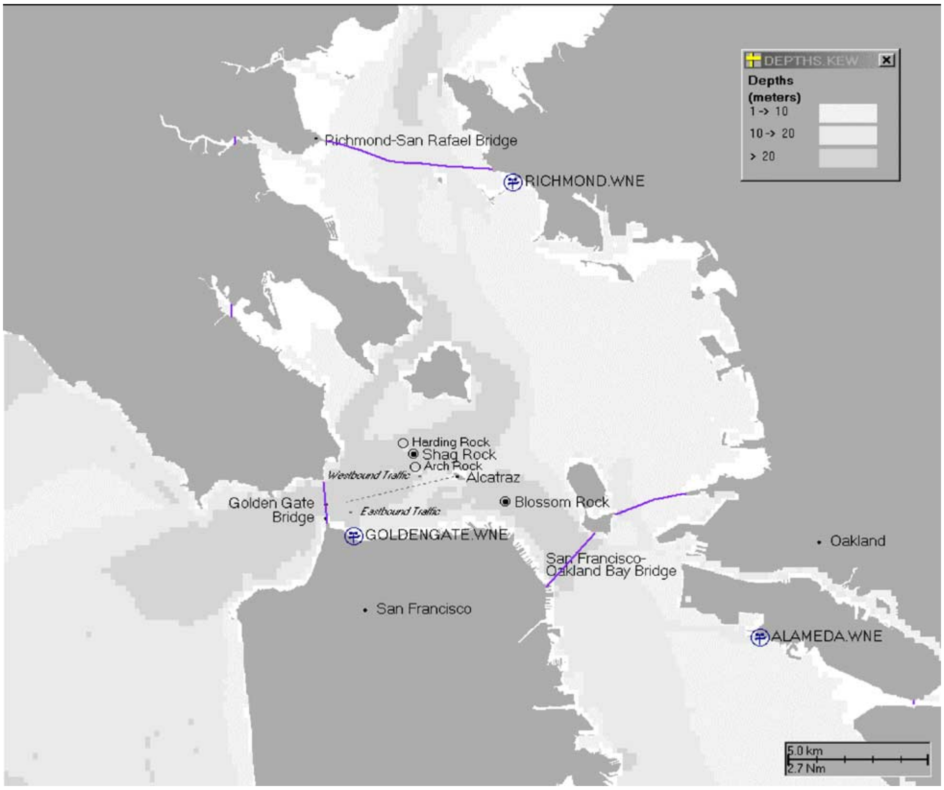

Fig. 1. Central San Francisco Bay, showing the four rock pinnacles (Harding, Shag, Arch and Blossom Rocks) and three wind records used in the modeling (Golden Gate, Richmond and Alameda PORTS stations).

(PDF) for oil spill size (probabilities of spills greater than each size over a range of potential sizes) was created by Etkin and Michel [4] for each oil type, based on relevant historical oil spill events, shipping traffic in San Francisco Bay, and analysis of various spillage volumes. The medium spill was the mean spill size, the small spill was the 20th percentile spill, and the large spill was defined as the 95th percentile spill for the relevant vessel corresponding to the oil type (Table 1). These percentiles represent the probability distribution of spill size *given that a spill occurred*.

For each spill site (i.e. at Shag Rock, representing spills at Harding Shag or Arch Rock, and at Blossom Rock, Fig. 1) and each of the 12 oil type-spill size scenarios, the model was run numerous times (100 was found adequate based on tests with up to 200 runs, i.e. probability of oil reaching various locations varied less than 5% if greater than 100 runs were made), with each run using a randomly varied spill date, such that environmental conditions were varied within the possible range of conditions (i.e. tidal current patterns, river flow conditions and wind data). The results were rank ordered such that the 50th (median) and other percentile spill dates-times could be identified. For each of the twelve scenarios (i.e. 20th, 50th and 95th percentile volumes for four oil types) at each of two spill sites, the 50th and 95th percentile runs, in terms of impacts, were examined in detail for ecological impacts and financial consequences as NRD, socioeconomic, and response costs.

2. Model description

The SIMAP (Spill Impact Model Application Package) model system developed by Applied Science Associates (ASA) was used for this study. This model, comprised of three-dimensional oil fates and biological effects models, originated from the Natural Resource Damage Assessment

Table 1
Oil types and spill volumes

Oil type	20th Percentile	50th Percentile	95th Percentile
Gasoline (Product Tanker)	151 MT (50,000 gal)	818 MT (270,000 gal)	3785 MT (1,250,000 gal)
Diesel (Product Tanker)	162 MT (50,000 gal)	875 MT (270,000 gal)	4049 MT (1,250,000 gal)
AK Crude (Crude Tanker)	332 MT (100,000 gal)	1990 MT (600,000 gal)	9949 MT (3 million gal)
Heavy fuel oil (Freighter)	92.3 MT (25,000 gal)	369 MT (100,000 gal)	1513 MT (410,000 gal)

Model for Coastal and Marine Environments (NRDAM/CME) that ASA developed for the US Department of the Interior for use in Natural Resource Damage Assessment (NRDA) regulations [5–7]. While the NRDAM/CME is focused on natural resource damage assessment for specific hindcasts, SIMAP may be run in stochastic mode to evaluate a probability distribution of results, rather than just a single result for a specific hindcast. Below is a summary of the conceptual design of the model. Details may be found in technical reports and papers as indicated below.

2.1. Physical fates

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. Processes simulated include slick spreading, evaporation of volatiles from surface oil, transport on the surface and in the water column, randomized dispersion, emulsification, entertainment of oil as droplets into the water, dissolution of soluble components, volatilization from the water column, partitioning, sedimentation, stranding on shorelines, and degradation. Oil mass is tracked separately for lower molecular weight aromatics (1 to 3-ring aromatics), which are soluble and cause toxicity to aquatic organisms [8], 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 [5,9,10]. The algorithms and assumptions of the 3-d fates model are described in French et al. [9] and French McCay [10].

In the SIMAP fates model, crude oils and petroleum products are represented by seven components, six of which (all but the residual) evaporate in the model:

1. Monoaromatic Hydrocarbons (MAHs), including BTEX (benzene, toluene, ethylbenzene and xylenes) and alkyl-substituted benzenes.
2. Two-ring Polynuclear Aromatic Hydrocarbons (PAHs), i.e. naphthalenes.
3. Three-ring PAHs
4. Volatile aliphatics (boiling point $< 180\,^{\circ}$C).
5. Semi-volatile aliphatics (boiling points 180–265 °C).
6. Low volatility aliphatics (boiling points 265–380 °C).
7. Residual fraction (both aromatics and aliphatics, boiling point $> 380\,^{\circ}$C).

The SIMAP fates model quantifies, in space and over time, for each individual model run:

- Oil mass, volume and thickness on the water surface.
- Oil mass, volume and thickness on shorelines.
- Subsurface total hydrocarbon concentrations.
- Dissolved aromatic concentrations.
- Total hydrocarbons and aromatics in the sediments.

2.2. Biological effects

The biological effects model [5,10] estimates the area, volume or portion of a population affected by surface oil, concentrations of oil components in the water, or sediment contamination. The model calculates the extent and duration of exposure based on the outputs of the oil fates model. A rectangular grid of habitats represents the area potentially affected by the spill, with each grid cell coded for habitat type. Habitats include various offshore, nearshore, reef, wetland and shoreline environments that have unique assemblages of species. A contiguous grouping of habitat grid cells with the same habitat code represents an ecosystem in the biological effects model. Fish, invertebrates, birds, mammals and production rates of organisms lower in the food chain are assumed constant and evenly distributed across an ecosystem within the time period of the simulation. Fish, birds and mammals are assumed to move at random within each ecosystem. Planktonic stages (eggs and larvae in the water column) are moved with the currents.

In the biological effects model, surface slicks impact wildlife (birds and mammals). A portion of wildlife in the area swept by slicks over a threshold thickness ($10\,\mu$m, based on data and calculations in French et al. [5]) are assumed to die based on probability of encounter with the slick and mortality once oiled. Area swept is calculated for the habitats occupied by the species of concern. Estimates for the mortality probabilities are derived from information on behavior and field observations of mortality under similar circumstances [5]. Wildlife mortality is directly proportional to area swept, probability of mortality, and species abundance per unit area.

Fish and their eggs and larvae are affected by dissolved aromatic concentration in the water or sediment. Because exposures in the water column are short (hours to days), 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 (i.e. the relationship of percent mortality to log(concentration) is the bell-shaped Gaussian distribution). LC50s for the mixture of the most toxic components of oil, dissolved MAHs and PAHs, are used to define the center of that log-normal function. The effects of the MAHs and PAHs are additive, and LC50s for the oil mixture are estimated using an additive (toxic unit) formula. LC50s for the 50th percentile of species tested [8] are used in this study to provide a mean expected impact.

For plankton, fish and invertebrates, movements of biota, either active or by current transport, are accounted for in determining time and concentration of exposure. Tracers representing schools or groups of animals move or remain stationary in the model according to the behavior of the animal type, and concentration and duration of exposure are recorded. Exposures are integrated over space and time by habitat type to calculate a total percentage killed [5,10].

The biological effects model has been validated using simulations of about 30 spill events where data are available for comparison [11,10]. In most cases [11] only the wildlife impacts could be verified because of limitations of the available observational data. However, in the *North Cape* spill simulations, both wildlife and water column impacts (to lobsters) could be verified [10].

To calculate injury for NRDA, 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 1-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 [5].

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 (termed “production foregone”) is estimated using the fisheries population model and added to the direct kill injury. The total injury 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 [12,13].

2.3. Natural resource damages

Natural resource damages are based on cost of restoration, in accordance with the Oil Pollution Act of 1990 (OPA) NRDA regulations and current practice by government trustees in the US. Two restoration approaches were modeled, which are the most likely alternatives to be used in the event of spills in California:

- Habitat restoration, scaled with Habitat Equivalency Analysis (HEA).
- Restocking/rehabilitation of seabird species, scaled with Resource Equivalency Analysis (REA).

2.3.1. Habitat restoration HEA costs

The methods for the habitat restoration model were developed for the *North Cape* oil spill NRD and restoration plan [12,13]. The concept is that the restored habitat leads to a net gain in wildlife, fish and invertebrate production. The size of the habitat (acreage) restored is scaled to just compensate for the injury (interim loss, expressed as production lost) when both are expressed in values indexed to the same year (i.e. the year of the spill). In San Francisco Bay, it is likely that saltmarsh restoration would be undertaken as restoration for wildlife, fish and invertebrate injuries. California has lost most of its native saltmarsh lands, and extensive efforts are underway to restore marshes all over the state. Thus, the NRDA costs were based on saltmarsh creation and HEA.

The injuries are scaled to the new primary (plant) production produced by the created habitat, as the entire food web benefits from this production. The amount of compensatory plant production that needs to be restored is calculated using a simple food web model, where injuries to upper trophic level organisms are translated to equivalent plant production using trophic transfer efficiencies from the literature [5]. The compensatory production is translated to area of habitat required by dividing by annual rate of production per unit area by a factor (Λ) that accounts for the expected project life of the restored wetland and discounting of future gains (based on the economic model of paying interest):

$$\Lambda = \sum_{y=1}^{y=\lambda} \left(\frac{1}{(1+d)} \right)^{y}$$

where $d = 0.03$ (an accepted rate for NRDAs [12,13]), λ is the project life, and y is years after restoration begins. The value of Λ approaches 31.6 if the summation is for a project life of 100 or more years (i.e. of assumed habitat existence after restoration). The scaling of habitat required is sensitive to the assumed project life. If the lifetime is 10, 20 or 50 years, the value of Λ is 8.5, 14.9, and 25.7, respectively. In this study, a conservatively low habitat requirement, using the factor $\Lambda = 31.6$, was assumed.

The calculation above assumes the habitat is fully functional from the start of the restoration project. If new habitat is created, as assumed in this study, there will actually be a period of “recovery” while the habitat develops to full function. The recovery curve is assumed sigmoid in shape, taking 15 years for development to 99% of full function [5]. Annual (full-function) primary production rates (280 g dry weight/m^2 per year) for San Francisco saltmarshes (*Spartina foliosa*) were obtained from [14]. Correction is included in the model for the further lag in the production of compensatory biomass during development of the marsh. Also, the compensatory acreage is increased 3% annually for each year in delay before the restoration project begins [13]. The cost of creating saltmarsh, translated to 2001 dollars, is US$ 47.58 m^{-2} [5].

2.3.2. Restocking REA costs

Because the benefits of the HEA restored habitat are indirect to seabirds, more direct restoration approaches are usually used to replace them. Thus, alternate estimates of seabird restoration costs were developed for this study. Restoration costs for injuries to seabirds were determined from a log-linear regression relating cost per bird (US$/bird) to average abundance per unit area (#/km^2). The premise of theoretical model is that scarce birds are more valued by the public and are more expensive to restore, and common species are less valued and are cheaper to restore [15].Table 2 contains the data used to develop

Table 2
Data used to estimate cost (US$ 2001) per bird restored as a function of species abundance (as California-wide annual mean #/km^2, except as noted)

Species	Source of cost data	US$/bird restored	Annual mean #/km^2
Mallard	Conservation Reserve Enhancement Program	120	240.0
Common murre	Devil's Slide Rock Murre Recolonization Project	600	223
Common murre	*Command*	714	223
Common murre	*Apex Houston*	819	223
Pelican	*American Trader*	13,430	0.77
Bald eagle	French et al. (1996a)	8100	0.0112 (Virginia)
Peregrine falcon	Montrose settlement	9000	0.01 (east coast)

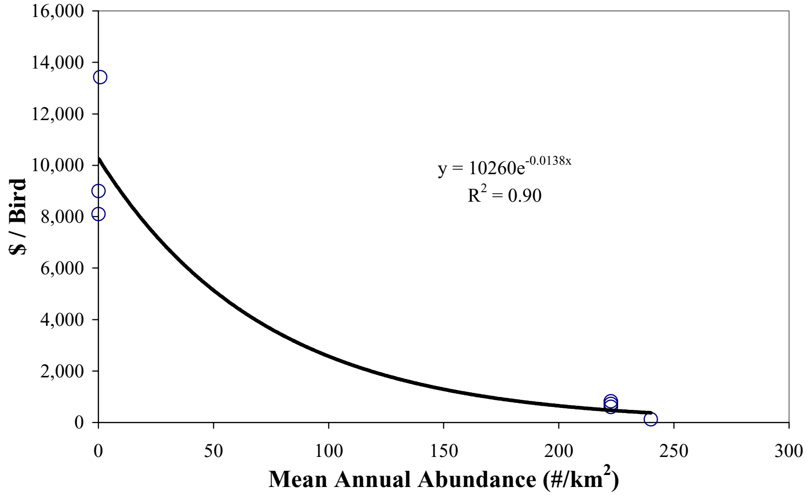

Fig. 2. Seabird restoration cost model, based on the data in Table 2.

the restoration cost model for seabirds. The mallard cost data (US$ 120/bird) were from a mallard project based on the Conservation Reserve Enhancement Program for Mallards in the Central Valley of California. The common murre cost data at US$ 600/bird was from the Devil's Slide Rock Murre Recolonization Project (S. Hampton, California Fish and Game, personal communication, 2001). Other cost estimates per bird were based on the amount allocated for bird restoration in recent settlements, including the *Apex Houston*, *American Trader* and *Command* oil spills ([16,17]; Steve Hampton, personal communication, 2002). The equation "fit" to these data (Fig. 2, obtained by regression of ln(cost/bird) versus annual mean abundance, #/km^2) is:

$$y = 10260\,e^{-0.0138x}$$

where x: annual mean abundance (#/km^2) and y: cost per bird (2001 US$). It is anticipated that more cost data for bird projects will become available in the future to develop a refined model. For the present analysis, this equation was used with annual mean abundance to estimate restoration costs for seabird species groups. Table 3 provides the annual mean number of bird/km^2 for each group of seabirds used in the REA scaling calculations and the estimated cost per bird by species. The abundance values are means for California waters [7].

Table 3
Annual average numbers per km^2 (California-wide) for groups of seabirds species used in the estimated cost/restored bird calculations and estimated cost per bird using the seabird restoration model

Species	#/km^2	Cost (US$/bird)
Albatroses	1.68	10,020
Small alcids	103	2,462
Cormorants	48.9	5,224
Guillemots	0.81	10,140
Gulls	171	968
Jaegers	1.74	10,020
Kittiwakes	2.63	9,890
Murres	230	428
Phalaropes	796	0.17
Shearwaters	2.83	9,870
Storm-petrels	10.5	8,880
Terns	3.53	9,770
Pelicans	0.77	10,150

3. Model input data

3.1. Geographical and environmental data

SIMAP uses a rectilinear grid to designate the location of the shoreline, the water depth (bathymetry), and the shore or habitat type. Digital shoreline data were gridded from Environmental Sensitivity Indices (ESI) coverages in the Environmental Sensitivity Atlas Geographical Information System (GIS) for the area obtained from NOAA HAZMAT in Seattle, Washington (on CD-ROM). ESI codes were translated to equivalent habitat codes for SIMAP. Vegetated subtidal habitats (seagrass and kelp beds) were mapped from coverages also provided in Environmental Sensitivity Atlas CD-ROM. Other subtidal areas were assumed to be sand (outside the bay) or silt–mud bottom (inside the bay). Depth data were obtained from Hydrographic Survey Data supplied on CD-ROM by the US Department of Commerce, National Oceanic and Atmospheric Administration, National Geophysical Data Center. The grid cell size was 0.12° of longitude and latitude (i.e. 176 m east–west by 222 m north–south).

Because of the large spatial variability of winds in and just outside of San Francisco Bay, multiple wind records of hourly wind speed and direction were used for the model runs: San Francisco NOAA buoy #46026 and San Francisco Bay Ports 9414750 (Alameda), 9414750 (Golden Gate), and 9414863 (Richmond); from a period of complete data, 11 February 1996 to 31 May 2001. While a longer wind record would be desirable, statistical analysis of the available longer-term (buoy) wind records showed year-to-year variability was relatively low, while spatially variability between stations was quite high. As the focus of the study was on the median and distribution of consequences, and not on extreme events, the shorter more spatially-complete wind record was judged more appropriate and adequate. The wind data were spatially interpolated between stations using a linear distance–weighed scheme for both speed and direction. While three wind stations would not be sufficient to hindcast a specific event, where localized wind patterns would be important, the general wind patterns the three stations provide yield reasonable distributions of expected oil impacts, measured as the area of surface water and shoreline oiled, as well as subsurface volume contaminated greater than a threshold concentration. These measures of impact are much less variable with wind direction than are trajectories.

Surface water temperature was varied by the month of the oil spill simulation, based on average monthly mean surface water temperature for central San Francisco Bay in French et al. [6]. The air immediately above the water was assumed to have the same temperature as the water surface, this being the best estimate of air temperature in contact with floating oil. Salinity near the spill site in San Francisco Bay was assumed 32 ppt [6].

ASA's boundary-fitted coordinate hydrodynamic model (BFHYDRO) was used to generate current data sets for the modeling. The boundary conforming system used is defined in general curvilinear coordinates to map the model grid to the shoreline of the water body being studied. It also allows enormous versatility in grid sizing so that many of the smaller features such as the rocks, rivers and embayments, may be resolved, along with the larger open water areas, without being penalized by an excessively small grid size (enormous number of cells). The model is fully described with test cases and a sample application in Muin and Spaulding [18,19].

The hydrodynamic model domain included San Francisco Bay beginning at the San Joaquin Delta, and including the coastal area from Monterey Bay to Point Reyes. We applied BFHYDRO in the two-dimensional (2-D), vertically averaged mode because the bay is highly energetic and predominantly well mixed vertically. The model was driven with freshwater inflow at the San Joaquin Delta (two conditions: dry season, low delta outflow, and wet season, high delta outflow) and tidal forcing at the open ocean boundary. The circulation in the bay is almost completely tidally driven and for the present analysis, the density driven (i.e. salinity induced) flows, were not considered. Wind-driven surface currents are calculated within the SIMAP fates model, based on local wind speed and direction using the algorithms of Youssef and Spaulding [20–22].

3.2. *Dissolved aromatic toxicity*

The PAH LC50 value for diesel, crude oil and heavy fuel oil for infinite exposure time was assumed to be 48 ppb [8]. All species were assumed to be of average sensitivity in this analysis, as most species are of near average sensitivity and sufficient data are not available to determine appropriate LC50s for each affected species within the range of possible values. Similarly, the LC50 for MAHs dominant in gasoline was the value for species of average sensitivity (50th percentile), 3.12 ppm [8].

The LC50s are for the concentration of *dissolved* MAHs and PAHs that would be lethal to 50% of exposed organisms for a long enough times of exposure for mortality to occur. For PAHs, this is for at least a week of exposure at warm temperature. For chemicals in general, toxicity is higher, and the LC50 lower, at longer time of exposure and higher temperature [8]. The duration of exposure is estimated in SIMAP and the LC50 is corrected accordingly, as well as for temperature.

3.3. *Biological abundances*

Biological data for fish and invertebrates in San Francisco Bay from the NRDAM/CME [7] were assumed the simulations. Bird abundance data were compiled in 1997 by ASA and Ecological Consulting (Portland OR, Glenn Ford, personnel communication) as part of an update to the NRDAM/CME for California Fish and Game (i.e. for NRDAM/CAL). Abundance varies monthly or seasonally, depending on available data. Separate data sets were developed and used for inside San Francisco Bay (Table 4) and in coastal waters just outside the bay (Table 5). Waterfowl include diving ducks, loons and grebes. Seabirds include common murres, cormorants, gulls, and terns.

Table 4
Total wildlife by group in San Francisco Bay (average #/km^2)

Group	Winter	Spring	Summer	Fall
Waterfowl	92	6	0	83
Seabirds	21	34	34	21
Wading birds	189	192	223	192
Shorebirds	2256	837	1901	3044
Kingfishers	0.2	0.2	0.2	0.2
Pinnipeds (seals)	1.4	0.2	0.2	1.4

Table 5
Total wildlife by group outside San Francisco Bay (average #/km^2)

Group	Winter	Spring	Summer	Fall
Waterfowl	2043	299	265	2103
Seabirds	39	59	207	96
Wading birds	188	190	221	191
Shorebirds	2310	892	1956	3099
Kingfishers	0.2	0.2	0.2	0.2
Pinnipeds (seals)	0.8	0.6	0.7	0.9

3.4. Fates model inputs

Two spill sites were modeled: Shag Rock (representing the three closely located rock pinnacles Harding, Shag, Arch) and Blossom Rock (Fig. 1). Table 6 summarizes the fates model input parameters for all scenarios. The removal of mass by cleanup and application of dispersants were not included in the model simulations. Oil is transported assuming no response, with the exception of deflection booms in designated protection areas according to the regional response plan. Oil reaching shore accumulates up to a holding capacity (varying by shore type and viscosity) and remains on shore, weathering at a natural rate [5,9].

4. Results

4.1. Physical fates

Exposures to each oil constituent (water surface, shoreline, dissolved aromatics in water) are analyzed over all runs to determine the median and 95th percentile conditions expected for the oil type and spill size scenario. Runs producing the 50th and 95th percentile result were identified for further impact analysis. The same model run is not the 50th or 95th percentile case for water surface, shoreline, and water column impacts. In fact, when shoreline impacts are highest, water column impacts tend to be relatively low, and visa versa. The impact measures from the stochastic modeling provide a quantitative method for determining which runs are 50th and 95th percentile cases for the resource of interest.

Birds and other wildlife are impacted in proportion to the water and shoreline surface area oiled above a threshold thickness for effects. Shoreline habitat impacts are proportional to surface area oiled above a threshold thickness for effects. Contamination in the water column changes rapidly in space and time, such that a dosage measure as the product of concentration and time is a more appropriate index of impacts than simply peak concentration. As toxicity to aquatic organisms increases with time of exposure, such that organisms may be unaffected by brief exposures to the same concentration that is lethal at long times of exposure. Toxicity data [8] indicate that the 96-h LC50 (which may serve as an acute lethal threshold) for dissolved aromatics (primarily PAHs) averages about 50 μg/l (ppb). Thus, this lethal exposure dosage threshold is 5000 ppb-h.

Recreational, tourism, boating/shipping, and other socioeconomic impacts are functionally related to the length of shore and area of water oiled. Duration of the impact on water may be captured by the sum of oil area and/or thickness (microns or g/m^2) times time oiled. Cleanup costs are related to volume spilled, water surface area, and area (or length) of shore oiled.

Impact indices were plotted as rank-order distributions:

- Water surface exposed to floating hydrocarbons, as the sum of area covered by more than 1 g/m^2 times duration of exposure (m^2-h).
- Shoreline area exposed to hydrocarbons of >100 g/m^2 (about 0.1 mm thick), which was the cleanup threshold

Table 6
Inputs to the fates model for all scenarios

Name	Description	Value(s)
Spill Site: Shag Rock	Latitude and longitude of the release	37°50.0604′N, 122°26.43480′W
Spill Site: Blossom Rock	Latitude and longitude of the release	37°49.1034′N, 122° 24.1956′W
Depth of release	Depth below the water surface of the release or 0 for surface release	11–12 m = bottom of ship deep enough to hit rock
Spill duration	Hours over which the release occurs	3 h
Model time step	Time step used for model calculations	0.1 h
Model duration	Length of each model simulation	7 days
Number of runs	Number of random start times to run in stochastic mode	100
Number of oil spillets	Number of Lagrangian elements used to simulate whole oil	10,000
Number of aromatic spillets	Number of Lagrangian elements used to simulate dissolved aromatics in the water	10,000
Horizontal turbulent diffusion coefficient	Randomized turbulent mixing parameter in x and y	1 m^2/s
Vertical turbulent diffusion coefficient	Randomized turbulent mixing parameter in z	0.0001 m^2/s

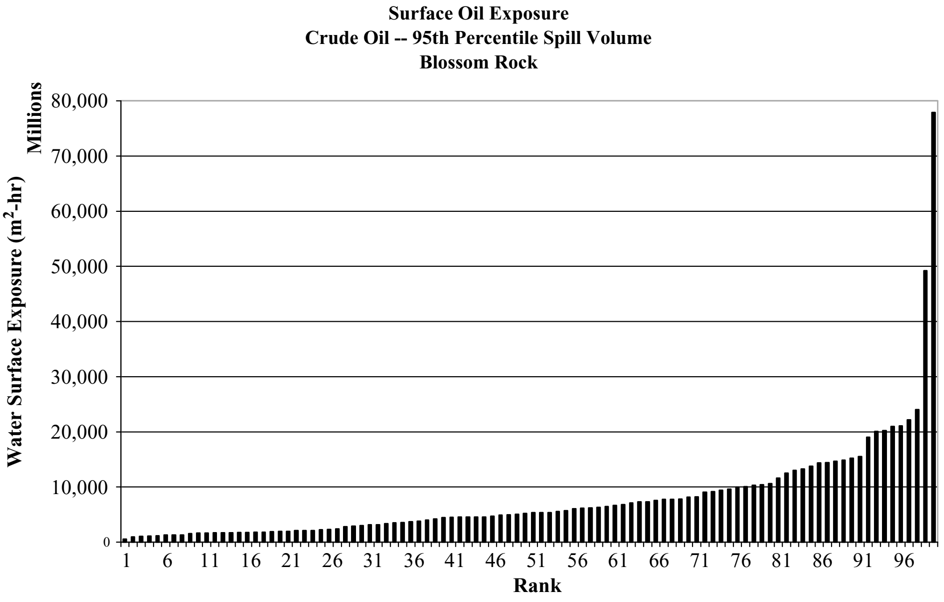

Fig. 3. Water surface exposed to floating oil.

assumed by Etkin [3] and is also the impact threshold assumed for oiling of birds. The thickness is the mean over a model grid cell, i.e. the cumulative mass of oil coming ashore within a cell, divided by the diagonal length of the cell (shore segment length) times the intertidal zone width for that shore type.

- Water volume exposed to >1 ppb of dissolved aromatic concentration at some time after the spill (threshold for effects based on [8]).
- Exposure dose of dissolved aromatics (ppb-h) in the water volume exposed to >1 ppb of dissolved aromatic concentration at some time after the spill.

Figs. 3–6 show the distribution of model results for all spill simulations for Blossom Rock within the crude oil 95th percentile volume scenario, indicating the range of possible impacts depending on the weather conditions and currents at the time of the spill. Similar figures were generated for the other 11 scenarios at Blossom Rock and 12 scenarios at Shag Rock. In most cases, there is a smooth frequency distribution about the median case. However, occasionally extreme events occur, i.e. the weather conditions are just right to cause the most impact. These figures indicate the median and distribution of impact indices, including the degree of variability and likelihood of extreme events.

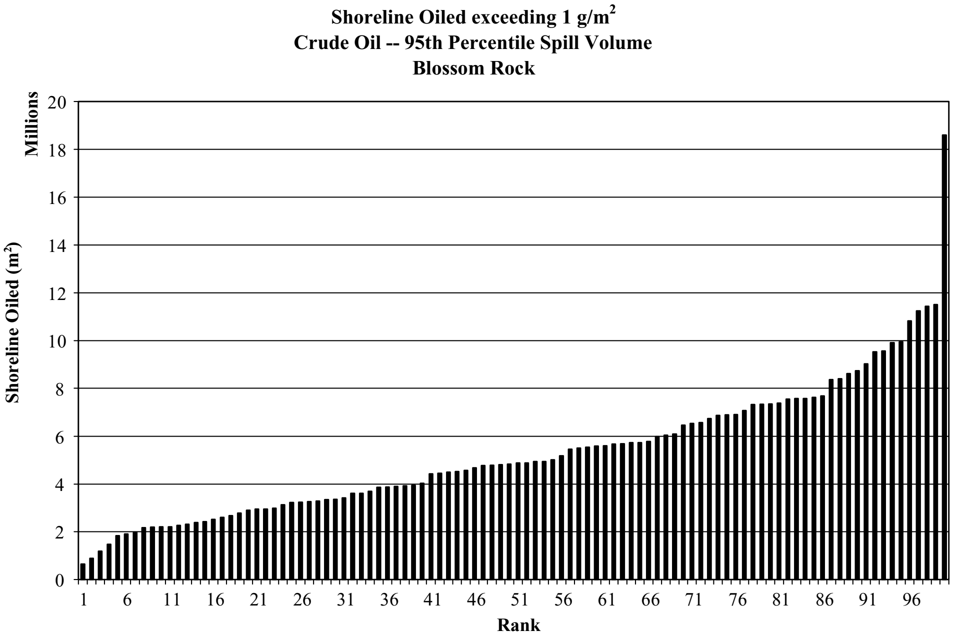

Fig. 4. Shoreline exposed to oil >100 g/m^2 (about 0.1 mm thick).

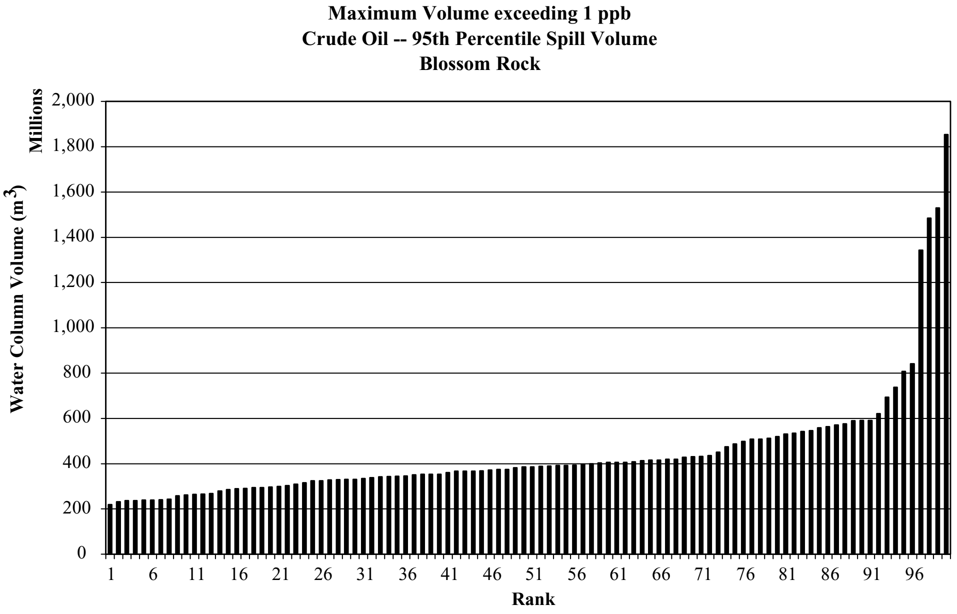

Fig. 5. Water volume exposed to >1 ppb of dissolved aromatic concentration at some time after the spill.

Table 7 contains the range of surface water exposure to >1 g/m^2 floating oil for spills for each type of fuel (both spill sites combined). The threshold of 1 g/m^2 (∼1 μm thick, equivalent to heavy sheen or more oil) is used here as an index. Exposures would be greater than the listed range only during environmentally extreme events with a frequency of occurrence <1%. The surface exposure of floating hydrocarbons for gasoline is relatively small and short-lived because gasoline is so volatile that as soon as it reaches the surface, it quickly evaporates. Therefore, the diesel and crude oil would have the most detrimental effects to the surface water based on exposure to floating hydrocarbons. This is reflected in the estimated impacts to wildlife and shorelines, response costs and socioeconomic impacts. The lower impact in the heavy fuel oil spills is because of the lower spill volumes, which are less than half the diesel volumes for each percentile volume. The crude oil spill volumes are about twice the diesel spill volumes, but diesel spreads faster and so covers more surface area per unit volume.

Table 7 summarizes the shoreline area exposed to hydrocarbons exceeding a threshold of 100 g/m^2 for each of the oils modeled (combining both spill sites). Diesel, crude oil and heavy fuel would be expected to oil the largest area of shoreline. Extreme events could cause exposure to as much

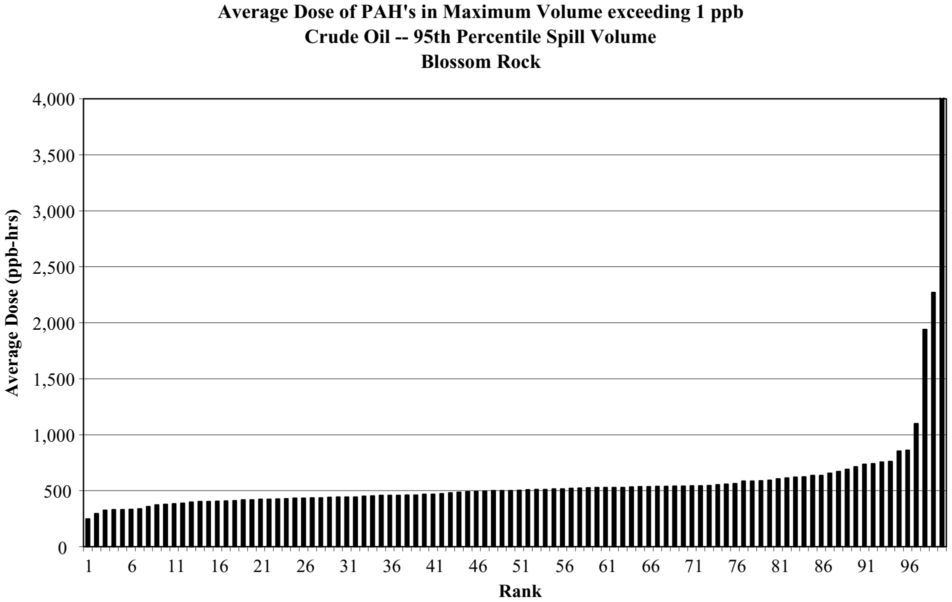

Fig. 6. Exposure dose of dissolved aromatics (ppb-h) in the water volume exposed to >1 ppb of dissolved aromatic concentration at some time after the spill.

Table 7
Range of surface water and shoreline exposure to oil for the spill sizes from Shag or Blossom Rock

	Surface water > 1 g/m^2 (millions m^2-h)			Shoreline > 100 g/m^2 (thousands m^2)		
	20th[a]	50th[a]	95th[a]	20th[a]	50th[a]	95th[a]
Gasoline	10–4,000	10–8,000	20–12,000	0.1–11	0.4–100	2–400
Diesel	31–13,000	90–20,000	130–82,000	3–400	60–1000	100–2500
Crude oil	121–40,000	310–85,000	560–84,000	10–400	50–1200	100–3000
Heavy fuel oil	110–7,000	320–10,000	550–6,800	10–300	40–800	100–2000

[a] Percentile size.

as 2–3 million m^2. Gasoline would only be expected to oil as much as 400,000 m^2 in a worst case event. Gasoline and diesel would evaporate off the shoreline rapidly, while the crude and heavy fuel would remain on shore until it is cleaned up. (For detailed results, see French McCay et al. [1].)

For the 95th percentile spill volume of gasoline, diesel and crude, the water volume exposed to >1 ppb at some time after the spill is on the order of 10^9 m^3. For heavy fuel oil, where the 95th percentile spill volume is smaller, the exposure volume is on the order of 10^7 m^3. The average dose in that volume is used as an index of exposure to determine the relative impact. In order to evaluate actual water column impact, the space- and time-varying concentrations need to be examined in sub-volumes of the exposed volume and compared to toxicity data. This is performed in the biological effects model (results discussed below).

The percent of spilled hydrocarbon mass reaching the sediments was evaluated. For gasoline, diesel and heavy fuel oil, the percentage is <1% for all runs. For crude, the percentage is <1% for most runs, but there are rare events where significant amounts of oil reach the sediment. These are high wind events causing high waves that entrain oil, resulting in high sedimentation in shallow water when the wind subsides.

For the heavy fuel and crude oil, environmental costs are largely driven by the impacts of surface oil, particularly by the shoreline cleanup costs. The wildlife and habitat impacts are generally proportional to shoreline oiling and cleanup costs. Thus, the 50th and 95th percentile runs were selected based on the frequency distribution of the shoreline cleanup costs. The order of model runs from lowest to highest impact is very similar for area of shore oiled by >100 g/m^2 and cleanup costs, varying only by the differences in cleanup costs per unit area for different shore types [3].

For the diesel and gasoline spills, cleanup costs are much lower [3] because there is much less oil that remains on the water surface and shorelines after the rapid evaporation period just after the spill. In addition, diesel and gasoline are much more easily entrained and dissolved into the water and potentially cause more water column effect than the heavier oils. Thus, theoretically, the environmental costs are more driven by the NRDA costs for impacts to the fish and invertebrates in the water than would be the crude and heavy fuel oil scenarios. Using this reasoning, the index for water column effects, the dissolved aromatic dose (ppb-h) in the volume of water where concentration exceeds 1 ppb at some time after the spill, was used to identify the 50th and 95th percentile runs to be examined further. The expectation was that water column impacts would be significant for the large spills, and these would dominate the NRDA costs. However, the results did not bear this hypothesis out, and the patterns are more complicated (as will be discussed below).

4.2. Biological impacts

The majority of the estimated killed birds are waterfowl (diving ducks and grebes), seabirds (murres), and shorebirds (sandpipers). The species impacted most agree with experience in oil spill cases in and near San Francisco Bay. Murres are commonly the most impacted species and the focus of restoration efforts in compensation for spill injuries.

There is a large variability introduced by variation in the month of the spill. The month has implications for temperature, which affects the rate of evaporation, but it is particularly significant to the biological impacts. The birds are highly variable in abundance by month of the year (Tables 4 and 5). Waterfowl (diving ducks, loons and grebes) are about 10 times more abundant in fall and winter than in spring–summer. Shorebirds are also more abundant in fall and winter. Outside San Francisco Bay, seabirds are five times as abundant in summer as in winter, whereas inside the bay seabird abundance does not vary much seasonally. Seabird abundance in the bay is the same order of magnitude as outside the bay in winter. The high seabird abundance outside the bay in summer is primarily due to the common murre and cormorants. Thus, summer spills exiting the bay and winter spills would impact the most birds. This complicates the interpretation of the results. Given that different species are most abundant in different months of the year (Tables 4 and 5), it would be difficult to identify a single worst-case month for impacts to wildlife based on abundance. Waterfowl (diving ducks, loons, grebes) would be most impacted in late fall and winter, while the impacts to murres are highest in summer if the spill is carried out of the bay on an out-going tide before coming ashore (because of the higher abundance outside the bay).

The results of the 50th and 95th percentile model runs for a given scenario (i.e. spill site, oil type and size) were used to construct probability distributions of wildlife impacts for all possible environmental conditions as follows. The

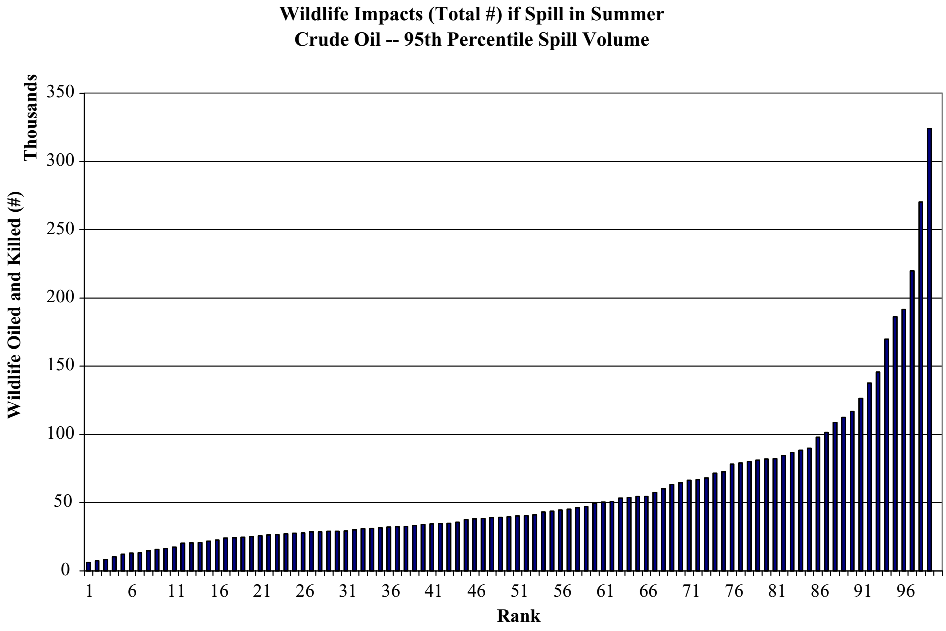

Fig. 7. Range of expected wildlife impacts for summer spills of crude (95th percentile volume) at Shag Rock.

water surface exposure (m^2-h, as in Fig. 3) and impacts for the individual model runs were used to calculate indices of wildlife oiled per m^2-h surface oil exposure in subtidal (water) areas. The area of shoreline oiled (m^2, as in Fig. 4) and number of shorebirds plus waders oiled for the individual model run provide an index of wildlife impacted per area of intertidal habitat oiled. The total wildlife impacted for each of the model runs was calculated from these indices and the degrees of exposure to floating and shoreline oil, generating a probability distribution for 100 potential environmental conditions that might occur after a spill of the specific volume and oil type at the spill site in consideration (Fig. 7).

If a scenario (i.e. spill volume, oil type, wind conditions, and current conditions) were to occur in a different month of the year, the impact to a species would change according to the ratio of abundance in the two months. In other words, the estimated wildlife kills are directly proportional to abundance. The probability distribution for other seasons is calculated using the ratios of abundance. Finally, a median and 95th percentile result is tabulated for each seasonal and as an annual mean. Tables 8 and 9 contain the annual mean results.

Table 8
Annual mean of estimated wildlife injuries for spills at Shag Rock

Spill size	Impact percentile	Gasoline	Diesel	Crude oil	HFO
20th	50	1,100	2,800	15,500	690
50th	50	6,000	13,300	15,300	2330
95th	50	5,900	21,000	30,700	3880
20th	95	5,400	8,700	60,600	1310
50th	95	32,100	40,900	75,400	4100
95th	95	27,600	62,300	121,100	6980

Table 9
Annual mean of estimated wildlife injuries for spills at Blossom Rock

Spill size	Impact percentile	Gasoline	Diesel	Crude oil	HFO
20th	50	150	9,800	44,900	1,800
50th	50	1,300	8,700	5,800	13,400
95th	50	7,000	23,700	17,600	11,800
20th	95	820	19,300	98,700	3,200
50th	95	5,200	38,500	13,900	23,200
95th	95	30,700	44,100	49,100	21,200

In the wildlife impact results, there remains considerable variability due to the exact pathway of the spill. This explains those results in Tables 8 and 9 where larger spill volumes did not oil more wildlife. The spatial variability in abundance is more influential on the result than area of water surface swept. The major uncertainty on the estimates is related to the abundance assumed. If the pre-spill abundance were, for example, a factor two different, the model kill estimate would change by that same factor.

Tables 10 and 11 summarize the model-estimated fish and invertebrate impacts for the simulations. The majority of the estimated killed animals are squid and small pelagic fish,

Table 10
Estimated total fish and invertebrate injuries for Shag Rock oil spills

Percentile volume	Percentile run	Gasoline	Diesel	Crude oil	HFO
95	50	0	860	2000	340
95	95	0.15	2000	2400	52
50	50	0	0.08	290	0
50	95	0.01	550	180	0
20	50	0	0.3	0.06	0
20	95	0	47	6.4	0

Table 11
Estimated total fish and invertebrate injuries for Blossom Rock oil spills

Percentile volume	Percentile run	Gasoline	Diesel	Crude oil	HFO
95	50	1.7	502	7000	200
95	95	13.5	29,600	7500	0.4
50	50	0.03	0.31	920	0
50	95	0	210	190	0
20	50	0	155	3.5	0
20	95	0.02	2.1	9.2	0

such as sea herring. Note again that if the pre-spill abundance were, for example, a factor two different, the model kill estimate would change by that same factor.

The only significant impacts to pelagic fish and invertebrates in the water, and demersal fish and invertebrates on the bottom and exposed to bottom water, were estimated to occur in the diesel and crude oil spills. The percent mortality of these organisms as a result of diesel and crude oil spills is estimated to be less than 10 percent in the volumes affected. The estimated impacts to water column organisms are very low, considering the large volumes of oil that is assumed released at 11–12 m below the surface. However, the currents are very strong, the water depth is very deep such that the dilution volume is large, and the natural dispersion is very rapid. Thus, even though the initial concentrations of dissolved aromatics are high, they decrease rapidly, diluting into a large volume and minimizing the impact.

It should be noted that these fish and invertebrate impacts were calculated assuming all the species were of average sensitivity to dissolved aromatics. Some species will be much more sensitive, and impacts to those species would be higher. There would also likely be species less sensitive than average. As there are insufficient toxicity data available to quantify the degree of sensitivity to aromatics for all species in San Francisco Bay, there is considerable uncertainty around the results based on average sensitivity. Experience with past modeling efforts indicate the uncertainty in the injury estimate related to species sensitivity is on the order of a factor ten higher or lower (95% confidence range). As there is a mix of species sensitivity present, the uncertainty in the total fish and invertebrate injury would be less than a factor 10.

The results indicate that seabirds impacts are relatively higher for the Shag Rock than the Blossom Rock spills, because the oil is more likely to exit the bay originating from Shag Rock and there are much higher abundances of seabirds outside the bay. The other wildlife impacts are relatively lower for the Shag Rock than the Blossom Rock spills, because of the much higher abundances of waterfowl and waders inside the bay. The fish and invertebrate impacts are in all cases much lower than for wildlife. For gasoline and heavy fuel oil spills, fish and invertebrate impacts are typically very low or insignificant, except in rare events for larger spills. (Details of the results are available in French McCay et al. [1]).

4.3. Costs

Tables 12 and 13 summarize NRDA costs for ecological damages (i.e. restoration costs, using REA costs for seabirds and HEA costs for other biota), along with estimated socioeconomic and response costs from Etkin [2,3]. The response costs and modeling of fates and effects assume a mechanical-only response strategy. It should be noted that the various cost categories are estimated using different economic methods, and are not the costs to any one entity or group of entities. However, they are totaled and compared to give a sense of total costs to the various entities involved. The socioeconomic and response costs are generally much higher than NRDA costs for the crude and HFO spills. This is because of the relatively large response effort and the persistence of the oil. For the light fuels where response costs are relatively lower, socioeconomic costs predominate. NRDA costs may be of similar order of magnitude under 95th percentile environmental conditions. However, the NRDA costs are typically 20% or less under 50th percentile environmental conditions. The NRDA costs vary considerably by spill condition, whereas the socioeconomic and response costs are more similar under 50th versus 95th percentile conditions. This is in accordance with available information on spill costs. Some cases involve high biological impacts and NRDA damages, while most are relatively modest in NRDA cost.

An interesting result is that the NRDA costs are, on average, modest in comparison to the socioeconomic and response costs. NRDA costs are often described as being very high for oil spills, much higher than response costs, and the largest monetary liability for a spill. These results show that it is often socioeconomic costs that dwarf the NRDA and response costs, and that response costs are usually much higher than NRDA costs. The exceptions are when birds are highly abundant, in certain months of the year (e.g. murres in summer and waterfowl in winter) or when aggregated in specific habitats (e.g. shorebirds in mudflats).

The relative costs of NRDA compared to response and third-party costs (which are a portion of socioeconomic costs) were investigated by Helton and Penn [23] for 48 spill incidents in the US, ranging from 5000 to 11 million gallons spilled, where NRDA claims were made. (NRDA claims are pursued in <1% of spills.) The modeled spills reported here were 25,000 to 3 million gal. Helton and Penn found that NRDA costs ranged from 3 to 95% of total known costs, with total known costs ranging from 0.3 to 183 million dollars for all but the *Exxon Valdez* spill (which totaled US$ 11,860 million [23]). The authors note that not all the response and socioeconomic costs are known, making their estimate of the NRDA share upwardly biased. The estimated costs for the modeled scenarios range from US$ 30–400 million, with NRDA costs 2–50% of these costs. Note that the 48 case histories were in various locations throughout the US and not in San Francisco Bay, and occurred in 1984–1997 in many cases when NRDA cost practices were not based on

Table 12
Summary of estimated NRDA costs for ecological damages, socioeconomic costs, and response costs (millions of 2001 US$) for spills at Shag Rock

Oil type	Volume percentile	Impact percentile	NRDA for ecological damages	Socio-economic costs	Response costs (mechanical)	Total costs
Gasoline	95	50	9	111	13	133
Gasoline	95	95	38	110	15	163
Gasoline	50	50	6	49	11	66
Gasoline	50	95	36	48	11	95
Gasoline	20	50	2	22	10	34
Gasoline	20	95	7	22	10	39
Diesel	95	50	34	135	27	196
Diesel	95	95	87	133	32	252
Diesel	50	50	15	53	19	87
Diesel	50	95	60	56	13	129
Diesel	20	50	3	28	12	44
Diesel	20	95	9	26	14	49
Crude oil	95	50	26	189	182	397
Crude oil	95	95	94	195	230	519
Crude oil	50	50	14	81	65	160
Crude oil	50	95	23	91	84	198
Crude oil	20	50	18	33	30	80
Crude oil	20	95	51	29	36	116
Heavy fuel oil	95	50	3	97	78	179
Heavy fuel oil	95	95	7	91	122	220
Heavy fuel oil	50	50	2	56	35	93
Heavy fuel oil	50	95	4	52	51	107
Heavy fuel oil	20	50	1	21	12	33
Heavy fuel oil	20	95	1	21	14	36

Table 13
Summary of estimated NRDA costs for ecological damages, socioeconomic costs, and response costs (millions of 2001 US$) for spills at Blossom Rock

Oil type	Volume percentile	Impact percentile	NRDA for ecological damages	Socio-economic costs	Response costs (mechanical)	Total costs
Gasoline	95	50	11	116	14	141
Gasoline	95	95	62	114	14	189
Gasoline	50	50	1	46	11	59
Gasoline	50	95	6	50	11	67
Gasoline	20	50	1	20	10	31
Gasoline	20	95	2	20	10	33
Diesel	95	50	29	130	30	189
Diesel	95	95	57	111	23	191
Diesel	50	50	13	54	14	81
Diesel	50	95	69	44	18	132
Diesel	20	50	13	23	11	47
Diesel	20	95	28	20	11	59
Crude oil	95	50	21	192	169	383
Crude oil	95	95	70	200	193	463
Crude oil	50	50	9	85	61	155
Crude oil	50	95	15	82	74	171
Crude oil	20	50	55	32	28	115
Crude oil	20	95	120	28	34	182
Heavy fuel oil	95	50	20	91	64	175
Heavy fuel oil	95	95	24	82	80	186
Heavy fuel oil	50	50	14	50	26	90
Heavy fuel oil	50	95	43	47	33	123
Heavy fuel oil	20	50	2	23	12	36
Heavy fuel oil	20	95	4	26	14	43

restoration costs. Never-the-less, the costs are of comparable magnitude and the modeled costs are reasonable predictions of potential costs in the near future if spills were to occur of this magnitude in San Francisco Bay.

5. Conclusions

Estimated impacts to birds ranged from a few hundred to nearly 200,000 birds, depending on the spill volume, month of the year, and environmental conditions that determines the locations and area swept by oil. There are several highly vulnerable species abundant in the area, including common murres, diving ducks, loons, grebes, and a variety of waders and shorebirds. Bird impacts were somewhat lower for the gasoline and heavy fuel oil spills examined (than for crude oil and diesel spills) because of the high volatility of the gasoline and the smaller potential spill volumes for the HFO.

In the central bay area that would be affected by spills resulting from groundings on the pinnacles, the water is very deep, currents are strong, and natural dispersion rates are high. Thus, the water column impacts of the spills examined were relatively low in consideration of the large volumes spilled and the assumption that the spill would occur at a depth of 11–12 m (such that the toxic components would dissolve in the water column more than for a surface spill where they would preferentially evaporate). These water column impact results indicate that the dilution capacity of central San Francisco Bay is high, and that impacts to water column resources would be significant only in rare incidents and for sensitive species. This result, in combination with the relatively high bird impacts predicted (and seen in many spills), suggests that use of dispersants in this area would be of net environmental benefit in reducing wildlife and shoreline impacts.

The impacts vary considerably by the month of the release, as the abundance of the most impacted group, the birds, varies by up to a factor of 10 on a seasonal basis. The results are also highly influenced by the particular path of the oil (i.e. in-coming versus out-going tide and wind conditions when the oil is released). Thus, an analysis of potential impacts of spills needs to describe this variability based on uncertainty of the model inputs and conditions at the time of the spill. The stochastic modeling approach used here provides the range of possible impacts and a statistical quantification of the variability. The statistical description could be expanded to include other uncertainties in model inputs, as well as model algorithms and assumptions (i.e. in a larger Monte Carlo type design).

The model estimates of NRDA costs were US$ 0.7–120 million. The total (NRDA + socioeconomic + response) estimated costs for the modeled scenarios range from US$ 30–400 million, with NRDA costs 2–50% of the total. Helton and Penn's [23] estimated total costs, and relative costs of NRDA compared to response and socioeconomic costs, indicate that the model estimates are reasonable if spills of these magnitudes were to occur in San Francisco Bay in the near future.

This work is significant as it demonstrates a statistically quantifiable method for estimating potential impacts that may be used in ecological risk assessment and cost-benefit analyses. The results of this study are being used by the Army Corps of Engineers San Francisco District in a cost-benefit analysis evaluating the trade-off of oil spill risk versus removal of rocks representing a hazard to shipping. The statistically-defined spill volumes and consequences provide an objective measure of the magnitude, range and variability of impacts to wildlife, aquatic organisms and shorelines for potential spills of four oil/fuel types shipped in the bay, each having distinct environmental fates and effects.

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