An Assessment of Applying Decision Analysis to Determining the Optimum Capping Level in the Disposal of Contaminated Sediments in Boston Harbor

by

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Chemical Engineering, B.S.
Massachusetts Institute of Technology, 1996

Submitted to the Department of Civil and Environmental Engineering In Partial Fulfillment of the Requirements for the Degree of

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ABSTRACT

This study examined the use of decision analysis in the recommendation of an optimal capping level in disposal of contaminated sediments in Boston Harbor. The research concentrated on validating the feasibility of applying a decision analysis methodology to such types of environmental issues.

Two contaminants were studied: Polycyclic aromatic hydrocarbons (PAHs) and Polychlorinated biphenyls (PCBs). Decision trees dealing with the bioaccumulation and the allowable water column concentrations of PAHs and PCBs were implemented.

By taking into account both disposal and environmental costs, factoring in user defined probabilities, and performing the appropriate calculations, it was determined that the optimal capping level for Boston Harbor was 1.5 meters.

Thesis Supervisors: Judith Pederson, Ph.D.: Manager, MITSG Center for Coastal Studies
David Marks, Ph.D.: Professor, Civil and Environmental Engineering
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- Xiao Mao (SS), whose love, caring, and spirit has been the light of my life. Thanks for all your wisdom, your contributions, and putting up with all my idiosyncrasies. Through tough times we shall conquer together....

- Steve, Ash, Jamie, for being a great friend and helping hand.

I would like to dedicate this thesis to my mother.... for her courage and her love.
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CHAPTER 1 – INTRODUCTION

The application of decision analysis to environmental decisions stems from the need to implement an unbiased, logical, and unequivocal methodology to address decisions that ultimately involve competing or conflicting values, sentiments, and interests. Decision analysis provides a way to use both factual and subjective information to determine the relative merits of alternative courses of actions. While decision analysis does not provide absolute solutions, it does however add valuable insights. The modeling approach invites policy-makers to explore disagreements and uncertainties about the subjective elements of the problem, thereby making decision-making more efficient. By integrating the participation of the parties with a stake in the environmental conflict at hand, and providing them with a systematic method to quantify and evaluate their viewpoints, decision analysis goes a long way in removing the political aspect of environmental decisions. One can imagine that where such decisions previously entailed emotionally charged debates, now are substituted by more rational policy analysis. In this sense, decision analysis serves a role much like that of a judicial process in the resolution of conflicts between individuals. Namely, a court decision is respected by the disputing parties largely because it is based on a set of rules both parties recognize, applied through a procedure that both parties are prepared, before knowing its outcome, to accept as unbiased. Of course, such analysis involves a calculus that must take into account intangible factors and provide a mechanism to evaluate ecological phenomena.
hierarchically. Indeed, at its base, environmental dilemmas revolve around the rationing of nature and its resources, the assessment of the benefits and costs of its use and destruction, and the placement of economic values to these factors. Moreover, in that environmental decisions invariably deal with issues that are controversial and highly publicized, it is important that the analytical framework adds legitimacy in the eyes of the public and of the various interests groups.

Decision analysis is not a new science. However, it has only been recently that decision analysis has been applied to the management of contaminated sediments. The first such use of decision analysis was in 1996 by Parametrix, Incorporated, in the case of the Asarco smelter site on Commencement Bay, Washington (Toll and Pavlou, 1992). There are numerous ways to make decisions in this field. Often the process is dictated by legal or political realities. In other scenarios, the process is directly a function of the complexity of the problem. Highly complex decisions necessitate not only the interpretation and analysis of a large amount of data, but also the reconciliation of considerable disagreement and uncertainty. Computer-based decision support adds tremendous value in situations where the issues are important and full of uncertainty, where there is political and emotional ramifications, and where the outcome must be acceptable to all parties. The management of contaminated sediments is certainly consistent with these criteria.
1.1 Statement of Objectives

The major objective of this thesis is to demonstrate the usefulness of and the feasibility of applying decision analysis to the disposal of contaminated sediments in Boston Harbor. Boston Harbor, for the past few hundred years, has been the receptor of largely unmitigated environmental pollution through sewage disposal, river runoffs carrying industrial contaminants, and various other forms of ecological degradation (Flores, 1998). Such practices led to extremely high levels of polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs), and metals in the sediments of Boston Harbor's seafloor. Research has shown that sediments are a potentially major contributor to the budgets of a variety of toxic chemicals in the harbor (Adams & Stolzenbach, 1998). These chemicals are available to organisms through ingestion and uptake from the water column. High levels of contamination are of particular concern in the context of the Boston Harbor Navigation Improvement Project (BHNIP), which encompasses the deepening of three tributary channels (Reserved Channel, Mystic River Channel, and Chelsea Creek Channel) and two areas in the Main Ship Channel (Inner Confluence and the mouth of Reserved Channel) to provide sufficient ship maneuvering areas (Figure 1; USACE & Massport, 1995).

BHNIP will result in the dredging and disposal of some 1 million cubic yards of contaminated sediments (USACE & Massport, 1995). The increased economic benefits of BHNIP – namely, increased navigational efficiency, ability to attract new shipping lines, and opportunity to remain competitive – are weighed against the potential ecological risks posed by such an activity. Among various alternatives explored for the disposal of this contaminated material, the in-channel disposal option – whereby cells will be constructed
Figure 1 – Boston Harbor Navigation Improvement Project

Source: USA CE & Massport, 1995

Source: USACE & Massport, 1995
and capped with sand after being filled with contaminated sediments – was selected as the most economically and environmentally viable solution. For a more detailed discussion of the options and process leading to a decision to use capped borrow pits for disposal, see USACE & Massport, 1995. The cap will isolate the silt material placed in the in-channel sites – reducing or eliminating the release of contaminants by advection to the water column above, benthic bioturbation effects, and resuspension and transport due to current and tides. Also, the cap will be designed to act as a barrier during future maintenance operations, preventing the dredge bucket from penetrating into the disposed material (Bellagamba, 1996).

Based on previous studies, it has been estimated initially that a 3ft or 1m sand cap will be necessary to isolate the contaminated dredge material (USACE & Massport, 1995). This value was estimated by assuming a minimal 25 cm cap thickness to protect against burrowing benthos, doubling that figure to account for chemical isolation, and then doubling once more to factor in dredging disposal inaccuracies (Shull and Gallagher, 1998). However, a systematic determination of the exact capping thickness necessary and a validation of the initial assumptions have yet to been performed to determine an environmentally sound cap thickness for different areas.

This research initiative will examine economic and environmental factors which affect different capping scenarios as applied to Boston Harbor. A decision analysis model is constructed to investigate the protection of marine organisms from bioaccumulation of
contaminants, and the isolation of the contaminants so that their concentrations in the water column will not exceed EPA specified levels. Contaminants studied are selected PAHs and PCBs. By implementing decision analysis, an optimum capping level will be suggested. Note that this thesis is less concerned with a precise quantitative representation of the Boston Harbor scenario, but rather more interested in assessing whether decision analysis will be useful for policy-makers. As such, the values – especially the some of the user-defined probabilities - utilized in the model may need sensitivity analysis to validate the assumptions made. Since the application of decision analysis to the disposal of contaminated sediments is a novel field, one must first demonstrate its usefulness in facilitating decisions, incorporating user participation, and making the entire policy-making process more efficient and rational. Future work is needed to expand and apply the model to more complex situations, with more exact numerical values.
CHAPTER 2 – DISPOSAL OF CONTAMINATED SEDIMENTS IN BOSTON HARBOR

2.1 Background

Since before the American Revolutionary War of the 1770’s, Boston Harbor has been a mecca for trade, shell-fishing, and shipping. Pollution of the harbor has been ongoing for the past three and a half centuries – from butchers dumping their offal, to the introduction of solid sewage wastes (USACE & Massport, 1995). Over the last twenty years, in the context of increasing environmental awareness, serious efforts – such as the Boston Harbor Navigation Improvement Project and the upgrade of sewage treatment – have been initiated to remediate the harbor. In conjunction with these projects, research and monitoring is being conducted to study the effects of sewage treatment upgrade and dredging and disposal of contaminants of the inner harbor.

The goal of the BHNIP is to deepen the channels of the harbor so that sea-faring ships can have easier access, thereby expanding Boston’s importance as a port city. However, one of the major ecological issues associated with this and most urban harbor dredging projects is how the displacement, removal, transport, and disposal of contaminated sediments will affect marine biota and human health. As stated previously, a capping methodology has been suggested to dispose of the dredged materials.
2.1.1 Physical Setting of Boston Harbor

Boston Harbor is located at the Northwest corner of Massachusetts. Two major passageways - President Roads and Nantasket Roads Bay (see Figure 2) - connect the harbor to Massachusetts Bay (Adams and Stolzenbach, 1998). Boston Harbor's circulation is quite complex and influenced by the many islands and peninsulas that exist within. The harbor occupies approximately 110 km$^2$ and encompasses a total shoreline length of approximately 190 km (Adams and Stolzenbach, 1996). Public works projects such as the construction of Logan Airport and the damming of Charles River, have significantly altered the shorelines of Boston Harbor. The maximum depth in the harbor is only 4.9 m (Adams and Stolzenbach, 1998). President and Nantasket Roads constitute the deepest portions of the harbor (outside of the dredged ship channels) at depths of about 15-25 m (Alber and Chan, 1994). Tidal range is 2.7m on average and the high tide volume of the harbor is about $8 \times 10^8$ m$^3$. Adams and Stolzenbach (1998) estimate fresh water inputs (44 m$^3$/s per year) into the harbor come from three major sources: (1) four major tributary rivers (21 m$^3$/s); (2) the sewage treatment plants at Deer and Nuts Island (17 m$^3$/s); and (3) groundwater inflows (1 m$^3$/s). Direct precipitation, runoff through storm drains, and combined sewer overflows make up the balance of the fresh water inputs (5 m$^3$/s) but the amounts are small compared to the other sources (Leo, 1993).

2.1.2 Boston Harbor Seafloor Conditions

In terms of the modern seafloor environment and sedimentary system of Boston Harbor, results obtained from sidescan sonographs and supplemental bathymetric, sedimentary, sub-bottom, and bottom-current data have indicated that there are three categories of modern seafloor sedimentary environments (Knebel, 1995).
Figure 2 – Map of Boston Harbor Area

Source: Adams and Stolzenbach, 1998
These are: “(1) environments of erosion or non-deposition comprised of exposed
bedrock, glacial drift, coarse lag deposits, and possibly coastal plain rocks
that contain sediments (where present) ranging from boulder fields to sandy
gravels to gravelly sands with mega-ripples and that occur in areas of
relatively high energy;
(2) environments of deposition that are blanketed by muddy sands, sandy
muds, and muds that have accumulated under dominantly weak bottom
currents; and
(3) environments of sediment reworking that contain sediment patches with
diverse grain sizes (ranging from sandy gravels to muds) that have been
produced by a combination of erosion and deposition and occur in areas
with variable bottom currents” (Knebel, 1995).

Therefore, the bottom environment of Boston Harbor is dynamic and suggests
considerable sediment movement in areas where there is erosion and reworking. This
needs to be taken into account when siting in situ disposal sites.

2.1.3 Sources of Contamination in Boston Harbor

Contaminants enter Boston Harbor via various pathways: sewage and sludge
discharges from treatment plants, tributary rivers, groundwater flows, runoff from storm-
water drains and combined sewer flows, and directly from the atmosphere (Adams and
Stolzenbach, 1998; Leo, 1993).
The major source of the discharge of "conventional" pollutants, total solids (TSS), biochemical oxidation demand (BOD), and nutrients (nitrogen and phosphorous; Adams and Stolzenbach, 1998) originates from sewage and sludge discharges from the treatment plants at Deer and Nuts Islands. Total input of solids to the harbor is estimated to be $43 \times 10^6$ kg/yr. Toxic metals – such as copper, lead, cadmium, chromium, mercury, nickel, and silver – are discharged into Boston Harbor. Note that adequate data exist only for zinc, copper, and lead which reports annual harbor deposition of 14 000 kgs, 25 000 kgs, and 21 000 kgs, respectively (Alber and Chan, 1994).

Boston Harbor is also contaminated from with polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), pesticides, and petroleum hydrocarbons (oil and grease). Of these, only PAH inputs into the harbor (20 000 kg/yr) are reliably calculated (Flores, 1998).
2.2 Various Areas of Research Associated with the Boston Harbor Navigation Improvement Project (BHNIP)

The Massachusetts Institute of Technology (MIT) Sea Grant College Program has been involved in a five-year multidisciplinary research project to better understand how contaminants are transported and transformed in Boston Harbor sediments, and how to best dispose of the dredged contaminated soils from the Boston Harbor Navigation Improvement Project (BHNIP). Specifically five projects deal with the following issues:

- sediment and contamination behavior during dredging and disposal
- effectiveness of caps
- changes to continuous loading of both metals and organic chemicals
- effect in the biota
- development of tools to in making decisions under uncertainty and risks

(see http://web.mit.edu/seagrant/www)

One of these projects involves the field monitoring of sediment dispersal during and following capping operations. The objective is to perform mass budget estimates of sources and sinks of organic contaminants associated with dredging activities. Specifically, the study aims to contrast various ways by which PAHs are introduced to the water column: dredging effects will be compared with advection, sediment-water exchange, air-water exchange (volatilization), internal losses (photo- and bio-degradation)
and other effluent sources. In addition, MIT Sea Grant is funding a project to monitor and model TSS (total suspended solids) due to vessel movement and dumping operations (see Flores, 1998; Israelson et al., 1998). This initiative will examine the effects of dredging, dumping, and capping operations (both with and without an environmental dredge bucket) on TSS to the effects generated by passing ship traffic.

Experimental and numerical modeling of the short-term fate of particles clouds has also been initiated. The goal is to evaluate the fate of contaminated sediment and/or capping materials within the water column. Short-term behavior of dredged sediments and capping materials as it relates to the loss of fine materials to the water column and the ability to accurately place sediments and capping materials will be the focus of the research.

(For a complete description and updates of these various projects, please visit the MIT Sea Grant and Marine Center websites at http://web.mit.edu/seagrant/www)

2.2.1 Evaluation of Environmental Risks Associated with Capping Contaminated Sediments

The major thrust of this thesis – in an attempt to tie in research with management aspects – addresses the issue of evaluating environmental risks associated with the capping of contaminated sediments. Specifically, the current research initiative introduces the application of decision analysis to the process of assessing risks associated with different
capping levels (i.e. $\leq 1.0\text{m},$ between $1.0\text{m}$ to $1.5\text{m},$ or $\geq 1.5\text{m}$). Decision-making processes are often fueled by economic and political motivations, frequently lacking sound scientific data and analysis. New approaches to risk assessment and management in environmental regulatory decision-making are being developed in an effort to institutionalize a more formal process which integrates available scientific data with regulatory and ecological concerns.

The Boston Harbor Navigation Improvement Project (BHNIP) will be disposing of contaminated materials in borrow pits, large trenches in the main shipping channel extending 20-40 feet below the channel depth (USACE & Massport, 1995). Regulatory agencies require a one meter cap of sandy materials to cover the contaminated materials and bring the top of the trench to the bottom of the harbor. The cap is estimated to increase the BHNIP cost from $40$ million to $60$ million (pers. comm., J. Pederson, MIT). According to the National Research Council (1994), decision analysis has rarely been, if ever, applied to dredged material management. This thesis examines the feasibility of using decision analysis to evaluate different capping levels and the associated risks. The aim is to create a methodology that can be applied to future, more complex situations.

Before a discussion of decision analysis and its application to the Boston Harbor capping scheme (Chapters 3 and 4), the proposed capping of dredged materials is described and then a value hierarchy (i.e. weighting system) of the various pertinent economic, social and environmental issues is established.
This hierarchy is a vital initial step to the creation of the decision analysis model and to the interpretation of its results.
2.3 Ocean Disposal of Contaminated Sediments

There are both immeasurable, as well as unimportant potential effects of ocean disposal of dredged materials on marine organisms and humans. These effects may differ at each disposal site, and under current practices are evaluated on a case-by-case basis for major dredging projects. Section 103 of the Marine Protection, Research, and Sanctuaries Act of 1972 (MPRSA), Public Law 92-532, specifies that all proposed operations involving the transportation and dumping of dredged material into ocean waters have to be evaluated to determine the potential environmental impact of such activities. The U.S. EPA and the U.S. Army Corps of Engineers have adapted a tiered approach for the evaluation of dredged materials proposed for ocean disposal (USEPA & USACE 1991). The manual outlines specific protocols, criteria, limits, and methodologies to be met in terms of protecting marine biota and minimizing human risks. By discussing these criteria, a method is proposed to help decision-makers conceptualize the type of experiments that needs to be conducted, identify data that need to be gathered, and recommend analyses that need to be performed in order to derive a value hierarchy to resolve environmental disputes (USEPA & USACE, 1991).

2.3.1 Environmental Impact

There are four areas identified for which criteria are adopted; trace contaminants, bioaccumulation, water-column concentrations, and the benthic community.

Trace Contaminants

The EPA prohibits dumping of certain constituents other than trace contaminants unless they are rapidly rendered innocuous (USEPA & USACE, 1991). According to the
EPA: “Trace contaminants are not defined in terms of numerical chemical limits, but rather in terms of persistence, toxicity, and bioaccumulation that will not cause an unacceptable adverse impact after dumping” (USEPA & USACE, 1991). Indeed, the evaluation process emphasizes potential biological effects – as measured by mortality (bioassay) and uptake (bioaccumulation) – rather than chemical presence, of the possible contaminants. It is generally agreed that both bioassay and bioaccumulation tests are imprecise predictors of environmental effects; nonetheless, they are regarded as the best methods available for integrating and evaluating biological effects of multiple contaminants. Appropriate sensitive test organisms are utilized for bioassay purposes (USEPA & USACE, 1991).

The mortality rate observed in a laboratory test may not mean that the population of that species around the disposal site would decline by the same percent if the proposed disposal occurs. However, a comparison of dredged-material and reference-sediment bioassay results may be used to ascertain whether the dredged material displays significantly higher toxicity (USEPA & USACE, 1991).

**Bioaccumulation**

Bioaccumulation or the retention of contaminants may have ramifications for marine organisms and human consumers. It reflects the biological availability of contaminants in the dredged material, as well as the potential for long-term storage of these contaminants within the tissues of aquatic organisms up to levels that might be harmful to consumers (USEPA & USACE, 1991). Pathways for bioaccumulation include the diffusion of contaminants from pore waters or supernatant water into gill tissue or
exposed integument (external body wall), and through the gut wall via ingested/digested particulate materials.

To factor in the element of bioaccumulation in a decision, a causal relationship must be established between the organism’s presence in dredged material and a meaningful deleterious elevation of body contaminant levels above that of similar organisms not exposed to the dredged materials (USEPA & USACE, 1991). Quantification of this relationship is difficult but statistical confidence levels are employed.

**Limiting Permissible Concentration – The Water Column**

After taking into account initial mixing, the limiting permissible concentration (LPC) is the concentration of any dissolved dredged-material contaminant that will not exceed applicable marine water-quality criteria (WQC; Table 1). The comparison of field data and the WQC levels will, in general, reflect the fate and transport properties of the contaminants with respect to ocean disposal (USEPA & USACE, 1991). A schematic of the transport and transformations of a pollutant in the water column is shown in Figure 3.

Section 304 (a)(1) of the Clean Water Act (33 U.S.C. 1314 (a)(1)) requires the Environmental Protection Agency (EPA) to publish and periodically update ambient water quality criteria. These criteria are to:

"accurately reflect the latest scientific knowledge (a) on the kind and extent of all identifiable effects on health and welfare including, but not limited to plankton, fish/shellfish, wildlife, plant life, shorelines, beaches, aesthetics, and recreation which may be expected from the presence of pollutants in any body of water; (b) on the concentration and dispersal of pollutants, or their by-products, through biological, physical, and chemical processes; and (c) on the effects of pollutants on biological community diversity, productivity, and stability”. (see USEPA website) Table 1 summarizes the water quality criteria of the various chemicals in question.
Figure 3 – Transport and Transformation of a Pollutant in the Water Column

Source: Leo, 1993
Table 1 – Water Quality Criteria

Note: The chart is for general information. Please see the criteria documents or detailed summaries in “Quality Criteria for Water Data” for regulatory purpose.

Source: USEPA & USACE, 1991
Because of comparison studies by the MITSG Marine Center, as test case examples polychlorinated biphenyls (PCBs) and selected polycyclic aromatic hydrocarbons (PAHs) are chosen to evaluate whether water quality criteria are violated under different capping scenarios.

- **Polychlorinated biphenyls**

  For PCBs, the criterion to protect salt-water aquatic life is 0.030 μg/L as a 24-hour average. This concentration based on bioconcentration factors measured in lab experiments, but field studies produce factors at least 10 times greater for fishes. The available data indicate that acute toxicity to salt-water aquatic life probably will only occur at concentrations above 10.0 μg/L (USEPA & USACE, 1991).

  For the maximum protection of human health from the potential carcinogenic effects of exposure to PCBs (16 species) through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentration should be zero. A value of approximately 0.79 ng/L is recommended.

- **Polycyclic aromatic hydrocarbons**

  The available data for PAHs (2-methylnaphthalene and benzo(a)pyrene) indicate that acute toxicity to saltwater aquatic life occurs at concentrations of 300 μg/L. For the maximum protection of human
health from cancer effects, the recommended level is 28.0 ng/L (USEPA & USACE, 1991).

Benthic Community

Research conducted by the U.S. EPA has demonstrated that the greatest potential for environmental impact from dredged material is to the benthic community. Two factors contribute to the benthic community effects: (1) deposited dredged material is not mixed and dispersed as quickly or as greatly as the portion of the material that may remain in the water column; and (2) bottom-dwelling animals live and feed in and on deposited material for extended periods (USEPA & USACE, 1991). It is recommended that major efforts be placed on evaluating the quality of deposited material and potential effects to the benthic community in considering the management of contaminated sediments. As such, data for the proposed dredged material for the specified disposal period should be established for: (a) a test-species benthic toxicity (bioassay test), and (b) test-species benthic bioaccumulation period for each contaminant that is likely to be of concern at that site.

2.3.2 Approximate Confidence Levels for Toxicity and Bioaccumulation

For the purposes of evaluating cap thickness, the DA approach uses the following confidence/risk levels which implies only the most stringent control and acute toxicity, based on U.S. EPA suggested criteria:

- If the mortality of organisms exposed to the dredged material is statistically greater than those exposed to the reference sediment and exceeds the reference sediment mortality by at least 10%, the dredged material is deemed to exceed the limiting permissible concentration and does not comply with the benthic bioassay criterion (Cohrssen, 1989; USEPA & USACE, 1991).
Bioaccumulation will be measurable after a 28-day exposure period. One evaluation examines concentrations of contaminants of concern in tissues of benthic organisms compared to pertinent Food and Drug Administration Action Levels for Poisonous and Deleterious Substances in Fish and Shellfish for Human Food. These levels are the limits that the FDA has set so that noncompliance implies legal remedies. The levels are based on human health as well as economic considerations. Note that they do not reflect environmental impact on the organism itself. Therefore, the following conclusion is reached:

- If tissue concentrations of one or more contaminants of concern are statistically greater than applicable FDA action levels: then, the dredged material exceeds the limiting permissible concentration (LPC) for bioaccumulation and does not comply with the bioaccumulation aspects of the benthic criteria (Cohrssen, 1989; USEPA & USACE, 1991).

Table 2 lists the Food and Drug Administration (FDA) Action Levels for Poisonous and Deleterious Substances in Fish and Shellfish for human consumption (USEPA & USACE, 1991). Notice that the values for hexachlorobenzene (a PAH) and polychlorinated biphenyls (PCBs) are highlighted; this thesis initiative uses PAHs and PCBs as benchmarks to test the validity of applying the decision analysis model to the capping methodology. Note that although values for the PAHs (2-methylnaphthalene and benzo(a)pyrene) selected in this thesis are not expressly listed by the FDA, the hexachlorobenzene value can be used as a base estimation.
Table 2
Food and Drug Administration (FDA) Action Levels for Poisonous and Deleterious Substances in Fish and Shellfish for Human Food

<table>
<thead>
<tr>
<th>SUBSTANCE</th>
<th>ACTION LEVEL (ppm)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Metals</td>
<td></td>
</tr>
<tr>
<td>Methyl Mercury</td>
<td>1.0</td>
</tr>
<tr>
<td>Pesticides</td>
<td></td>
</tr>
<tr>
<td>Benzene Hexachloride (BHC)</td>
<td>0.3</td>
</tr>
<tr>
<td>Chlordane</td>
<td>0.3</td>
</tr>
<tr>
<td>Chlordecone (Kepone)</td>
<td>0.3</td>
</tr>
<tr>
<td>DDT + DDE</td>
<td>5.0</td>
</tr>
<tr>
<td>Dichlorophenoxyacetic acid</td>
<td>1.0</td>
</tr>
<tr>
<td>Dieldrin + Aldrin</td>
<td>0.3</td>
</tr>
<tr>
<td>Endrin</td>
<td>0.3</td>
</tr>
<tr>
<td>Fluridone</td>
<td>0.5</td>
</tr>
<tr>
<td>Heptachlor + Heptachlor Epoxide</td>
<td>0.3</td>
</tr>
<tr>
<td><strong>Hexachlorobenze (HBC)</strong></td>
<td><strong>0.3</strong></td>
</tr>
<tr>
<td>Isopropyliamine</td>
<td>0.25</td>
</tr>
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</tr>
<tr>
<td>Dioxin</td>
<td>25.0 ppt</td>
</tr>
</tbody>
</table>

Source: USEPA & USACE, 1991

The octanol/water partition coefficient ($K_{ow}$) of organic compounds can be used to estimate the relative potential for bioaccumulation (USEPA & USACE, 1991). The U.S. EPA recommends that compounds for which the log $K_{ow} > 3.5$ be indicated for further study. Table 3 lists the $K_{ow}$ values of selected compounds.

Another approach is to calculate a theoretical bioaccumulation potential (TBP) for non-polar organic chemicals. If the dredged material in question is assumed to be the only
source of contaminant to the organism, there is some confidence in this value. The TBP is in essence an approximation of the equilibrium concentration in the organism's tissues. The rationale behind the TBP calculation is beyond the scope of this thesis but can be referenced in (Campbell & Pavlou, 1982; USEPA & USACE, 1991). The TBP, which is expressed on a whole-body wet-weight basis, is calculated as follows:

\[
\text{TBP} = 4 \cdot (C_s / \%\text{TOC}) \cdot \%L \quad \text{(Formula 1)}
\]

Where,
- \(C_s\) is concentration of non-polar organic chemical in the dredged material or reference sediment (any units of concentrations may be used);
- \%TOC is total organic carbon content of the dredged material or reference sediment expressed as a decimal fraction; and
- \%L is organic lipid content as a decimal fraction of whole body weight.

Typical values are 1 - 4%TOC and 5 - 25%L. Lipid content is rarely measured and this approach is not generally used. For this study, a lipid content of 25% was assumed as a worst case scenario.
Table 3 – $K_{ow}$ Values

<table>
<thead>
<tr>
<th>Pollutant</th>
<th>Octanol/Water Partition Coefficients (log $K_{ow}$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Indeno (1,2,3-cd) pyrene</td>
<td>7.7</td>
</tr>
<tr>
<td>PCB-1260</td>
<td>6.9</td>
</tr>
<tr>
<td>Benzo(k) fluoranthene</td>
<td>6.8</td>
</tr>
<tr>
<td>Benzo(b)fluoranthene</td>
<td>6.6</td>
</tr>
<tr>
<td>PCB-1243</td>
<td>6.1</td>
</tr>
<tr>
<td>Benzo(a) pyrene</td>
<td>6.0</td>
</tr>
<tr>
<td>PCB-1242</td>
<td>6.0</td>
</tr>
<tr>
<td>Dibenzo(a,h) anthracene</td>
<td>6.0</td>
</tr>
<tr>
<td>PCB-1016</td>
<td>5.9</td>
</tr>
<tr>
<td>Benzo(a) anthracene</td>
<td>5.6</td>
</tr>
<tr>
<td>Hexachlorobenzene</td>
<td>5.2</td>
</tr>
<tr>
<td>2-Chloronaphthalene</td>
<td>4.7</td>
</tr>
<tr>
<td>Chlorobenzene</td>
<td>3.8</td>
</tr>
<tr>
<td>Naphthalene</td>
<td>3.6</td>
</tr>
</tbody>
</table>

Source: USEPA & USACE, 1991
2.4 Capping Dredged Materials

There are in general three types of capping materials: inert materials, active materials, and synthetic liner materials. Inert materials include sand, silt, and clay and may be one of several states: fine-grained, coarse-grained, or uncontaminated dredged soils. Active cover materials include limestone, green-sand, oyster shells, alumina, ferric sulfate, and gypsum (SAIC, 1996; USDOC, 1988). The effectiveness of different cover/capping materials depends on a number of factors: "(1) turbidity and dispersion generated during application of the material; (2) impacts on benthic organisms; (3) scouring and re-suspension of cover materials once in place; and (4) resistance to leaching of contaminants" (USDOC, 1988). The capping material used in the BHNIP is a mixture of large grained sands and clays.

Capping with inert or active materials adds to the cost of a project. A critical question to be addressed, if capping is chosen as the ocean disposal method, is how much cap material is needed to accommodate burrowing effects, erosion from ocean waves, and the escape of contaminated compounds to the water column above. Traditionally, open ocean disposal projects begin with dredging and disposal of the most contaminated sediments first, followed by dredging and disposal of progressively cleaner sediment, culminating in coverage with clean capping material. In the current research initiative, the disposal method is one of burrow pits and capping. There will be initial smothering and burial of bottom-dwelling animals; these effects are beyond the scope of this thesis.
Historically, ratios of capping material to contaminated sediment range from 1:1 to 11:1 (volume based) with most cap volume ratios falling between 3:1 and 6:1 (SAIC, 1996).

The addition of capping material can add significantly to the overall cost of a dredging project. As such, it is vital to make the cap only as thick as required for minimizing exposure risk. Studies have indicated that a 1 to 2m thick sand cap may be sufficient (SAIC, 1996). However, the exact cap thickness needed differs with each site and depends on both the hydrographic setting and the seafloor characteristics. A panel of experts was assembled to review and recommend a process for determining appropriate cap thickness for the New York Bight (SAIC, 1996). Below are the issues considered:

- minimum thickness of capping material needed to comprehensively isolate contaminants buried below the cap from the aquatic food web;
- benthic community response to the disposal of dredged materials (beyond the scope of this thesis; for burial and smothering effects see reference (SAIC, 1996);
- benthic community response to the introduction of capping material – initially, organisms will be buried but there are long-term considerations;
- speed at which disposal sites are colonized, and do the colonizers take up contaminants – thereby, affecting seafood; and
- colonization effect on cap integrity.

The panel recommends that cap thickness be expressed as:

\[
\text{Final Cap Thickness} = T_b + T_i + T_e + T_c + T_e
\]

(Formula 2)

Where:
- \(T_b\) = an estimated thickness for biological isolation
- \(T_i\) = a calculated thickness for chemical isolation
- \(T_e\) = a predicted thickness for armoring against erosion
- \(T_c\) = a predicted thickness for consolidation
- \(T_e\) = a thickness to account for errors during dredging/dumping
2.4.1 Bioturbation and Capping

The concept of bioturbation alludes to the activities of marine organisms in seafloor sediments (Figure 4). These activities – burrowing, feeding, defecating, and irrigating – aid in the mixture of sediment particles, provide oxygen for sediment pore waters, and facilitate the direct exchange of pore waters with overlying seawater (SAIC, 1996). The species that are most likely to penetrate caps and recycle contaminants back into the water column are those that burrow deeply into the seafloor. They include macrofaunal deposit-feeders, deep burrowing megafauna (e.g. stomatopod shrimp, crabs, lobsters), and burrowing demersal fish (e.g. sand lance or tile fish). Many of these animals feed directly on the sediments, subsequently affecting the physical and geo-chemical properties of the sediments dramatically (SAIC, 1996). It is suggested in the literature that a bioturbation zone be set aside as part of the cap to account for active sediment or pore water movement (advection) due to biological activity. Within the bioturbation zone there will be witness, advective transport to and from the zone, active pore water movement, and oxidation/reduction reactions. Below the bioturbation zone, there should be little or no active water movement except that due to the consolidation of the dredged materials. Diffusive processes (molecular exchange) and anaerobic chemical reactions will occur on this part of the cap. In essence, if the structural integrity of this portion of the cap can be guaranteed, then the underlying contaminated dredged materials should be effectively isolated from marine animals for thousands of years (SAIC, 1996).
Figure 4 – Graphical Representation of Capping Disposal Option

Source: SAIC, 1996
According to studies in New York Bight, a practical capping thickness for biological isolation or the bioturbation zone (i.e. $T_b$) is 0.30 m – this zone is an effective barrier for macrofauna but does not account for megafauna and demersal fish. The 0.30 m value is 3-fold thicker than the “universal mean bioturbation depth” for macrofauna ($0.098 \pm 0.045$ m) and may vary with salinity. Although not extensively investigated in the study, it was projected that approximately an additional 0.70 to 1.0 m of cap thickness was necessary to account for chemical isolation and erosion and consolidation of cap materials. In oligohaline regions (< 5 ppt) an effective bioturbation capping thickness is estimated to be 0.15 to 0.30 m, in mesohaline regions (5-18 ppt), burrowing increases to 0.30 to 0.50 m while in polyhaline (18-30 ppt) and euryhaline (30-40 ppt) regions, large crustaceans that burrow to 0.50 to 1.00 m; all of which dictate cap thickness based on bioturbation (SAIC, 1996). To date, further work on how deep burrowing animals (such as stomatopod shrimp and tilefish) affect overal cap effectiveness, has yet to yield any conclusive results and is usually not taken into account.

2.4.2 Benthic Community Response to Disposal of Capping Material

Capping material, in general, engenders the same qualitative effect on benthic animals as dredged material. Over time, the animal community on a sand cap may converge with the community on bottoms in species composition, feeding type, and animal density. In essence, re-colonization will allow typical communities to be re-established. The thickness of sand cap is unlikely to alter any smothering or burial patterns. As such this factor is a minor, if not insignificant one in the proposed decision analysis model.
SAIC (1996) has summarized data indicating that while re-colonization of benthic animals profoundly affect the physical and chemical properties of the sediments, the effects are limited to bioturbation zone and do not affect cap integrity.
2.5 Constructing the Value Hierarchy

In order to conduct a decision analysis model for cap thickness as required in Boson Harbor, a value hierarchy has been constructed is an attempt to quantify some of the issues discussed in Sections 2.3 and 2.4. The assigning of numerical values or weights to economic, social, and environmental impact of different capping strategies is extremely difficult. There are implicit ambiguities and contradictions in quantifying phenomena which are qualitative observations and intuitive estimations. However, by recognizing an interactive process will yield better estimates, the exactness of the value scheme is a secondary consideration. The validity and soundness of the actual decision analysis model is of foremost consideration, and would typically involve several working sessions with experts. For this first attempt, the numbers used for the model can be adjusted in future sensitivity studies. The eventual goal of such a value hierarchy is to systematically minimize project cost (i.e. capping cost) while meeting the risk requirements for human health and marine biota survival.

2.5.1 Rationale for Assigned Capping Thickness

For simplicity of calculations, this thesis initiative has chose five capping levels to apply decision analysis to: (a) no cap; (b) 0.5 meters; (c) 1.0 meters; (d) 1.5 meters; (e) greater than 1.5 meters. These particular thicknesses were chosen in accordance with New York Bright findings (see Section 2.4.1) for the minimal capping needed to account for bioturbation zones consistent with macrofauna, megafauna, and demersal fish and large crustacean behavior. As initial benchmarks, the following examples are chosen:
- **Macrofauna**

  Final Cap Thickness = \( T_b + (T_i + T_e + T_c + T_\cdot) \)

  \( = 0.30 \text{ m} + (0.7\text{ m}) \)

  \( = 1.00 \text{ m} \)

- **Megafauna**

  Final Cap Thickness = \( T_b + (T_i + T_e + T_c + T_\cdot) \)

  \( = 0.50 \text{ m} + (0.7\text{ m}) \)

  \( = 1.20 \text{ m} \)

- **Demersal fish and Large Crustaceans**

  Final Cap Thickness = \( T_b + (T_i + T_e + T_c + T_\cdot) \)

  \( = 0.80 \text{ m} + (0.70\text{ m}) \)

  \( = 1.50 \text{ m} \)

Note that the varying biological isolation thickness \( (T_b) \) account for the differences in burrowing capabilities of macrofauna, megafauna, and demersal fish and large crustaceans. The thicknesses for chemical isolation \( (T_i) \), erosion protection \( (T_e) \), and consolidation effects \( (T_c) \) were lumped together and assumed to be 0.70 m (see SAIC, 1996). This assumption is site-specific and may change with other disposal options, but is a reasonable first estimate for BHNIP.

Initially, a 1 meter cap is required for Boston Harbor based on the assumption that the bioturbation zone is 0.25 m and doubling that value to account for consolidation and dredging/dumping errors (see Section 4.3.4).

### 2.5.2 Economic Value Attribute

Engineers and policy-makers involved in the analysis and decision-making of environmental issues are challenged to quantify risk and uncertainty, to make optimal use of resources including labor, material, capital, and technology, and to apply effectively the
techniques of engineering economics and cost analysis. Usually, this quantification becomes an economic cost-benefit assessment. Indeed, to measure the desirability of one alternative relative to others available, it is necessary to select a meaningful basis which must incorporate all of the independent economic variables. These variables include the best approximations of equity investment, cost of capital, operating costs, taxes, revenues, and the salvage value at the end of the economic life. Specific to the dredging of contaminated sediments in Boston Harbor, economic expenditures (dredging and capping the sediments) must be reconciled with moral and ethical considerations such as the value of protecting marine and human life, as well as with the value of protecting the marketability of the harbor as a commercial fishery and lobster habitat.
3.1 Definition/Justification for Using Decision Analysis

3.1.1 Defining Decision Analysis

Decision analysis is a relatively new field. Although its intellectual roots can be traced back hundreds of years, this field developed into an organized body of knowledge and methods only within the last twenty years. The term “decision analysis” (DA) implies a general methodology for modeling and analyzing decision situations. DA encompasses a wide variety of tools for structuring decisions, assessing subjective probabilities and preferences, and analyzing decision models. When decision analysis assists with making decisions in which the consequences are unknow, DA structures the problem, defines optimal choices, and identifies the optimal strategy. Environmentally speaking, the probability of harm or risk plays a major role in determining what choices decision-makers will make. The use of mathematical modeling methods permits the overall probability of harm to be computed from statistical judgments made on specific scientific issues.

There is a general agreement in the literature that the role of analysis should be to organize information, not to provide formula that would compel the decision-maker to choose a certain path (Dawes, 1988). According to the consensus in the field,
environmental legislation, executive orders, and court decisions have generally supported additional levels of analysis, but have provided little direction to regulatory agencies as to how analysis should be used in their decision processes. Decision analysis may be regarded as an extension of cost-benefit or cost-effectiveness analysis to include a quantification of uncertainty. Like cost-benefit analysis, decision analysis frequently involves issues such as environmental and property damage, mortality, and aesthetic and political concern. Mathematical modeling methods and sensitivity analysis can help to determine which of the many uncertainties and value issues are most important in determining the best decision alternatives.

In environmental scenarios such as the management of contaminated sediments in Boston Harbor, decision analysis defines a specific regulatory choice based on determination of unreasonable risk. The basic concepts of the decision analysis methodology include: (1) the definition of alternatives, (2) the use of judgmental probability to quantify uncertainty, (3) the evaluation of health and environmental damage and costs, and (4) representation of the value of additional information that reduces or removes uncertainty. The concepts of decision analysis not only apply to individual decisions, but they also provide a basis for priority setting and coordination of decisions affecting a multitude of phenomena (Campbell & Cohen, 1982).

In essence, decision analysis is in tune with the move away from deterministic models (deNeufville, 1990). Decision situations can be classified into those with assigned probabilities for future events and those without assigned probabilities, i.e. these
classifications are typically expressed as the set of possible decisions $D(i)$, the set of uncertain events $E(j)$, and the set of outcomes $O(i,j)$. Decision analysis also takes into consideration elements of game theory, which involves the interaction among two or more decision-makers.

3.1.2 Justifying Using Decision Analysis

The distinction between decision process and decision outcome is very important. Even the most carefully thought out decision process can be derailed by an unlucky outcome. The question is raised as to why decision analysis techniques should be used? Usual answers include "gaining insight" and "being coherent", but the best reason to use a particular technique would be that doing so would maximize the chance of achieving the outcome one wants.

Undoubtedly, decision analysis can show an individual how to be coherent in making inferences and choices. That is, adherence to DA theory principles will confirm that one's decisions will not be self-contradictory. However, assessing whether decision analysis is truly effective is a research project fraught with challenges. The answers are beyond the scope of this thesis which accepts the fact that decision analysis is imperfect, that it can not offer an absolute and precise forecast of future outcomes, and that it serves primarily in the Boston Harbor Project as an aid to making decisions in the management of contaminated sediments.
3.1.3 Fundamentals of Decision Analysis Methodology

Perhaps the most important aspects of decision analysis are the usage of probability and statistics to analyze past data, to forecast future events, and to make decisions accordingly. There are certain fundamentals to incorporating probability into decision analysis that needs to be addressed (deNeufville, 1990): Expected Value (EV), Expected Value of Perfect Information (EVPI), and Decision Trees.

**Expected Value (EV)**

An expected value calculation requires that a decision-maker first have an intuitive estimate of the probability of each state occurring. These probabilities may be subjective or may reflect personal preferences, personalities, or individual interests. The calculation simply is the product of the probability of a particular event and the payoff for a particular decision.

**Expected Value of Perfect Information (EVPI)**

Given that today's world exists within the context of an information age, one can utilize information to improve our decision-making. An important question is what is the value of the information. For instance, some information is free on the World Wide Web, but there is a search cost to find it. In other cases, one pays companies for information. Indeed, if one had the *perfect* information about the future, one would always make the right decision. The question that arises is what is the value of having the information that allows one to make perfect decisions. The expected value of perfect information (EVPI)
helps answer this question. EVPI assumes that one will make the “right decision” for each event. The calculation begins by determining the expected value of perfect prediction (EVPP). Then, the EV is subtracted from the EVPP to arrive at a value for EVPI.

**Decision Trees**

Decision trees are a graphic tool which describe the actions available to decision makers, the events that can occur, and the relationships between the two. Decision trees are critical to structuring a decision process, providing understanding of the interactions among decisions/alternatives and outcomes and the flow of the process. Decision or event trees provide a way of analyzing problems that involve an ordered sequence of decisions and chance outcomes that depend on earlier decisions and chance outcomes. The tree shows subjective probabilities (judged by experts) and magnitudes of the outcomes. Their depiction of the tree is usually computerized.

A decision tree consists of nodes and branches. A decision point is represented by a square box or decision node. For any particular decision point there may be several alternatives, which are the branches. A probability node is designated as a circle. The branches from any circles show the probability of an event occurring. Events signify what might happen (i.e. the random variable). They are shown as branches from circle event nodes and are assigned probabilities. The conditional profit is found at the end of each branch, for each alternative and event. The expected value is calculated for each event node and placed in the circle. Bars are used to denote non-optimal branches.
3.1.4 Primitive Decision Analysis Models

There are four traditional primitive decision analysis models (deNeufville, 1990 and Clemen, 1952):

1) **Maximax strategy** selects the act that maximizes the maximum profit – it accentuates a “go for broke” strategy and ignores the chances and consequences of other events. For each decision, the maximum payoff is elected. Then the maximum of the maxima is calculated.

2) **Maximin strategy** selects the act that has the smallest maximum loss. It tends to be overly conservative and ignores the potential for large gains. For each decision, the model first selects the minimum payoff and then takes the maximum of the minima.

3) **Minimax regret strategy** selects the act that minimizes the maximum regret. The degree of regret is the difference between the choice and the best choice. First, the maximum payoff for each state of nature (i.e. actual events that may occur in the future) is selected and then a regret table is calculated. In effect, the degree of regret is an opportunity loss.

4) **Average payoff decision strategy** selects the alternative that has the best average payoff, where each state of nature is assumed to have an equal chance of occurring. All decision analyses attempt to maximize expected utility.
3.1.5 Multi-attribute Decision Analysis

The standard decision analysis approach to examining a multi-attribute utility model assumes that preferences follow rational rules of choice. One implication of these rules is that the decision-maker’s preferences are known with certainty. Although external events are often viewed as uncertain, and therefore can be assigned probabilities, the value or utility of each possible outcome is represented by a single point estimate, not by a probability distribution of possible values (deNeufville, 1990). In multi-attribute decision analysis, preference assessment entails delineating a set of attributes describing the properties of outcomes, assessing single-attribute value functions over the levels of each attribute, and assessing attribute weights. This is the methodology employed in this thesis research initiative.
3.2 Examples of Application of Decision Analysis Models

Decision analysis methods applied to social decisions often involve entities that cannot be described as a single decision-maker. Typically, DA assumes that a single individual, or a group with consistent views, has responsibility and authority for making public decisions. In other cases, a "supradecision-maker", a benevolent dictator of sorts must create a social utility function from the preferences of the individuals affected rather than his own preferences (Campbell & Cohen, 1982; Feather & Harrington, 1985). This thesis initiative demonstrates that decision analysis is of particular utility when incorporating the input of multiple decision-makers, of multiple parties. This is a dynamic process and the beauty of DA is that its format allows for easy manipulation and restructuring of the model.

Interviews between decision-makers and experts help provide a more reliable outcome compared to a conceptual basis for the subjective probabilities and values utilized in environmental decision analysis. The process is dynamic, changes with the nature of the answers and the type of respondents, and can require hundreds of judgments and decisions from each individual. However, this very process still underlines the premise that the probabilities are extremely subjective. That is, they are individual beliefs or approximations of the likelihood of events rather than observed frequencies. For the purposes of this thesis, such dynamic and interactive elements (i.e. interviews) were not incorporated into the model. This will be left for future development of the decision analysis methodology as applied to the management of contaminated sediments.
Decision making under uncertainty exists everywhere in the real world. Whether the application of decision analysis is used in the approximation of a project’s completion or in the prediction of market trends or in assessing environmental cleanup consequences, there is strong evidence validating its usefulness. The aim of this thesis, as stated previously, is to demonstrate that decision analysis – with the aid of computer modeling - can indeed be applied to environmental dilemmas such as capping contaminated sediments in Boston Harbor. But before the bulk of this validation is presented in Chapter 4, it is necessary to first discuss if and how has DA been applied to environmental problems in general, and to investigate the importance of computers in such processes.

3.2.1 Environmental Applications of Decision Analysis

Environmental remedial and disposal efforts require large amounts of resources – both financial and human. Resources, however, are limited and thus it is impossible to fund every effort or allow multiple trials of the same remediation task should one fail. As such, environmental cleanup decision-making, at its base, involves the selection of a subset of alternatives from a larger set (Brooks, 1997).

In dealing with ecological problems, decision analysis can be utilized in several areas:

- the determination of environmental and health problems associated with a variety of activities and substances (for instance, hazardous-waste
disposal and the use of various chemicals) in terms of dredging and
disposal of contaminated sediments

- the comparison of new and existing technologies or the determination
  of the effectiveness of different control and mitigation techniques
designed to reduce risks
- the selection of disposal sites
- the establishment of management priorities, such as which of several
  activities should be considered first for regulatory or corrective action
- the assessment of the risks and benefits of particular substances to the
  environment so as to facilitate their regulation

There are perhaps two major issues in the application of decision analysis to
environmental considerations which require further attention: the value of life and the
definition of unreasonable risk. It seems clear that Congress felt there was no single,
objective definition of unreasonable risk; as stated in the House of Representative report,
the determination of unreasonable risk is a complex process of information gathering,
analysis, and judgments which involves (Cohrssen, 1989):

"...balancing the probability that harm will occur and the
magnitude and severity of that harm against the effect of
proposed regulatory action on the availability to society
of the benefits of the substance or mixture, taking into
account the availability of substitute for the substance or
mixture which do not require regulation, and other adverse
effects which such proposed action may have on society."

In essence, both concepts require the placement of some sort of value, whether
monetary or social, to phenomena that are intrinsically priceless and unquantifiable.
Indeed, it is precisely this assessment of value that is at the heart of heated environmental disputes and the point of contention between various interests groups. And it is these difficulties that has led to the current research in this thesis.

As such, the utilization of decision analysis in this thesis hopes to provide a systematic and mathematical process by which such value assignments and uncertainties can be quantified. Subsequently, computers provide an effective, efficient, and consistent methodology by which this quantification can be accomplished.
3.3 Computer Modeling and Decision Analysis

3.3.1 Computer Aids for Modeling Uncertainty

Most of the efforts for dealing with uncertainty require significant computer support. One of the primary reasons that decision analysis has seen relatively little use in policy analysis is the intimidating task of developing the necessary computer software. However, recently there has been a different trend with the advent of the personal computer age, increasing knowledge amongst policy-makers, and the ready availability of spreadsheet and decision-aiding software (Winston & Albright, 1997).

There are two fundamentally different approaches in the provision of software support for policy analysts. The more conventional entails the development of special purpose packages that perform one or another specific task. Most readers are familiar with packages such as spreadsheets, equation solvers, and statistical tool kits. At least one commercial spreadsheet package, “@Risk”, marketed by Palisade Corporation, supports substantial treatments of uncertainty through stochastic simulation. Software packages satisfying the special needs of policy analysis include commercially available tools that support the preparation, averaging out, and folding back of discrete decision trees. Other examples include “Arboist,” marketed by Texas Instruments; “Supertree,” marketed by the Strategic Decision Group; “Decision – 1 2 Tree,” marketed by Riskcalc Associated; and “Treeplan,” developed by Michael Middleton (Decision Analysis Systems website) Section 4.2 discusses Treeplan in greater detail.
An alternative approach to providing software help is the development of a general computer environment to support a variety of needs in quantitative policy modeling, including the treatment of uncertainty.

The focus of this thesis research follows the first approach. The goal has been to develop a methodology, combining the spreadsheet capabilities of Microsoft Excel and the graphical ability of Treeplan, that addresses the amount of sand capping needed in Boston Harbor. The discussion of how this model is created, the implementation and usage of the software package, the integration of the value hierarchy discussed in Chapter 2, and the results are presented fully in Chapter 4.

3.3.2 Decision-aiding Software

The essence of decision-aiding software is that it consists of various forms of microcomputer programming designed to enable users to: "(1) process a set of goals to be achieved; (2) determine alternatives available for achieving them; and (3) establish the relations between goals and alternatives in order to choose the best alternative, combination, allocation, or predictive decision-rule" (Clemen, 1952). Decision-aiding software should be distinguished from at least two other kinds of software that are relevant to making decisions but do not process goals, alternatives, and relations in order to arrive at predictive conclusions. One similar type of software is information retrieval software. It can be very useful in ascertaining the amount of money spent on a particular item in a certain year, the court cases that are relevant to a certain subject, or any kind of information that might be found in a statistical almanac or an encyclopedia. The second
related type of software is known as "office-wares", which can be useful for word processing, filing and retrieving data, or performing financial accounting.

The development of decision-aiding software can take a variety of forms. Some of the most common types that might be applied to Boston Harbor are:

- Decision tree software for making decisions under conditions of risk, such as whether to go on strike or accept a management offer. A decision tree is usually pictured as looking like a tree on its side with branches and sub-branches. The branches generally represent alternative possibilities that depend on the occurrence or nonoccurrence of probabilistic events.

- Linear programming software for allocating money, time, people, or other scarce resources to activities, places, tasks, or other objects to which the resources are to be allocated. In terms of form rather than function, linear programming involves maximizing or minimizing an objective function or algebraic equation subject to constraints generally in the form of inequalities like greater than or less than.

- Multi-criteria decision-making (MCDM) software, which emphasizes multiple goals to be achieved, as contrasted to decision trees, linear programming, and statistical regression analysis, which emphasize a single objective function or a single-dependent variable.

Undoubtedly, there are numerous other software programs which can also be considered decision-aiding: artificial intelligence (AI) and special weighting, among other. However, their discussion is beyond the scope of this thesis.
There are various obstacles to systematic decision making that decision-aiding software helps overcome. These obstacles include:

- Multiple dimensions on multiple goals.
- Multiple missing information. In the simplest form, this problem involves knowing the benefits and costs for a number of alternatives with the exception of one benefit or one cost.
- Multiple alternatives that are too many to analyze individually.
- Multiple and possibly conflicting constraints.
- The need for simplicity in drawing and presenting conclusions in spite of all that multiplicity. This is where spreadsheet-based software can be especially helpful.

In evaluating whether a particular decision-aiding software is effective requires the engendering of a set of criteria. These criteria can be classified as: "attitudinal effects, general and indirect effects, user/system interaction, flexibility, and economy/efficacy". For an in depth discussion of what each of these categories represent please refer to Morgan & Henrion, 1992.
3.4 Spreadsheet Modeling and Decision Analysis

Until recently, simulation modeling has been reserved for intensive research centers and “think-tanks” – organizations with access to large amounts of resources, large-scale computers, and personnel. With the advent of spreadsheets, however, modeling is now available to anyone with a personal computer, including public officials and policy-makers. The advantages of using spreadsheets to implement decision analysis models are:

- Simplicity of programming level: no advanced knowledge in programming is required;
- Easy modification of worksheets;
- User-friendly; spreadsheets allow users who are not proficient in computer to understand and implement models; and
- Flexibility; spreadsheets have built-in functions for statistical analysis, financial modeling, graphical representation, sensitivity analysis, and data management.

For the BHNIP, these models are fundamentally based on mathematics that ensures quantitative representation. The purpose of a mathematical model is to represent the essence of a real-life problem in a concise form. This thesis has sought to create a rather simple decision support system which is really an “interactive computer-based systems that help decision-makers utilize data and models to solve unstructured problems”. Indeed, ‘interactive’ and ‘computer-based’ are essential features. Decision-makers potentially no longer communicate with models through an analyst but have direct access to available models and information. The methodology developed in this thesis: (1) defines the problem and formulates a model; (2) presents the model; (3) evaluates the
model and its alternatives; (4) performs sensitivity analysis; and (5) generates or reports results. (see Chapter 4)

3.4.1 Spreadsheet Modeling with Microsoft Excel

The ease of use of Microsoft Excel and the advent of Microsoft Windows operating system, justified its being the spreadsheet environment chosen for this thesis. The ability to attach add-in tools (such as Treeplan) and the graphical capabilities of Excel make it an excellent choice in representing the application of decision analysis to the Boston Harbor capping methodology. A number of Excel’s built-in functions are useful for working with problems involving uncertainty. Probability is the basic mathematics of uncertainty. Whenever there is something we do not know, our uncertainty can (in principle) be described by probabilities. Several of Excel’s functions deal with specific probability distributions (J.L. Kellogg Graduate School of Management website):

- \( =\text{BINOMDIST}(\text{number}_s, \text{trials}, \text{probability}_s, \text{cumulative}) \)
- \( =\text{NORMDIST}(x, \text{mean}, \text{standard}_\text{dev}, \text{cumulative}) \)
- \( =\text{NORMINV}(\text{probability}, \text{mean}, \text{standard}_\text{dev}) \)
- \( =\text{NORMSDIST}(z) \)
- \( =\text{NORMSINV}(\text{probability}) \)
- \( =\text{EXPONDIST}(x, \lambda, \text{cumulative}) \)
- \( =\text{POISSON}(x, \text{mean}, \text{cumulative}) \)

For a primer on how to use Microsoft Excel and an introduction to its capabilities please refer to Using Excel for Windows 95. Please see (Morgan & Henrion, 1992) for a review of the probability theory behind the above functions, as well as a discussion of the simulation of uncertainty in spreadsheets.
3.4.2 Treeplan

Treeplan is a Microsoft Excel add-in developed by Michael Middleton in 1992. It is designed to graphically represent and generate decision trees. Microsoft Excel provides a means for both creating and solving complicated mathematical models of real-world situations. However, decision trees present difficulties for Excel. One must utilize Excel’s spreadsheet capabilities and also its graphical capabilities to depict the decision tree. This procedure is extremely complex. Fortunately, Treeplan if manipulated and implemented correctly, allows one to create and solve decision trees in Microsoft Excel. Although the process can be somewhat boring and slow, once implemented, sensitivity analysis can be readily performed. That is, once the decision tree is developed and all of the numerical inputs (monetary values and probabilities) are entered by the user, it is simple to vary inputs and observe how EMVs and the optimal decision change. A brief user manual on how to link Microsoft Excel with Treeplan, create decision trees, perform sensitivity analysis, and report results is included in the Appendix B.
4.1 Economic Risk Management and Utility Discussion

Scarce resource allocation is a critical issue in public policy analysis. Resources such as natural resources, the fruits of human labor, must be rationed and distributed accordingly (McConnell, 1992). The choice among alternative plans for the management of contaminated sediments takes into account trade-offs among the uses of society's scarce resources. However, with respect to environmental costs, harm is caused by the presence or re-suspension of contaminants, which most likely will affect the marine food chain adversely and thereby cause deleterious human health effects. As such, the challenge for policy makers is to attempt to quantify this broad range of environmental costs and combine it with dredging costs in making management decisions regarding sediment disposal (McConnell, 1992).

Economic risk management affect both direct costs and risk of environmental impacts. The product of the likelihood of a particular environmental risk and the cost should that risk occur gives rise to the expected cost of a particular remediation alternative (Welsh & Wilson, 1996). The lack of certainty in the occurrence, degree of, and cost of
particular events may be vital to erecting the optimum strategy. Different types of costs - such as capital costs and operating costs - may be treated differently to specific regulations/rules (Welsh & Wilson, 1996). Erroneous insights or wrong conclusions can occur if the decision analysis represent the various choices too simplistically. Indeed, uncertainty about the risks involved must be taken into account. Within the decision tree, the number of branches chosen and the estimated and assigned probabilities for the branches are a matter of judgement and depend on how much information about the uncertainty is available and meaningful. Once the uncertainty and decision has been represented, a “roll-back” method is employed to compute the expected values for all of the nodes. Finally, the optimal strategy is determined based on the minimal value from these calculations. The actual decision tree and results for Boston Harbor will be discussed later in sections 4.2-4.4.

But first, the following sub-sections of the thesis will discuss the sediment disposal and environmental costs, which are subsequently implemented into the applied decision analysis model to yield, appropriate expected value calculations. In essence, the determination of these costs help in assessing which capping level should be employed for the Boston Harbor scenario. It has been determined that the amount of capping material used is the single greatest contributor to the overall project cost. These quantifications are undoubtedly imprecise and constitute, at times, intuitive estimations and first-pass approximations. Note that present net worth of these costs was not predicted because these costs were inexact to begin with. Adding the extra layer of discount rate and net present value calculations are unnecessary given that the major thrust of this thesis is to
demonstrate the usefulness of decision analysis, rather than to perform an in-depth cost analysis of the capping methodology. This will be left to future work once the decision analysis methodology has been perfected.

4.1.1 Sediment Clean-up Costs

In the context of the management of contaminated sediments, most discussions of costs begin with disposal costs. Although there is a range within the order of magnitude of these costs, few ambiguous or controversial issues are involved in calculating them. These costs can be calculated based on market prices, and they involve tangible goods or services, such as wages and salaries, purchases of raw materials, rentals of equipment, and purchases of land. The calculation of these costs—namely, monitoring, bucket, disposal, long-term monitoring, and contingency costs—is an application of cost accounting and presumably were used by USACE in their determinations.

Based on the Commencement Bay Feasibility Study (U.S. Department of Commerce, 1988), the estimated sediment core costs—including boat and crew time—are within the magnitude of $1500 per core for 10 to 50 cores in a sampling event. These are within the range for similar analyses in New England. Chemical analysis costs—which includes grain size, total organic carbon, selected metals, PAHs, and PCBs—will vary depending on the contaminants in question but in general an extra 25% is added for quality control purposes.
The cost of modifying a clamshell bucket to make it watertight was estimated at $20,000, a value that needs to be evaluated on a case by case scenario. The estimated operating expenses for a clamshell dredge with a capacity of 200 yd$^3$/h are $1.25$/yd$^3$. Barge transport costs for hauling sediment up to 5 miles were estimated at $0.50$/yd$^3$ (U.S. Department of Commerce, 1988). Although the bucket used in BHNIP was previously modified, this cost was not included since its value was small when compared to overall costs, and since it is difficult to ascertain.

For open-water confined aquatic disposal, major costs include construction of the disposal well and the placement of capping materials. A disposal charge of $0.12$/yd$^3$($0.20$/ft$^2$) was approximated, based on the experience from other open-water confined aquatic disposal projects. A 15ft fill depth and a 3ft cap were also assumed (U.S. Department of Commerce, 1988).

Disposal site operation and management costs consist primarily of inspection, resuspension control, repairs, and other unanticipated costs are estimated at $3000/ac/yr based on data provided by U.S. EPA (1985). It is suggested that a monitoring program involving physical, chemical, and biological sampling would be conducted at each disposal site during years 1, 2, 3, 5, 7, and 10 of operation (U.S. Department of Commerce, 1988). The value of monitoring may be decreasing progressively so that the costs for no action monitoring may be higher in later years than for more conservative approaches of capping. This is beyond the scope of thesis however, and is reserved for future research.
Note that on all cost items, a 20% contingency is applied. A 10% markup for mobilization, bonding, and insurance is added to engineering costs plus contingency. An additional 15% is applied to that for project administration, engineering design, and fee (U.S. Department of Commerce, 1988).

Figure 5 gives a sample costing scheme for capping at Commencement Bay. A similar scheme is applied in this thesis.

4.1.2 Environmental Costs

The growth of the environmental movement, as well as increased understanding of the ecological and environmental effects of the dredging and disposing of sediments, has changed the cost-benefit calculation by introducing a third component, known generally as environmental costs. These costs represent injury or the threat of injury to a resource or to users of a resource. Natural resources are defined by two kinds of economic value: use value and non-use value (Brooks, 1995 and McConnell, 1992). Use value is simply the economic value provided by the opportunity to use resources for recreation, commercial fishing, and other direct uses. Nonuse value is the economic value of goods and services, which must be sacrificed in order to preserve the resource in its current state. A study of the economic losses attributed to contaminated sediments focused on the presence of
Figure 5 – Costing Scheme for Capping of Contaminated Sediments in Commencement Bay

<table>
<thead>
<tr>
<th>INITIAL COSTS</th>
<th>Unit Costs ($)</th>
<th>Costs ($)</th>
<th>Subtotal ($)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sediment Core (one per 4,000 yd³)</td>
<td>$1.500/core</td>
<td>$288,500</td>
<td>$354,000</td>
</tr>
<tr>
<td>Chemical Analysis (one per 4,000 yd³)</td>
<td>water-dependent</td>
<td>$47,200</td>
<td>$56,640</td>
</tr>
<tr>
<td>In-situ Capping</td>
<td>Dredge Operating Cost (6 ft cap)</td>
<td>$3.00 /yd²</td>
<td>$1,800,000</td>
</tr>
<tr>
<td></td>
<td>Barge Transport (up to 5 miles)</td>
<td>$1.00 /yd²</td>
<td>$510,000</td>
</tr>
<tr>
<td>Subtotal</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Contingency (20%)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Mobilization, Bonding, Insurance (10%)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Subtotal</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Administration, Engineering (15%)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Total Initial Costs</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

C&O COSTS - Present worth

| Chemical analysis | | | |
| Number of monitoring stations | 1 station/2 acre (20 maxima) | 12 | 12 | 12 |
| Core Acquisition | 1 core/station | $1,500/core | $18,000 | $21,600 |
| Chemical Analysis | 3 samples/station | $200/sample | $33,000 | $39,600 |
| * semi-annually for year 1 to 5 | | | |
| | $1,200/sample at Wheel & 0 | | |
| Every two years for years 6 to 30 | $2,300/sample at Ruston-Pt. Defiance | | |
| Biological analysis | 1 station/2 acre (20 maxima) | 12 | 12 | 12 |
| | 5 repic./station | $150/station | $2,250 | $2,700 |
| | Sentic Analysis | $750/sample | $45,000 | $54,000 |
| | Epibenthic Sampling | 5 replic./station | $50/station | $2,500 | $3,000 |
| | Epibenthic Analysis | 15 replic./station | $24,550 | $29,460 |
| Contingency (20%) | | | | $219,637 |

COST SUMMARY

| Present Worth of C&O Cost (10% Discount, 30 yr) | $1,517,000 | $1,282,000 |
| Total Alternative Costs | $2,229,000 | $1,954,000 |

Source: U.S. Department of Commerce, 1988
PCBs in the coastal waters off Los Angeles. These chemicals had various deleterious biological effects including: bioaccumulation in marine organisms, mortality of organisms, and threats posed to human health. Researchers estimated the value of the economic losses per household in California to be about $55 or for a total of $575 million if multiplied by the population of the state (McConnell, 1992).

The intuitive meaning of the damage calculation can be interpreted in the general context of resource allocation, as well as encompassing health and ecological costs. Health costs result from increased morbidity and mortality resulting from exposure to contaminated sediments. Ecological costs include the economic costs of damage to the ecological functioning of a natural resource (McConnell, 1992).

Although environmental costs are extremely difficult to quantify, for the purposes of this thesis, environmental costs will be dealt with mostly in terms of costs accrued due to inadequacy of the capping level. In other words, given that this research initiative focuses on applying decision analysis to determine the optimum capping thickness, environmental costs can be represented in two manners:

1) by costs incurred from switching to another capping level (this cost included remobilization and opportunity costs – see Section 4.2 below); and

2) by the costs approximated via successful or unsuccessful protection of marine biota from bioaccumulation (i.e. use and nonuse of these organisms as natural resources) and the successful or unsuccessful maintaining of water column concentrations for the specific contaminants (i.e. PAHs and PCBs). For the purposes of this study, this cost
is assumed to be plus or minus $10 million for successful and unsuccessful protections of the environment, respectively. Further research is needed to create a model whereby these costs are dealt with more systematically such as the BRAT analysis or applications from oil spill assessments.
4.2 Implementing Decision Analysis Models in Treeplan

Treeplan, as stated previously, is an add-in for Microsoft Excel, which enables the combination of graphical representations of decision trees with specific expected value calculations. In Treeplan, decision trees comprise of three kinds of nodes and two kinds of branches. A decision node is a point where a choice must be made; it is represented by a square. The branches extending from a decision node are decision branches, each branch representing one of the possible alternatives or courses of action available at that point. The set of alternatives must be mutually exclusive (if one is chosen, the other can’t be chosen) and collectively exhaustive (all possible alternatives must be included in the set).

An event node is a point where uncertainty is resolved (a point where the decision-maker learns about the occurrence of an event). An event node, sometimes called a "chance node", is shown as a circle. The event set consists of the event branches extending from an event node, each branch representing one of the possible events that may occur at that point. The set of events must be mutually exclusive (if one occurs, the others cannot occur) and collectively exhaustive (all possible events must be included in the set). Each event is assigned a subjective probability; the sum of probabilities for the events in a set must equal one.

In general, decision nodes and branches represent the controllable factors in a decision problem; event nodes and branches represent uncontrollable factors.
The third kind of node is a terminal node, depicting the final result of a combination of decisions and events. Terminal nodes are the endpoints of a decision tree, shown as the end of a branch on hand-drawn diagrams and a triangle or vertical bar on computer-generated diagrams. See Diagram 6 for a summary of the nodes and branches employed in Treeplan. Note that a User Manual on how to use Treeplan is included in Appendix B.

**Figure 6 – Decision Nodes and Branches in Treeplan**

Each terminal node has an associated terminal value sometimes referred to as a payoff value, outcome value, or endpoint value. Each terminal value measures the result of a scenario; the sequence of decisions and events on a unique path leading from the initial decision node to a specific terminal node. To determine the terminal value, one approach assigns a cash flow to each decision branch and event branch and then sum the cash flow values on the branches leading to a terminal node to determine the terminal value.
Within the decision analysis framework, different strategies are implemented into Treeplan. A strategy specifies an initial choice and any subsequent choices to be made by the decision-maker. The subsequent choices usually depend on events. The specification of a strategy must be comprehensive; if the decision-maker gives the strategy to a colleague, the colleague must know exactly which choice to make at each decision node. Most decision problems have multiple strategies. In the Boston Harbor scenario there are 5 strategies, each consistent with a different capping thickness employed. Each strategy has an associated payoff distribution, sometimes called a risk profile. The payoff distribution can be shown as a list of possible payoff values, x, and the discrete probability of obtaining each value, P(X=x), where X represents the uncertain terminal value. Since a strategy delineates a choice at each decision node, the uncertainty about terminal values depends only on the occurrence of events. The probability of obtaining specific terminal value equals the product of the probabilities on the event branches on the path leading to the terminal node. Since each strategy can be characterized completely by its payoff distribution, selecting the best strategy becomes a problem of choosing the best payoff distribution.

A certainty equivalent is a certain payoff value, which is equivalent, for the decision-maker, to a particular payoff distribution. If the decision-maker can determine his or her certainty equivalent for the payoff distribution of each strategy, then the optimal strategy is the one with the highest certainty equivalent. In other words, the certainty equivalent is the minimum selling price for a payoff distribution; it depends on the
decision-maker's personal attitude towards risk: risk preferring, risk neutral, or risk avoiding. In this thesis it is assumed that the decision-maker (i.e. user) is risk neutral, and as such the expected monetary value (EMV) is the appropriate certainty equivalent for choosing among the strategies. Note that the EMV of a payoff distribution is calculated by multiplying each terminal value by its probability and summing the products. The EMV is simply a weighted average of the possible monetary values, weighted by their probabilities. Formally, if $v_i$ is the monetary value corresponding to outcome $i$, and $p_i$ is its probability, then EMV (Morgan & Henrion, 1992) is defined as:

$$EMV = \sum v_i p_i \quad (Formula \ 3)$$

Thus, for a risk neutral person, the optimal strategy is the one with the highest expected monetary value (EMV).

It is not necessary to examine every possible strategy explicitly. Instead, a popular method known as "rollback" is used to determine the single best strategy. This method is the one utilized in this research initiative. The rollback algorithm, sometimes called backward induction or "average out and fold back", starts at the terminal nodes of the tree and works backward to the initial decision node, determining the certainty equivalent rollback values (or also referred to as EMVs) for each node. Rollback values are determined as follows:

- At a terminal node, the rollback value equals the terminal value.
- At an event node, the rollback value for a risk neutral decision-maker is determined using EMV; the branch probability is multiplied by the successor rollback value (EMV), and the products are summed.
- At a decision node, the rollback value is set equal to the highest rollback value on the immediate successor nodes.

On the trees, rollback values are located to the left and below each decision, event, and terminal node.

Please refer to Fig. 8-11 and Table 4 for greater depiction of the assumptions utilized in this decision analysis methodology.
4.3 Capping of Contaminated Sediments in Boston Harbor Decision Tree: Inputs, Tree Setup, and Strategies

For the capping of contaminated sediments in Boston Harbor, two sets of contaminants were studied: PAHs (2-methylnaphthalene and benzo(a)pyrene) and PCBs. For each contaminant, a decision tree representing bioaccumulation and one representing water column concentration were created. Within each tree, there are initially five decisions, corresponding to the five different capping levels being investigated: namely, no cap (0m), 0.5m cap, 1m cap, 1.5m cap, and >1.5m cap. Note that bioaccumulation and water column concentration bring together the issues of disposal costs, environmental costs/injuries, and the establishment of an equilibrium between ecological well-being and project cost-effectiveness. By entering the appropriate probabilities and costs, performing the related statistical and expected value calculations, and implementing the tree in Treeplan, the optimal capping level can be determined via decision analysis.

4.3.1 Inputs

The inputs to the decision analysis model are the probabilities for making a particular decision and the associated costs. For the bioaccumulation trees, an additional form to allow the decision maker to generate a Theoretical Bioaccumulation Potential (TBP) is included (see Section 2.3). In Formula 1, \[ \text{TBP} = 4 \times \left( \frac{\text{Conc. of chemical in dredged material}}{\text{Total carbon content of dredged material}} \right) \times \text{Organism lipid content}. \] This TBP is calculated based on experimental data and case/site specific conditions and will be referenced with EPA standards within the decision tree (USEPA, 1989).
The probabilities for making a particular capping level choice, for the purposes of this research initiative, have been based on intuition and experience. In decision analysis, there is no explicit mechanism to allow for discretizing probabilities amongst various choices. In other words, the methodology assumes that the probability for making any one of the choices to be equal. So if there are 5 decision choices, the probability for the user to make any one of the choice is assumed to be 0.2. However, in real life these probabilities are likely to be unequal with respect to a decision maker and are specific to his needs, his experiences, and his intuitions. Indeed, before utilizing decision analysis, it is entirely appropriate for the user to have some first impressions and preferences for which particular capping thickness he feels will be sufficient. It is through decision analysis that one determines whether the decision maker’s initial intuition is consistent with the actual optimum capping level. To solve this dilemma, one reflects a decision maker’s preference for a particular capping level decision through the associated costs. The costs accompanying each choice are, in part, based on the disposal and environmental costs considerations discussed above in Section 4.1. Moreover, a “preference” factor will be added to the costs to represent an individual decision maker’s initial desire for a particular capping level. If, for example, the user’s probabilities for choosing the five capping levels are 0.1, 0.2, 0.4, 0.2, and 0.1 respectively, each of these probabilities will be multiplied by a constant economic parameter. This parameter will be a function of the opportunity cost of selected capping alternative/strategy. The product of the parameter and the respective probability will be added to the disposal and environmental cost associated with each decision choice.
4.3.2 Decision Tree Set-up

There are four major decision trees analyzed within this research initiative – one tree each corresponding to the *Bioaccumulation* and to the *Water Column Concentration* for PAHs and PCBs. For each tree there are five initial decisions, numbered 1 through 5, representing the five different capping levels to be investigated (see above). Each decision then confers two events. For the *Bioaccumulation* trees, these events correspond to the complete successful protection of marine biota (i.e. acceptable levels of bioaccumulation) or failure to protect marine biota. One may take one particular capping level decision as an example to illustrate and facilitate understanding of the construction of the entire decision tree. See Figure 7 for a sample capping level decision sub-tree. Note that the N/A denote not applicable and refer to values from the roll-back method, which will be discussed later.

**Figure 7 – Sample Decision Analysis Subtree**
In the event where marine biota is not protected (Event 1b), the analysis terminates and another capping level decision strategy must be utilized. However, in the event where marine biota is protected (Event 1a), the decision maker is faced with two additional decisions: to continue to use the current capping level strategy (i.e. maintaining the status quo), or to change to a lesser capping level strategy (Decision 1a and 1b, respectively). If the decision is to change to a lesser capping level, two further events will result: namely, one is able to switch to an alternative capping level (Event 1c) or one is unable to do so (Event 1d). The distinction between these two events are probably predicated upon additional costs to remobilize and the practicality of switching at that point in time.

The probability of whether a particular capping level choice successfully protects the marine biology most likely will depend on experimental data and expertise that the decision maker has accrued. Again, this is a variable, which the decision maker can manipulate in the decision tree. However, for the purposes of this thesis, the probability of unsuccessful and successful protection will be approximated as 0.5:0.5, subject to future sensitivity studies.

The above sub-tree for Decision 1 is repeated for each of the other capping level strategies. Once the costs are entered, the terminal values will be computed and the roll-back method will yield the optimum strategy (i.e. the optimum decision) which corresponds to the maximum utility or expected value. The term “utility” is utilized to describe a particular set of values that represents a decision maker's preferences among outcomes. According to modern normative utility theory, one should always make
decisions that maximize expected utility (DeNeufville, 1990). Decision models calculate the expected value of the outcome for each decision alternative (i.e. strategy) and identify the alternative with the greatest expected value.

Note that the only difference between the Bioaccumulation decision trees (Figures 8 & 9) and the Water Column Concentration decision trees (Figures 10 & 11) is that in the latter scenario, Events 1a and 1b from Figure 7 above would refer to the adherence of FDA water quality standard or the exceeding of FDA water quality standard, respectively.

4.3.3 Decision Trees for Bioaccumulation and Water Column Concentration of PAHs and PCBs

Based on the various experimental data, first-pass estimates, and the weighting criteria discussed previously, the following decision trees were created to help determine the optimal capping level for the disposal of dredged materials in Boston Harbor. These charts take into account environmental concerns, as well as cost-benefit issues. Figures 8 - 11 displays the decision trees for the Bioaccumulation and Water Column Concentration of PAHs and PCBs, respectively. These schematics were interfaced from Microsoft Excel and Treeplan.
Figure 8 – Conceptual Decision Tree of Bioaccumulation of PAHs in Marine Organisms in Boston Harbor

Sample Subtree Delineation

Decision 1: No Cap (No)
Event 1a: Complete successful protection of marine benthos
(i.e., acceptable levels of bioaccumulation of contaminants)
Event 1b: Failure to protect marine benthos
Decision 2a: Continue to use current chronic capping level
Decision 2b: Discontinue use of current chronic capping level
Event 1c: Able to switch to alternative chronic capping level
Event 1d: Unable to switch (projected water costs)

STRATEGY 1

STRATEGY 2

STRATEGY 3

STRATEGY 4

STRATEGY 5
Figure 9 – Conceptual Decision Tree of Bioaccumulation of PCBs in Marine Organisms in Boston Harbor

Sample Subtree Delineation

Decision 1: No Cap (no)
Event 1a: Complete successful protection of marine life (e.g., acceptable levels of bioaccumulation of PCBs)
Event 1b: Failure to prevent marine life
Decision 2a: Condition to use current chosen capping level
Event 2a: Able to switch to alternative capping level
Event 2b: Unable to switch (proposed action costs)

STRATEGY 1

Decision 2: -1.2E+07
-1.2E+07
Decision 3a: 0.475000
-0.1E+07
Decision 3b: 0.5
-0.75E+07
-1.75E+07
Decision 4a: 0.5
-0.1E+07
Decision 4b: 0.2
-0.2E+07
Decision 5a: 0.5
-0.2E+07
Decision 5b: 0.2
-0.2E+07

STRATEGY 2

Decision 3: 0.475000
-0.1E+07
Decision 3a: 0.475000
-0.1E+07
Decision 3b: 0.5
-0.75E+07
-1.75E+07
Decision 4a: 0.5
-0.1E+07
Decision 4b: 0.2
-0.2E+07
Decision 5a: 0.5
-0.2E+07
Decision 5b: 0.2
-0.2E+07

STRATEGY 3

Decision 3: 0.475000
-0.1E+07
Decision 3a: 0.475000
-0.1E+07
Decision 3b: 0.5
-0.75E+07
-1.75E+07
Decision 4a: 0.5
-0.1E+07
Decision 4b: 0.2
-0.2E+07
Decision 5a: 0.5
-0.2E+07
Decision 5b: 0.2
-0.2E+07

STRATEGY 4

Decision 3: 0.475000
-0.1E+07
Decision 3a: 0.475000
-0.1E+07
Decision 3b: 0.5
-0.75E+07
-1.75E+07
Decision 4a: 0.5
-0.1E+07
Decision 4b: 0.2
-0.2E+07
Decision 5a: 0.5
-0.2E+07
Decision 5b: 0.2
-0.2E+07

STRATEGY 5

Decision 3: 0.475000
-0.1E+07
Decision 3a: 0.475000
-0.1E+07
Decision 3b: 0.5
-0.75E+07
-1.75E+07
Decision 4a: 0.5
-0.1E+07
Decision 4b: 0.2
-0.2E+07
Decision 5a: 0.5
-0.2E+07
Decision 5b: 0.2
-0.2E+07

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Figure 10 – Conceptual Decision Tree of Water Column Concentration of PAHs in Boston Harbor

Sample Subtree Delineation

Decision 1
- No Cap (the)
  Event 1a
  Complete succcessful protection of marine lives
  (i.e., acceptable levels of PAH contamination in water columns)

Decision 1b
- Failure to protect marine lives
- Decision 1a continues at current chosen coping level
- Decision 1b discards use of current chosen coping level

Event 1d
- Able to switch to alternative coping level
- Event 1d
  - Unable to switch (projected water usage)

STRATEGY 1

STRATEGY 2

STRATEGY 3

STRATEGY 4

STRATEGY 5
Figure 11 – Conceptual Decision Tree of Water Column Concentration of PCBs in Boston Harbor

Sample Subtree Definition

Decision 1
No Cap (Low)
Event 1a
Successful protection of marine bios
Event 1b
Failure to protect marine bios
Decision 1a
Continue to use current chronic capping level
Event 1c
Able to switch to alternative capping level
Event 1d
Unable to switch (projected extra costs)

STRATEGY 1

Event 1a
6135000
0.1

Event 1b
-1.4E+07

Decision 1b
0.5

Event 1c
6350000

Event 1d
-1.4E+07

Decision 1c
0.5

Event 1e
6000000

Event 1f
-1.4E+07

Decision 1f
0.5

Event 2a
3230000

Event 2b
1.7E-07

Event 2c
-1.4E+07

Decision 2c
0.5

Event 2d
-1.7E+07

Decision 2d
0.5

Event 2e
1.4E+07

Event 3a
-0.2

Event 3b
-6.4E+06

Decision 3b
0.5

Event 3c
2.8E+06

Event 3d
-5.6E+06

Decision 3d
0.5

Event 3e
-2.5E+07

Event 3f
-1.2E+07

Decision 3f
0.5

Event 4a
-1.9E+07

Event 4b
-1.9E+07

Decision 4b
0.5

Event 4c
-9.2E+06

Event 4d
-1.2E+07

Decision 4d
0.5

Event 4e
-5.8E+06

Event 4f
-2.8E+07

Decision 4f
0.5

Event 5a
-1.5E+07

Event 5b
-1.5E+07

Decision 5b
0.5

Event 5c
-9.8E+06

Event 5d
-2.8E+07
4.3.4 Capping Level Strategies

As described in Section 4.2 above, each strategy in a decision analysis methodology is comprised of an initial choice from a set of decisions and any subsequent choices to be made by the decision maker. In essence, a strategy is really a particular path along the overall decision tree, which yields specific terminal expected values. In the current BHNIP context, there are five strategies (labeled Strategy 1 – 5 in the diagrams above) which corresponds to five different sub-trees. Let’s isolate Strategy 3 for the Bioaccumulation of PAHs in Marine Organisms (extracted from Diagram 8) to illustrate and discuss the numerical and conceptual ramifications which contributed to its construction. Strategy 3 was chosen because initially, it has been predicted that a 1.0 meter cap was probably most preferable (a probability of choosing this strategy was estimated at 0.4 or 40%; note that this probability is reflected in the initial costs associated with making this particular choice).

(i) **Strategy 3 – 1.0m Cap with respect to bioaccumulation of PAHs in marine organisms**

Strategy 3, which corresponds to a 1.0 m capping level, is the one most preferred by decision makers initially. This is consistent with the findings of Shull and Gallagher (1998) which estimated that in a “worse-case-scenario” a 25cm cap thickness would isolate dredged material from burrowing benthos (this is the bioturbation thickness discussed in Section 2.4.1). Note the initial prediction (before application of decision analysis) of a 1 meter cap was derived by doubling the Shull and Gallagher value to allow for protection of the marine biota and doubled once more to allow for dredging/disposal errors. Therefore, for purposes of the decision analysis model, and in tune with the formula proposed for Final Cap Thickness in Section 2.4, the minimal total capping level necessary is intuitively predicted to be 1.0 meter. When a decision maker selects an initial
choice between the five strategies, he is more likely to choose Decision/Strategy 3. (see Diagram 12 below) That is the rationale for assigning a probability of 0.4 to one choosing Strategy 3 in the beginning (before decision analysis is performed).

Figure 12 – Sub-tree of Strategy/Decision 3 (1.0m Capping Level)

The implication of the initial probability of choosing Strategy 3 is reflected in the cost parameter which is included in the costs associated with making such a decision. To elucidate this point, refer to Table 4 below for a breakdown of the initial costs which will be inputted by the user:
Table 4 Initial Decision Costs (taken from USACE, 1988)

<table>
<thead>
<tr>
<th>Cost Item</th>
<th>Quantity</th>
<th>Cost/Quantity</th>
<th>Total Item Cost</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sediment core cost</td>
<td>50 cores</td>
<td>$1500 /core</td>
<td>$75000 (fixed)</td>
</tr>
<tr>
<td>Dredging with bucket</td>
<td>1</td>
<td>$20000/</td>
<td>$20000 (fixed)</td>
</tr>
<tr>
<td>Bucket Operating Cost</td>
<td>50 yds x 50 yds x 50 yds</td>
<td>$1.25 /yd³</td>
<td>$160000</td>
</tr>
<tr>
<td>Barge Transport Cost</td>
<td>10 km</td>
<td>$1.00/ yd³</td>
<td>$110000</td>
</tr>
<tr>
<td>Disposal Cost</td>
<td>4.9 m</td>
<td>$0.20 /ft²</td>
<td>$1000000</td>
</tr>
<tr>
<td>Site Operation and Management Cost</td>
<td>-----------</td>
<td>$3000/ac/yr</td>
<td>$100000</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Subtotal: $1465000</td>
</tr>
<tr>
<td>Sand/clay capping material cost for every 0.5m of cap</td>
<td>2 quantities of 0.5 m</td>
<td>$3000000 per 0.5 m</td>
<td>$6000000</td>
</tr>
<tr>
<td>Opportunity cost assoc. with making choice</td>
<td></td>
<td>$10000000 * (1-Pi) where Pi is the probability of making a particular choice (0.4 for Strategy 3)</td>
<td>$600000</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>TOT: $8065000</td>
</tr>
</tbody>
</table>

Note the various cost values were referenced from the Commencement Bay project (U.S. Department of Commerce, 1988). The last cost parameter in the table – namely, the opportunity cost associated with making a particular initial choice of a strategy – is of particular importance. If Pi is the probability of the decision maker choosing this strategy (Strategy 3 in this case), then 1-Pi represents the probability that this strategy will not be chosen. Subsequently, it is intuitive that there must be some sort of associated
opportunity cost. In other words, if the user were to choose Strategy 3, he must incur some opportunity cost that would not have been added to the overall costs had he not made this particular choice. Note that the more likely a decision maker is to choose a strategy (i.e. the larger the $P_i$) then the smaller the associated opportunity cost. For Strategy 3, $P_i = 0.4$, therefore the opportunity is $1,000,000 \times (1-0.4) = $600,000. As such the total cost of selecting Strategy 3 is then $8,065,000. In the decision sub-tree displayed above (Figure 12) this value is entered directly below the label Decision 3. Note that it is negative to indicate a cost incurred. Positive numbers in the tree will represent value added.

Note that the overall cost for choosing Decision 3 was calculated to be approximately $8,065,000, which was based on data from Commencement since BHNIP has yet to officially publish their cost analysis. However, in the Boston Harbor context, this number is more likely to be in the order of magnitude of $60,000,000. The discrepancy can be attributed to the fact that the above table is assuming a dredging volume which is about seven times less than that will be actually dredged in Boston Harbor – namely, some 900,000 cubic yards. But this difference will not affect the output or the decision analysis, since all initial costs will only have to be multiplied by a factor to account for the greater dredging volume in real life. Due to the fact that the major objective of this thesis is to test the feasibility and to justify the usefulness of applying decision analysis, this numerical discrepancy is unimportant.
Once Decision 3 has been made, there are two possible events: Event 3a and 3b, corresponding to successful and unsuccessful protection of marine biota from bioaccumulation respectively. Above the labels Event 3a and 3b are the probabilities of these two events occurring (0.7 and 0.3, respectively). These values are also user inputs and depend on experimental data that computes the Theoretical Bioaccumulation Potential (TBP), user preference, and the number of options available. The TBP is then referenced with the U.S. EPA levels for acceptable bioaccumulation levels with respect to the particular site conditions and species in question. A statistical analysis will indicate whether the experimental data will more likely yield TBP values which exceed the U.S. EPA levels or fall below it. It has been determined for Strategy 3, that the calculated TBP values will be within acceptable EPA values 70% of the time. If the capping level fails to protect against bioaccumulation, then the decision analysis stops and a cost of -$10,000,000 is incurred, reflecting damage to the environment. However, conversely if the capping level successfully protects against bioaccumulation, there is added ecological value of +$10,000,000. These two numbers are directly below the Event 3a and Event 3b labels in the sub-tree diagram (Figure 12).

After determining that the capping level has been successful with respect to bioaccumulation protection, the decision maker is again faced with two choices: whether to continue to use the current capping level (Decision 3a) or whether to switch to a lower capping level (Decision 3b). At this juncture, the probability of which of these two decisions he selects is irrelevant. The only pertinent aspect is the opportunity cost associated with making such a choice. The system is setup to reflect an opportunity plus remobilization cost of -$1,050,000 if he decides to switch and zero cost if he doesn’t.
The advantages of switching is evidenced by lower costs of capping material used which can significantly affect the overall cost of the project. However, there is the associated opportunity cost and re-mobilization (grouping together crew, barge, etc.) cost. Remobilization might also be a logistical nightmare if the companies contracted to perform the dredging and disposal are unavailable should switching to a lower capping level be necessary. Also, if one conceptualizes this in a realistic setting, it is entirely unclear how one would go about physically reducing the capping level once the cap has been placed. In other words, how does one remove 0.5 meters of cap material from say a 1.5 meter cap? The costs associated with such an endeavor are hard to predict. But perhaps, that is further justification for utilizing decision analysis so that these types of questions are answered via the model and computer simulation. Sensitivity analysis and fine-tuning of the model will indicate the optimal capping level to be used, thereby almost eliminating the possibility of actually having to remobilize and reduce capping level.

Finally, once the decision maker decides to switch to a lower capping level, the decision tree reflects two possible events: that one is able to switch (Event 3c) or one is unable to (Event 3d). The probability of these two events occurring is estimated to be equal at this first-pass approximation. But as discussed above, in real-life this may very likely not be the case. If one is able to switch, there is an added value of $1,000,000 because this should offset the opportunity cost of switching. The rationale is that if one is indeed able to switch, then there is no real harm done, or no new additional cost incurred except for the remobilization cost.
Note that as displayed in the sub-tree above in Figure 12, the numerical values appearing to the right of the entered values are the rollback EMVs. These EMVs are calculated starting from the terminal values and working backwards across the tree from right to left. (see Formula 3) The final rollback value for Strategy 3 is -$4,065,000.
4.4 Summary of Outputs

The construction of the entire decision trees for PAHS and PCBs are identical to way in which Strategy 3 sub-tree was accomplished. The overall tree is comprised of all five strategies, each with a specified initial probability of choosing that strategy. The only distinction between PAHs and PCBs comes from the data and TBP calculation that indicate it’s more likely a 1.5m cap (Strategy 4) will prevent bioaccumulation of PCBs (see Diagram 9 in Section 4.3.3) With respect to the decision analysis model for maintaining acceptable levels of contaminants in the water column (Diagram 10 and Diagram 11), refer to Section 2.3.1 for a discussion of the acceptable FDA and U.S. EPA water quality values. The data indicates that only a concentration of 0.030 µg/L of PCBs is acceptable (U.S. EPA standard) in the water column to prevent acute toxicity of salt-water aquatic level, as opposed to 300 µg/L for PAHs in general. This difference bore the result that while a 1.0 meter was sufficient to comply with the U.S. EPA level for PAHs, a 1.5 meter cap was necessary with respect to PCBs. Because the model assumes the same advective transport of PAHs and PCBs into the water column, the difference in the capping levels calculated stems from more stringent U.S. EPA levels for the concentration of PCBs allowable in the water column. Also, the initial costs for the water column strategies stem from additional costs to perform specific water quality monitoring, quality control, and in-lab reference sediment tests to compare with the specified EPA standards. Please refer to the Appendix for the actual Microsoft Excel spreadsheet forms for exact calculations, formulas, and numerical analysis.
In Table 5 below, the decision analysis results for assessing the optimal capping level to be used in the disposal of contaminated sediments in Boston Harbor are summarized.

<table>
<thead>
<tr>
<th>Contaminant</th>
<th>Optimal Capping Level to Prevent Unacceptable Bioaccumulation Levels</th>
<th>Optimal Capping Level to Prevent Unacceptable Contaminant Concentrations in the Water Column</th>
</tr>
</thead>
<tbody>
<tr>
<td>PAHs</td>
<td>1.0 meter cap</td>
<td>1.0 meter cap</td>
</tr>
<tr>
<td>PCBs</td>
<td>1.5 meter cap</td>
<td>1.5 meter cap</td>
</tr>
</tbody>
</table>
4.5 Sensitivity Analysis

The value of the above decision analysis framework is that one can easily perform a sensitivity analysis on any or all of the input parameters of the problem. The methodology allows for the easy examination of the impact of reasonable changes in base-case assumptions. In order to determine how sensitive the optimal capping strategy is to the costs of making specific initial decisions, one can simply go to the spreadsheet and re-enter input values. If one had constructed a decision analysis model by pen and paper, sensitivity analysis would have been next to impossible. However, the facility of the spreadsheet is that by simply changing various user-controlled values, the tree will yield possibly different optimal outputs.

Although sensitivity analysis is a fundamental methodological concept, it is an intuitively simple idea: the determination of whether something one’s uncertain about with respect to the decision analysis model is important. Sensitivity analysis plays a critical role at two stages (Toll & Pavlou, 1992):

1) At the inception of the structural model development, there may be several alternative formulations of a particular relationship that are being considered for use in the model. A simple version of each can be tried out and the impact on the outcome observed. If the outcomes change significantly, more detailed modeling is suggested.

2) Once a first-pass model is complete, including specifications of a decision criterion, sensitivity analysis is carried out on each of the uncertain variables. The value of the
variable is varied over its maximum possible range and observations on its impact on the decision are recorded.

Sensitivity analysis is part of future research that must be done to validate the usefulness of the proposed decision analysis methodology to the management of contaminated sediments. Furthermore, it is likely that sensitivity analysis will be preformed as biological and chemical monitoring of the disposal site yield new data reflecting the efficiency and efficacy of the particular capping level chosen.
4.6 – Recommendation of Optimal Capping Level Based on DA Results

From the results of the decision analysis for PAHs and PCBs, taking into account both bioaccumulation and water column concentration criteria, it is apparent that a 1.5 meter cap is recommended as the overall optimal capping level which should be utilized in the Boston Harbor scenario. Of course, sensitivity analysis may be performed to establish the validity of these results, and perhaps to reconcile an intermediate capping level between 1.0 – 1.5 meter cap.
CHAPTER 5 – CONCLUSIONS/
FUTURE WORK

Decision analysis of the optimum capping level to be used in the disposal of contaminated sediments in Boston Harbor was determined to be 1.5 meters, although only 1m has been required by regulatory agencies including U.S. EPA. This value reflects an analysis of the bioaccumulation of PAHs and PCBs in marine organisms, as well as the allowable water column concentrations of these compounds.

The utilization of decision analysis in this thesis is just a first step, base-case attempt at implementing such a systems analysis approach to the disposal of contaminated sediments, particularly in the Boston Harbor context. The major thrust of this research was to demonstrate the usefulness of decision analysis. Sensitivity analysis must be performed to validate some of the assumptions made in regards to probabilities, costs, and other parameters. Moreover, in the current framework, detrimental environmental costs were assumed to be $10,000,000: in essence, the failure of the decision analysis model to protect the environment. However, the derivation of a precise calculus of such environmental costs must be made. Indeed, the economic evaluation of damages to the marine biota must be systematically justified.

Decision analysis is an invaluable tool to policy-makers trying to tackle complex environmental dilemmas. Not only does it allow for continuous and dynamic interactions
between the parties, it also gives opportunity to refine the decision repeatedly. Such a scheme lessons the possibility of error and failure when applied to real life, and reduces overall project costs and efficiency.
REFERENCES


J.L. Kellogg Graduate School of Management. Northwestern University.
http://www.kellogg.nwu.edu


APPENDIX A

A Discussion of Polycyclic Aromatic Hydrocarbons
Appendix A

1.0 Introduction

During colonial times, Boston Harbor was a mecca for trade, shellfishing, and shipping. Pollution of the harbor has been ongoing for the past three and a half centuries - from butchers dumping their offal, to the introduction of solid sewage wastes (Stolzenbach & Adams, 1998). Over the last two decades, in the context of increasing environmental awareness, serious efforts -- such as the Boston Harbor Project -- have been initiated to remediate the harbor. More recently, the Boston Harbor Navigation Improvement Project has been began, which entails dredging the inner harbor. The desire is to deepen the channels of the harbor so that sea-faring ships can have easier access, thereby expanding Boston’s importance as a port city. One of the major ecological issues associated with such a project is how the displacement, removal, transport, and disposal of contaminated dredged sediments will affect marine biota and human health. It has been suggested that a capping methodology be utilized to dispose of the dredged material. As such, sediments with polycyclic aromatic hydrocarbons (PAHs) must be addressed. Invariably, the need to better understand the fate and transport processes of polycyclic aromatic hydrocarbons is critical to the success of the project (Brocard, 1994). This is a major motivation for the writing of this paper.

The potential damage to marine biology, coupled with threats to human welfare (mainly due to PAH carcinogenesis), make polycyclic aromatic hydrocarbons an important chemical species to be investigated. Sixteen unsubstituted PAHs have been identified by the EPA as priority pollutants. As PAHs are introduced into the environment via natural and anthropogenic combustion processes, their source, fate, and transport processes become critical issues to be addressed (Adams, 1997).

The aim of this paper is two-fold: (1) to outline the source and chemical characteristics of PAHs; and (2) to discuss the ramifications of PAH transport and fate in Boston Harbor.

2.0 What are PAHs

Polycyclic aromatic hydrocarbons (PAHs) are also designated by the term polynuclear aromatic hydrocarbons. PAHs are relatively inert (having little reactivity), lipophilic, hydrophobic compounds. In order to understand the chemical characteristics of PAHs it is first worthwhile to briefly discuss hydrocarbons in general.

2.1 Hydrocarbons

Hydrocarbons are chemicals or compounds which consist of carbon and hydrogen and range in simplicity from methane to very complicated ring structures. Methane is the simplest of hydrocarbons and consists of one carbon atom surrounded by four hydrogen atoms which are bonded to the carbon with a simple sharing of electrons or saturated bond. Detailed chemical bonding are beyond the scope of this report. However, it is important to note that bonding is significant in terms of the properties of the different classes of hydrocarbons: i.e. n-alkanes, branched alkanes or isoalkanes, cyclic alkanes or cycloalkanes, etc.
2.2 Aromatic Hydrocarbons

The "aromatic" name comes from the "aromas" of some of the compounds in this class of chemicals. The simplest example of aromatic hydrocarbons is benzene. In this case the carbon atoms are all bonded to one another with bonds different than those in cyclic alkanes and each carbon has only one hydrogen atom bonded to it. The extra electrons not bonded to the second hydrogen atom per carbon molecule are shared between the carbon atoms in a pi or \( \pi \) bond. The electrons are actually in a "cloud" above and below the ring of carbon atoms. The manner in which this cloud is positioned in the multiple condensed rings such as pyrene and benzopyrene is critical to the physical and chemical properties of PAHs -- subsequently, governing the environmental and biological fate.

The motivation for the above discussion on the chemical structure of PAHs is as follows. Over numerous decades, an important lesson has been learned during the course of researching PAHs:

While general rules for environmental behavior and biological effects an be set forth, it is vital to understand that exact chemical structures govern the critical details.

Two examples pertaining to polycyclic aromatic hydrocarbons illustrate this observation (Mass. Bay Marine Studies Consortium, 1996):

1) Examine the structures of benzo(a)pyrene and benzo(e)pyrene and one notices the same number of carbon and hydrogen atoms -- leading to the conclusion that they are isomers. However, while benzo(a) pyrene is a potent protocarcinogen, benzo(e)pyrene is not. The difference in their functionality is due to the way in which the five carbon rings are arranged and connected together.

2) Examining the three ring structures of phenanthrene and anthracene, and one observes that they too are isomers. Phenanthrene is soluble to the extent of approximately 2 - 6 micromoles per liter of seawater depending on the temperature and salinity. In comparison, anthracene is soluble to the extent of approximately 0.05 - 0.25 micromoles per liter of seawater. Indeed, anthracene is 20 to 40 times less soluble.

3.0 Sources of PAHs in the Environment

In the environment there are five major sources of polycyclic aromatic hydrocarbons (Mass. Bay Marine Studies Consortium, 1996):

1) Petroleum (crude oil) and petroleum products - PAHs can enter the environment in this category via:

   a) spills
   b) chronic inputs from human activities, and
c) natural seeps and erosion of sediments which contribute material to rivers and coastal areas and having almost mature petroleum dispersed in the ancient sediment being eroded.

2) **Fossil fuel combustion products (FFCP) including:**

   a) incomplete combustion of coal, oil, gas - e.g. soot emanating from incinerators; PAH particles from automobile and diesel exhausts; and PAH in used crankcase oil.

   b) creosote

3) **Industrial processes:** e.g. wasting of carbon electrodes in certain electrochemical industrial processes - e.g. some aluminum production.

4) **Forest and grass fires**

5) **Early diagenesis of organic matter in surface muds.** For instance, some of the early transformation products of naturally biosynthesized organic molecules such as abietic acid from pine trees and steroids and hopanoids from a variety of plants, animals, and bacteria are turned into aromatic or partially aromaticized hydrocarbons by microbial and mineral catalyzed reactions. In many cases the exact reaction pathways are not yet known, but the presence of the aromatic compounds in the sediments has been clearly identified and shown not to originate from oil pollution, sediment erosion, or combustion sources.

3.1 **Distinguishing Between Sources of PAHs**

Combustion product sourced PAH and those with petroleum or fossil fuels as a source can be distinguished from one another. Petroleum (and coal) contains a large abundance of alkyl substituted aromatic hydrocarbons relative to the unsubstituted parent compound. For example, phenanthrene (and many other PAHs) in crude oil is characterized by a series of parent compound with no substituted alkyl groups on the aromatic ring carbons and also by a series of methyl-, dimethyl-, ethyl-, trimethyl-, tetramethyl-, ethyl dimethyl-, etc. phenanthrenes. If the relative abundance of alkyl substituted phenanthrene is graphed relative to the number of alkyl substituents, a plot similar can be generated. The distribution is indicative of the fact that the slow formation process for the PAH in crude oil in the ancient sediments allows for more extensive scrambling of the alkyl substituents.

Combustion product PAHs, on the other hand, have predominantly the parent ring structure for several PAHs and relatively few PAHs with alkyl substituents. In general, the hotter the combustion process and given sufficient oxygen, only the parent PAH is present in any appreciable amounts. As the combustion temperature is lowered, there are increasing amounts of the alkyl substituents relative to the parent compound, but the parent compound still predominates.

It is often difficult to specify the source of PAH further than identifying whether the PAHs in a given sample emanate from a combustion source of some sort, from a petroleum or coal...
source, or from some combination of the two. In some rare instances, it may be possible to analyze very specific mixtures and ratios of PAHs and determine the exact source of the PAH.

4.0 Biochemical Cycles and Ecological Distributions of PAHs in Aquatic Environments

In numerous respects, a biogeochemical cycle can be characterized as the routes of movement into and through the ecosystem, the reactions acting on the chemicals in question (in this case, PAHs), the rates of both the movement and the reactions, and the reservoirs where chemicals (i.e. PAHs) will temporarily reside for some periods of time. See Fig. 4 and Fig. 5 for a representation of such a cycle for PAHs into aquatic environments (i.e. oceans, rivers). One of the ongoing challenges in understanding the fate and transport of PAHs is to erect more rigorous quantification. In Boston Harbor, efforts are continuing in attempting to provide site-specific quantitative models for PAH transport (please see Section 5.0).

There are several important characteristics implicit in the biogeochemical cycle of PAHs:

(1) Since PAHs are hydrophobic, they can dissolve in small amounts in seawater. The PAHs, do however, have an affinity to be attached to particles in the water - both living and dead (detritus). The important ramification is that many of the PAHs that enter the marine environment, end up in the surface sediments;

(2) The hydrophobicity of PAH compounds leads to them being transported by marine organisms via gill and other surfaces and being partitioned into the organism. If an organism has been in an area of high concentrations of PAH and moves to an area of low concentrations, then the PAH can be released or re-partitioned back into the water;

(3) PAHs can be ingested by marine biota;

(4) PAHs can be metabolized by a variety of marine organisms. In essence, the chemical structure of PAHs can be changed as a result of enzymatic metabolism (cytochrome P450 enzymes) in animals and bacteria. The structural changes affect a) PAH reactivity that can result in carcinogenesis, and b) make PAHs more hydrophilic which influences the rate at which the compounds are eliminated from the body. And,

(5) Under the right conditions, PAHs can be transformed by bacteria and remineralized to carbon dioxide and water (Alber & Chan, 1991). This leads to rapid accumulation in surface muds and transport to bottom sediments.

5.0 Fate and Transport of PAHs in Boston Harbor

5.1 Sources of PAH Contamination in Boston Harbor

Boston harbor receives a variety of hydrocarbon inputs including polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), pesticides, and petroleum hydrocarbons...
(oil and grease). Of these, reliable estimates of total inputs are available only for PAH (Menzie et al., 1991; Alber and Chan, 1991). The total PAH input is about 20,000 kg/yr, almost all of which is attributable to the sewage effluent and sludge flows. However, it has been shown that the identity of the major source may be different for individual PAH compounds. Studies have demonstrated that inputs of pyrene (197 kg/yr mostly from tributaries, effluent, and sludge), benzo(a)pyrene (22 kg/yr mostly from tributaries, effluent, stormwater, and the atmosphere), and 2-methylnaphthaene (1785 kg/yr almost all from sewage effluent), see Table 1.

<table>
<thead>
<tr>
<th>Source of Input</th>
<th>Pyrene (kg/yr)</th>
<th>Benzo(a)pyrene (kg/yr)</th>
<th>2-Methylnaph. (kg/yr)</th>
<th>Total PAH kg/yr</th>
</tr>
</thead>
<tbody>
<tr>
<td>Effluent</td>
<td>58.7</td>
<td>6.28</td>
<td>1,651</td>
<td>18,250</td>
</tr>
<tr>
<td>Sludge</td>
<td>6.7</td>
<td>N/A</td>
<td>24.4</td>
<td>67-2375</td>
</tr>
<tr>
<td>CSO</td>
<td>1.9</td>
<td>0.76</td>
<td>1.6</td>
<td>N/A</td>
</tr>
<tr>
<td>Industry</td>
<td>N/A</td>
<td>N/A</td>
<td>N/A</td>
<td>N/A</td>
</tr>
<tr>
<td>Stormwater</td>
<td>13.7</td>
<td>4.76</td>
<td>22.2</td>
<td>49-496</td>
</tr>
<tr>
<td>Tributaries</td>
<td>111.1</td>
<td>7.85</td>
<td>82.1</td>
<td>3.9-39</td>
</tr>
<tr>
<td>Groundwater</td>
<td>N/A</td>
<td>N/A</td>
<td>N/A</td>
<td>27-36</td>
</tr>
<tr>
<td>Atmospheric</td>
<td>4.3</td>
<td>2.38</td>
<td>4.4</td>
<td>51-68</td>
</tr>
<tr>
<td>TOTAL</td>
<td>196.7</td>
<td>22.02</td>
<td>1,785.4</td>
<td>19,856</td>
</tr>
</tbody>
</table>


Note that although there are 16 PAH compounds on the EPA priority pollutant list, detection limitations restrict the number which can actually yield reliable data. 2-methylnaphthalene, pyrene, and benzo(a)pyrene are more easily represented due to the fact that they are low, mid, and high molecular weight (MW) compounds, respectively. Low MW compounds come primarily from fuel oil, high MW compounds are formed as combustion products, and intermediate weight compounds probably come from a combination of the two. These three PAHs have been shown to have different distributions in Boston Harbor and Massachusetts Bay.

5.2 PAH Contaminant Inventories in the Water Column and Sediments

PAH compounds are strongly bound to the organic carbon fraction of solid materials and are found at elevated levels throughout Boston Harbor. Surficial sediment concentrations are generally in the range of 0.1-100 ppm, with average values of 1 ppm in depositional areas and 10-100 ppm near sources. The average distribution of PAH concentration among the subregions of the harbor is relatively uniform (with the Inner Harbor providing a high point). The estimated
inventory of total PAH in the top ten centimeters of the sediments is on the order of 10-20, 000 kilograms, see Table 2 below.

<table>
<thead>
<tr>
<th>Region of Harbor</th>
<th>Pyrene (ppm)</th>
<th>Benzo(a)pyrene (ppm)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Inner Harbor</td>
<td>20.1</td>
<td>20.3</td>
</tr>
<tr>
<td>Northwest harbor</td>
<td>3.13</td>
<td>1.32</td>
</tr>
<tr>
<td>Central Harbor</td>
<td>4.14</td>
<td>0.57</td>
</tr>
<tr>
<td>Southeast Harbor</td>
<td>0.68</td>
<td>0.94</td>
</tr>
<tr>
<td>TOTAL HARBOR</td>
<td>4.31</td>
<td>2.34</td>
</tr>
<tr>
<td>Inventory in the top 10 cm of the sediment (kg)</td>
<td>16,000</td>
<td>9,000</td>
</tr>
</tbody>
</table>


Assuming 0.5 g dry solids/cm³ of wet sediment and a total depositional area of 76 x 106 m³.

A limited, but representative number of sediment cores have been analyzed in detail to determine the dynamics of PAH compounds. The distribution of PAH concentration with depth in the sediment is variable, indicating a response to changes in PAH input to the harbor, as would be expected for these largely anthropogenic contaminants. As discussed later, the PAH deposited in the sediments may become a source of PAH to the water column through the mechanism of sediment-water exchange. Before investigating the transport of PAHs across this important sediment/water column interface, it is pertinent to first give a general overview of the transport and deposition of contaminated particles in Boston Harbor.

5.3 Transport and Deposition of Contaminated Sediment in Boston Harbor

The relative amount of pollutants contributed by different sources does not reflect the entire transport and fate of contaminants in Boston Harbor. Suspended particles are transported horizontally by water movements and vertically by their own settling or turbulent motions within the water column. Deposition occurs when a particle either settles to the bottom or is brought into contact with bottom sediments by the flow. Upon deposition, a particle can be resuspended into the water column or remain to be incorporated into the permanent sediments. The fate of particles discharged into a water body depends upon the interplay between transport, deposition, and resuspension. Considerations of the location of the source, the size of particles to which the contaminants are attached, and the physical characteristics of the source region, as well as the circulation of the harbor as a whole must be taken into account.

Most toxic contaminants are attached to particles. Very small particles remain suspended in the water for a long time (weeks or months) and move with the water. Unless that water stays in an area for at least the time required for a particle to settle, the particle will eventually leave the area. Heavier particles that are not flushed out of the harbor eventually settle to the bottom in areas of the harbor that favor deposition. The deposition of sediment in the harbor is affected by
the shape of the seafloor. For instance, in flat areas, sediment tends to accumulate in seabed depressions where the currents are slightly slower. By comparison, deep shipping channels increase the speed of tidal current; these currents carry away fine sediments, leaving heavier sand and gravel behind. Around the shoreline and in very shallow areas, breaking waves erode sediment. Water movement and transport of particle-bound contaminants in Boston harbor can vary considerably over very short distances.

Under the influence of a storm or an unusual tide, particle-bound contaminants in the surface layer of the bottom sediments can be resuspended into the water, carried to a different location, and sink again. This process of resuspension and transport moves contaminants away from their sources and uniform the harbor.

The sediments on the Boston Harbor floor consists primarily of natural particles that enter the harbor from offshore, or are generated by erosion of the harbor’s shoreline. Particles from sewage effluent, sludge, and CSOs are deposited along with these “clean” particles throughout the harbor.

Therefore, in summary, following discharge from a source into a defined region, a contaminated particle can undergo one of the following four processes: (1) if the particle is initially deposited in to a region where no resuspension ever occurs, the particle will remain there; (2) if the particle is initially deposited in a location where resuspension does occur, the particle may eventually be transported to and deposited at a site with no resuspension where it will remain; (3) an initially deposited and resuspended particle may be transported out of the region without ever reaching a site of ultimate deposition; and (4) a particle may be transported out of the region without ever being deposited (Alber & Chan, 1994; Stolzenbach & Adams, 1998).

Note that there have been several efforts to quantify and model the transport and deposition of contaminated sediments in Boston Harbor -- from modeling of flushing of Boston Harbor, to estimation of dispersion and particle deposition rates, to particle retention and residence time distribution approximations for various subregions of the harbor. These models are quite complicated and whose reliability are yet to be completely validated. As such, they are beyond the scope of this paper. For more information please refer to Contaminated Sediments in Boston Harbor by Keith Stolzenbach and Eric Adams, 1998.

5.4 Exchange of PAHs Between Sediments and the Overlying Water Column

In addition to possible effects on organisms living in or on the sediments, the harbor floor is potentially a source of contaminants to the overlying water. The exchange of contaminants between the water and sediments is governed in part by the relative concentrations of contaminants in each. Therefore, concerns have been raised that, as discharges of sewage to the water are abated, the sediment may become a more important source of contaminants (i.e. PAHs) to the water and could even result in violations of water quality criteria. The exchange of contaminants between the sediment and the water column depends on:

(1) the type of contaminant;
(2) the partitioning of the contaminant between solid, dissolved, and colloidal phases, and its solubility;
The profiles of PAH in the sediments indicate that the rate of deposition of these contaminants has been variable over time and that the rate of accumulation has been relatively less in recent years as PAH sources have decreased, leading to the possibility that PAH may now or in the future be transported back to the water. For environmental managers, it is of special interest to learn what the rate of PAH exchange between the water column and the sediments has been in the past, and what it will be in the future. This rate will influence whether the sediments will be a significant source of PAH to the harbor, and if so, for how long before the existing inventory of PAH in the surficial sediments is exhausted or buried by subsequent sedimentation.

There have been two initiatives undertaken to predict the future fluxes of PAH compounds pyrene and benzo(a)pyrene, from the sediments into the water column. The assumptions undertaken by both models are as follows:

a) The overlaying water above the sediments has zero concentration of dissolved PAH.

b) Pyrene and benzo(a)pyrene sorbed to sediments and colloids are in equilibrium with dissolved concentrations according to known partition coefficients and measured colloid concentrations.

c) Solution phase and sediment molecular diffusivities for dissolved contaminants and colloids were calculated from established empirical relationships

d) and, sedimentary porosity and the fraction of organic carbon are determined on the basis of measured values.

The first attempt modeled the transport of the two PAH compounds from the sediments of the Fort Point Channel core location. The fundamental transport of PAH was assumed to be by molecular diffusion of dissolved and colloidally-bound components because irrigation and bioturbation rates were assumed to be zero because of the low organism activity at this site. The study concluded the following:

1) pyrene transport by colloidal diffusion is negligible compared to molecular diffusion of dissolved phase but not for benzo(a)pyrene

2) Molecular diffusion reduces the dissolved and sorbed concentration in the sediments at a rate of reduced diffusion rate of \( D_{\text{eff}} = \frac{D_{\text{stot}}}{R} \) where \( D_{\text{stot}} = D_s + D_{\text{scol}}K_{\text{scol}}C_{\text{col}} \) and \( R = 1 + (1 - \phi)p_sK_s/\phi \), such that the depth of the region of depleted concentration is \( \delta_s = 2(D_{\text{eff}})^{1/2} \). Note \( D_{\text{eff}} \) and \( D_{\text{stot}} \) are diffusion coefficients and \( K_s \) is the partition coefficient.

3) the maximum flux from the sediments to the water is for the case with no sedimentation. An approximate theoretical value for the average flux over a period \( T \) is given by:
4) The depletion of PAH in the sediments is superposed on the burial of the contaminated material under new clean material at a rate, \( w \). Therefore, the average flux over a period \( T \) is

\[
J_s = \frac{c_s D_{sm} \Phi}{4 \pi K_v w T} \left[ \frac{(D_{sm} \Phi(1 - \Phi))}{4 \pi K_v T} \right]^{1/2}
\]

for \( T > D_{eff}/w^2 \)

See Table 3 for the values for PAH flux as predicted by this particular model for the sediment/water interface.

Table 3. PAH fluxes through the Sediment-water Interface According to the First

<table>
<thead>
<tr>
<th>Case No.</th>
<th>Burial Rate (cm/yr)</th>
<th>Pyrene (Model Value)</th>
<th>Pyrene (Estimated Value)</th>
<th>Benzo(a)pyrene (Model Value)</th>
<th>Benzo(a)pyrene (Estimated Value)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>0</td>
<td>0.13</td>
<td>0.04</td>
<td>0.0040</td>
<td>0.003</td>
</tr>
<tr>
<td>2</td>
<td>0</td>
<td>0.13</td>
<td>0.04</td>
<td>0.0061</td>
<td>0.007</td>
</tr>
<tr>
<td>3</td>
<td>0.01</td>
<td>0.12</td>
<td>0.04</td>
<td>0.0038</td>
<td>0.007</td>
</tr>
<tr>
<td>4</td>
<td>0.1</td>
<td>0.038</td>
<td>0.02</td>
<td>0.0010</td>
<td>0.0007</td>
</tr>
<tr>
<td>5</td>
<td>0.25</td>
<td>0.014</td>
<td>0.014</td>
<td>0.000028</td>
<td>0.0003</td>
</tr>
</tbody>
</table>

Model


Units for fluxes \( \mu g/cm^2/yr \)

The second study modeled the transport of pyrene and benzo(a)pyrene from the sediments of all three sampling sites in Boston Harbor. There are several different assumptions which distinguish this attempt with the first. The most important is the assumption that the concentration...
profile and the flux of the contaminant from the sediments to the water have reached steady-state values and that the concentrations of pyrene and benzo(a)pyrene in the sediment are constant at given values of depth in the sediments. Vertical transport of contaminants in the sediments was assumed to be by molecular diffusion and by diffusive bioturbation. The major conclusions of this second initiative was as follows:

1) Contaminants in the sediments are mostly in the sorbed phase and are moved vertically primarily by bioturbation of the sediments;

2) The actual flux through the sediment-water column interface in controlled by a combination of water-side diffusive resistance and the specified rate of bioturbation as given by:

\[
J_s = \frac{c_s}{K_s} = \frac{\delta_w L}{D_{tot} + (1 - \phi)\rho_s K_s D_b}
\]

where \(D_{tot} = D_s + D_{scol}K_{scol}c_{col}\)

See Table 4 for the predicted values according to the above model.

Table 4. PAH fluxes through the Sediment-water Interface According to the Second Initiative

<table>
<thead>
<tr>
<th>Core Location</th>
<th>Pyrene (Model Value)</th>
<th>Pyrene (Estimated Value)</th>
<th>Benzo(a)pyrene (Model Value)</th>
<th>Benzo(a)pyrene (Estimated Value)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fort Point</td>
<td>2.5</td>
<td>3.00</td>
<td>0.23</td>
<td>0.24</td>
</tr>
<tr>
<td>Channel</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Peddocks Island</td>
<td>0.17</td>
<td>0.18</td>
<td>N/A</td>
<td>N/A</td>
</tr>
<tr>
<td>Spectacle</td>
<td>4.30</td>
<td>5.00</td>
<td>0.19</td>
<td>0.19</td>
</tr>
<tr>
<td>Island</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>


Units for fluxes \(\mu g/cm^2/yr\)

6.0 Conclusions

A better understanding of the transport of polycyclic aromatic hydrocarbons is needed to better quantify the fate of PAHs in the marine environment. Only after such a rigorous modeling...
methodology is undertaken can the ecological and human risks associated with PAHs be truly anticipated. In addition, the Boston Harbor Navigation Project's success depends directly on the ability to reliably represent the deposition of contaminated material and the transport of the contaminated material afterwards. The paper has attempted to give a brief overview on pertinent aspects of the source of PAHs, the deposition of PAHs in Boston Harbor, and certain preliminary modeling of the exchange of PAHs between sediments and the water column.
Appendix A References

Sources, Fate and Effects of Polycyclic Aromatic Hydrocarbons in Massachusetts Bays: The Science Behind the Management Issues; Massachusetts Bay Marine Studies Consortium (1996).


Internet searches.

Interviews.

Boston Harbor Project Weekly Discussions.
APPENDIX B

User Manual for Treeplan

Source: Decision Analysis System website
TreePlan helps you build and modify decision tree diagrams in Excel worksheets. TreePlan automatically puts formulas on your worksheet for evaluating these trees.

To build a new decision tree, first create a new worksheet. Then start TreePlan in one of two ways:

1. Choose Decision Tree... from the Options menu (Excel 4) or the Tools menu (Excel 5, 7, and 97), or
2. Press the shortcut key: Control + t.

When the TreePlan...New dialog box appears, choose New Tree. TreePlan builds a tree diagram with an initial decision node and two branches. The tree diagram begins with the upper left corner of the diagram near the active cell at the time New... is chosen. TreePlan assigns the name TreeDiagram to the range of the tree diagram and initially sets Excel's Print Area equal to TreeDiagram.

To change the structure of the tree diagram, select a node, start TreePlan, and choose commands from a TreePlan dialog box.

On the left side of each branch of the tree diagram, there is a name field above the branch line and a partial-cash-flow value field below the line. On the left side of event branches, there is also a probability field above the name field. Decision nodes also have a number in them indicating the optimal branch.

On the right side of each branch of the tree diagram, there is a rollback Expected Value field below the line near each node. TreePlan automatically puts the rollback EV formula into this cell: maximum of successor EVs at a decision node, and expected value of successor EVs at an event node.

To the right of the terminal nodes, there is an endpoint value field that sums all of the partial cashflows in the tree.

Internally, TreePlan uses the TreeData range, near cell GV1000, to construct the TreeDiagram.

CAUTION: Do not insert or delete rows or columns in the TreeDiagram or TreeData ranges. Modify the tree diagram only by using TreePlan's menu options.
Add branch
   Adds a single branch after the selected node.
   (No more than 5 branches are allowed.)

Copy subtree
   Copies the selected node and all its successors to the
   TreePlan clipboard.

Insert decision
   Inserts a decision node and single branch before the
   selected node.

Insert event
   Inserts an event node and single branch before the
   selected node.

Change to decision
   Changes the selected event node to a decision node and
   erases the probability fields from the event branches.

Change to event
   Changes the selected decision node to an event node.

Shorten tree
   Removes the selected node and its single successor
   branch.

Change to terminal
   Changes the selected node to a terminal node. All
   successor branches are erased.

Remove branch
   Erases the selected node, the previous branch, and any
   successor branches and nodes.

TREPLAN...TERMINAL dialog box

Change to decision node
   Changes the selected terminal node to a decision node
   with one to five successor branches.

Change to event node
   Changes the selected terminal node to an event node
   with one to five successor branches.

Paste subtree
<table>
<thead>
<tr>
<th>TREEPLAN...OPTIONS dialog box</th>
</tr>
</thead>
<tbody>
<tr>
<td>Certainty Equivalents</td>
</tr>
<tr>
<td>The default is to rollback the tree using expected values.</td>
</tr>
<tr>
<td>If you chose to use exponential utilities, TreePlan will</td>
</tr>
<tr>
<td>compute utilities and certainty equivalents at each node.</td>
</tr>
<tr>
<td>For the Maximize option, the rollback formulas are</td>
</tr>
<tr>
<td>$U = A - B \cdot \exp(-X/RT)$ and $X = -\ln((A - U)/B) \cdot RT$, and for the</td>
</tr>
<tr>
<td>Minimize option, $U = A - B \cdot \exp(X/RT)$ and</td>
</tr>
<tr>
<td>$X = \ln((A - U)/B) \cdot RT$.</td>
</tr>
<tr>
<td>NOTE: TreePlan uses the name RT to represent the</td>
</tr>
<tr>
<td>risk tolerance parameter of the exponential utility</td>
</tr>
<tr>
<td>function; the names A and B determine scaling.</td>
</tr>
<tr>
<td>If the names A, B, and RT don't exist, they are initially</td>
</tr>
<tr>
<td>defined as $A=1$, $B=1$, and $RT=999999999999$. The</td>
</tr>
<tr>
<td>name UseExpUtility is a flag indicating whether to use</td>
</tr>
<tr>
<td>exponential utilities or expected values.</td>
</tr>
<tr>
<td>Decision Node EV/CE Choices</td>
</tr>
<tr>
<td>The default is to Maximize profits. If you choose to</td>
</tr>
<tr>
<td>Minimize costs instead, the cash flows are interpreted</td>
</tr>
<tr>
<td>as costs, and decisions are made by choosing the</td>
</tr>
<tr>
<td>minimum expected value/CE rather than the maximum.</td>
</tr>
<tr>
<td>TreePlan uses the name MinimizeCosts as a flag</td>
</tr>
<tr>
<td>indicating whether to maximize profits or minimize</td>
</tr>
<tr>
<td>costs.</td>
</tr>
</tbody>
</table>