A review of issues related to the development of a food web model for important prey of endangered species in Massachusetts and Cape Cod Bays

Massachusetts Water Resources Authority

Environmental Quality Department
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FINAL

A Review of Issues Related to the Development of a Food Web Model for Important Prey of Endangered Species in Massachusetts and Cape Cod Bays

submitted to

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The conclusions drawn in this report are primarily those of the lead author based on the information provided by the co-authors and MWRA. The extensive and detailed reviews by Mr. Ken Keay and Dr. Michael Mickelson of MWRA are acknowledged and appreciated

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

EXECUTIVE SUMMARY

In May of 1998, Region I of the US Environmental Protection Agency (EPA) and the Massachusetts Department of Environmental Protection (MADEP) issued a draft NPDES permit for the Massachusetts Water Resources Authority's (MWRA) new outfall (outfall T01) for public comment. Included in the draft permit was a requirement for the development of a scope of work for a food web model by December 31, 1998. The final NPDES permit that was released on May 19, 1999 expanded the permit language to address actions already completed by the MWRA in 1998 in response to the draft permit language. The cognitive permit section (7. Ambient Monitoring Plan) now reads:

a. The MWRA shall: (1) implement the monitoring plan described in Attachment N, (2) update, maintain, and run the three dimensional hydrodynamic water quality "Bays Eutrophication Model" developed in 1995 by HydroQual and the USGS, on a routine basis (at least every year), for the purpose of predicting conditions caused by nutrient loading and in order to support decisions about the need for nutrient limits and the appropriate level of any such limit for the discharge, and (3) implement plume tracking, including the use of acoustical technology, to understand the dilution available for the discharge. The MWRA has developed a scope of work for a food web model to characterize the seasonal abundance for important prey species of endangered species in the Massachusetts and Cape Cod Bays. EPA and the MADEP, in consultation with the OMSAP discussed below, shall provide the MWRA with comments on this scope of work. Within ninety (90) days after receipt of these comments, MWRA shall submit a revised scope of work for review by OMSAP, and for approval by EPA and the MADEP. After receipt of the revised scope of work, EPA and the MADEP will determine whether implementation of the food web model is warranted. The food web model shall: (a) include phytoplankton, zooplankton, planktivorous fish and marine mammals including endangered whale species, (b) allow an evaluation of the strength and likelihood of potential stressors that may alter the food web, (c) be based on results of ongoing monitoring, special studies of plankton (phytoplankton and zooplankton) dynamics and any other current or historical research in Cape Cod Bay. The MWRA may choose to fulfill the obligations described in this paragraph by ensuring that these items are performed by another entity.

The MWRA developed a conceptual food web model (Kelly *et al.* 1998) and scope of work for developing a food web model (Hunt *et al.* 1999) for consideration by the Outfall Monitoring Science Advisory Panel (OMSAP) in 1998. The latter was presented to the OMSAP in December 1998. The MWRA distributed the full scope of work for review by OMSAP, the Interagency Advisory Committee (IAAC), and Public Advisory Committee (PIAC) membership in May 1998. The scope of work is available at http://www.mwra.state.ma.us/harbor/enquad/pdf/99-09_enquad_report.pdf).

The first task defined in the scope of work is a review of the major environmental impact evaluations conducted during the outfall planning and construction. These included the EPA (1998) Supplemental Environmental Impact Statement (SEIS) and the EPA/NMFS endangered species consultation (EPA 1993; NMFS 1993). These Federal technical reviews of the MWRA project determined that upgrading sewage treatment and relocating the effluent discharge into Massachusetts Bay would have not have adverse consequences on the Massachusetts and Cape Cod Bays ecosystem (EPA 1998) and that the relocation would not jeopardize endangered marine mammals within the Bays (EPA 1993; NMFS 1993). The review to be conducted under the first task would revisit, using new information, two questions: "will environmental conditions worsen as a result of the outfall relocation?" and if so "is such change likely to harm whales?"

The logic of the scope of work maintains that if the answer to the first question is no, then the value of developing a food web model is questionable. However, continued monitoring in its present form would be indicated and would continue as required under the permit. If the answer to the first question were yes, an evaluation of the potential harm to the endangered species would be made. If the information from this evaluation indicated harm to the whales was not likely, monitoring would continue as conducted presently. If harm were indicated from the assessment, additional research would be indicated which could include the development of a food web model.

The purpose of this report is to implement the first task of the scope of work. To address the questions posed in the scope of work, several activities were pursued. The first was a review of the recent monitoring data to determine if conditions were different than assumed under in EPA (1988) and Biological Assessment (EPA 1993) and Biological Opinion (NMFS 1993). The second was to compare dilution fields and expectations based on the final 3-D hydrodynamic modeling of the effluent dilutions (note the BA used early information from the model which was subsequently finalized in Signell *et al.* 1996). The third was to perform sensitivity and mass balance modeling using the calibrated Bays Eutrophication Model (BEM) model to determine expectations for changes in nutrient fields and plankton biomass as measured by chlorophyll. The fourth was to develop a better understanding of food web modeling approaches and capabilities relative to their ability to address the occurrence of right whales in Massachusetts Bay and evaluate potential linkages to the MWRA outfall.

The data review and comparisons determined the following:

1. Present nitrogen loading from the MWRA treatment plants is less than assumed in 1988.

2. The Deer Island effluent contributes a small fraction (~3%) of the total nitrogen entering the system.

3. Nitrogen entering at the boundaries of Massachusetts Bay exerts more influence on the total nitrogen concentrations in the farfield areas than the effluent discharge does.

4. BEM and 3-D hydrodynamic model results demonstrate that nutrient concentrations above the background variability will be confined to a small area near the outfall.

5. Elevated nutrient levels in the coastal region (from Boston Harbor southward towards Plymouth) will be unchanged or slightly lower with transfer of the effluent discharge location to Massachusetts Bay.

6. BEM model results predict little change in spatial or temporal patterns of nutrient concentrations in Cape Cod Bay relative to the current and future effluent discharge locations.

7. 3-D hydrodynamic model computations estimate the area in Massachusetts and Cape Cod Bays that would be under measurable influence from the discharge is small (only 7 km^2 which is <0.2percent of the combined area of Massachusetts and Cape Cod bays).

8. 3-D hydrodynamic model computations predict that the effluent nutrient concentrations will be diluted to 200:1 within a few kilometers of the outfall diffuser, and thus will be indistinguishable from background.

9. Change in the nutrient fields in Massachusetts Bay will be highly localized and have little to no impact on the phytoplankton and zooplankton species distributions and communities in the Bay.

10. Nutrient levels in Massachusetts and Cape Cod Bays will not be enriched to levels that promote the growth of nuisance species such as the "red tide" organism *Alexandrium.*

11. BEM computations project small increases in the DO in bottom waters of the nearfield in the summer.

These results are consistent with those found in the previous ecological assessments completed for the MWRA outfall in Massachusetts Bay. The results also indicate that the conclusions and projections drawn in the previous assessments were based on conservative assumptions. Thus, the data from the monitoring program and refined model computations indicate that the environmental conditions in Massachusetts Bay will not be worse than projected. Rather, they indicate that the system is likely to experience even less change than previously predicted.

Based on the review completed in this report, it is concluded that adverse changes to the ecology and functioning of the Massachusetts Bay system will not occur as a result of the outfall relocation. Recent model computations indicate that ecological impact may be less and have less spatial extent than projected in the environmental assessments. This further argues for no net change in the system after relocation.

The major farfield area affected will be Boston Harbor where the effects from nitrogen loading are expected to lessen. As a result, chlorophyll levels in the harbor are expected to decrease and dissolved oxygen levels in the inner harbor to rebound to high concentrations. Planktonic communities (either biomass or species distributions) in Massachusetts Bay are not expected to change as a result of the relocation. Plankton communities in the Cape Cod Bay and Stellwagen Bank areas are also not expected to change as result of the relocation. Thus, shifts in the food supply (either species or abundance) of the right whale are not expected. This species responds to many factors and conditions; most of these are external to the Bays. Therefore, because the nutrient inputs, concentration, and distribution, and plankton distributions will not change with the relocation, it is unreasonable to assume that detrimental effects on the occurrence of the whales will occur.

Moreover, the development of a food web model that endeavors to link the outfall discharge to the occurrence of right whales in the Bays would likely be an exercise in futility. The futility arises from several factors. The first is that these food web models are most effective when addressing measurable perturbations in a system, and such perturbations are not expected to result from outfall relocation. The second is the requirement that the food web models have complete and accurate species-by-species biomass information. This set of data is difficult to obtain and its accuracy cannot be easily ascertained. The third is uncertainty in the overall importance of the Bays to the energetics of the whales (i.e., inability to close the food web model domain). The fourth is that food web model development at a local or habitat specific scale is unwarranted given the importance of external factors that affect the distribution of the whales. As identified in a 1998 workshop convened to address knowledge of right whale distribution and predictability of the whale distribution (Clapham 1998), much research must be conducted to understand the factors that affect the population and its distribution. It is clear from the discussions and conclusions of this workshop that federal research dollars must be made available to address the fundamental questions raised. These questions must be addressed before predictive models can be developed.

The recommendations in Clapham (1998) provide a clear set of research and modeling directions related to the right whale and its occurrence in not only Massachusetts Bay, but over its entire range. Thus, funding of the key research and modeling needs identified from the workshop, which are more likely to fill the integrated long-term, large-scale research demanded for the overall management of right whales, is recommended. Moreover, the clear large scale spatial issues related to the protection and management of this species points to the need for broader agency involvement (federal, regional, and state levels) to effectively address the pressing issue of the salvation of the northern right whale population.

1. INTRODUCTION

1.1 Background

In May of 1998, Region I of the US Environmental Protection Agency and the Massachusetts Department of Environmental Protection (MADEP) issued a draft NPDES permit for the Massachusetts Water Resources Authority's (MWRA) new outfall (outfall T01) for public comment. Included in the draft permit was a requirement for the development of a scope of work for a food web model by December 31, 1998. The draft NPDES permit (page 9) specified that as part of ambient monitoring:

> "The MWRA shall: … by December 31, 1998, develop a scope of work for a food web model to characterize the seasonal abundance for important prey species of endangered species in Massachusetts and Cape Cod Bays. The food web model shall: (a) include phytoplankton, zooplankton, planktivorous fish and marine mammals, (b) allow an evaluation of the strength and likelihood of potential stressors that may alter the food web, (c) be based on results of ongoing monitoring, special studies of plankton (phytoplankton and zooplankton) dynamics and any other current or historical research in Cape Cod Bay, and (d) be reviewed by the science panel described under section 7d below. The MWRA may choose to fulfill the obligations described in this paragraph by ensuring that these items are performed by another entity."

The draft permit further indicated that "on or after December 31, 1998, EPA will review all available information, including the results of all on-going monitoring and special studies, and models, and develop any appropriate requirements for additional monitoring and modeling in Massachusetts and Cape Cod Bays. The monitoring plan described in Attachment N of the permit shall be modified to reflect these additional requirements."

EPA's overview of the permit (http://www.epa.gov/region01/reginit/overview.html) indicated:

> "Concerns have been raised about the potential impact of the outfall on plankton species in Massachusetts and Cape Cod Bays, especially the formation and composition of zooplankton patches which are a key food source for right whales. The permit requires the MWRA to develop a scope of work for a study that would evaluate and model the food web for endangered species in Massachusetts and Cape Cod Bays. When that scope of work is completed (required by the end of 1998), EPA will develop appropriate additional requirements for monitoring and modeling activities. These additional requirements will be incorporated into the monitoring plan."

The overview above, plus clarification from EPA (see minutes of the 10/27/98 meeting of the Outfall Monitoring Science Advisory Panel http://www.epa.gov/region01/omsap/index.html), document that the intended focus of the effort is on right whales rather than on other endangered species.

The MWRA initiated planning to respond to this permit condition in early 1998 by developing a conceptual food web model (Kelly *et al.* 1998). The conceptual model was presented to the Outfall Monitoring Task Force in the spring of 1998. This effort continued in late 1998 with development of possible modeling approaches. These were presented to the OMSAP for review and guidance at their October 1998 meeting (see minutes to the October 27, 1998 OMSAP meeting). The modeling goals and approaches that were presented included the following:

1. ***To understand the abundance (population density on a scale of tens of kilometers) of endangered species prey.*** The conceptual food web model presented in the Kelly *et al.* (1998) was considered the first step toward describing the food web of right whale prey especially in relation to outfall nutrient effects.

2. ***To further understanding of the availability (meter-scale population density or patchiness, and age structure) of right whale prey***. This approach would have entailed development of a patch formation model, and be essentially unrelated to the outfall or to food web modeling. It would, however, more directly address some of the known issues of concern for whale feeding.

3. ***To understand the effect of the outfall on whale prey***. The approach for this model would have been to extend the Bays Eutrophication Model (BEM) (HydroQual and Normandeau 1995) to include zooplankton at the species level or at some representation of species groups. However, BEM's grid scale is about 100 times coarser than the patches that are most relevant to right whales.

Discussion by the OMSAP members during their October 27 meeting did not favor any of the three approaches that the MWRA put forth (OMSAP Meeting Minutes October 27, 1998). The MWRA interpreted the noted modeling uncertainties, the plethora of unanswered questions, as well as the lack of consensus about a possible Food Web Model (FWM) approach to indicate that to "lock in" on a given modeling approach would be premature. The MWRA therefore developed a more incremental approach for the model development process as a response to the permit requirement. In addition to the tasks necessary to develop a FWM (Hunt *et al.* 1999), the proposed approach incorporated evaluations required to establish a full understanding of the present condition and potential for change in Massachusetts Bay. These would be based on recent monitoring and research data and the calibrated BEM modeling results. The approach included a review of key assumptions made in the environmental impact assessment conducted for the outfall siting (EPA 1988) and subsequent biological reviews (EPA 1993; NMFS 1993). These reviews were deemed necessary to ensure the OMSAP was fully apprized of the historical context of the impact assessments and current context of the monitoring program, and from which they could recommend whether or not actual model development is warranted, and if so, the levels at which the development should proceed. This process culminated in December 1998 with a presentation of the incremental approach the MWRA developed in responding to this permit condition (see minutes to the December 18, 1999 OMSAP meeting).

Subsequent to this activity, the final NPDES permit was released on May 19, 1999 and took effect June 19, 1999. The entire permit can be found at http://www.epa.gov/region01/pr/files/052099.html. Section 7 of the permit includes the following relative to the food web model (italics added):

"a. The MWRA shall: (1) implement the monitoring plan described in Attachment N, (2) update, maintain, and run the three dimensional hydrodynamic water quality "Bays Eutrophication Model" developed in 1995 by Hydroqual and the USGS, on a routine basis (at least every year), for the purpose of predicting conditions caused by nutrient loading and in order to support decisions about the need for nutrient limits and the appropriate level of any such limit for the discharge, and (3) implement plume tracking, including the use of acoustical technology, to understand the dilution available for the discharge. *The MWRA has developed a scope of work for a food web model to characterize the seasonal abundance for important prey species of endangered species in the Massachusetts and Cape Cod Bays. EPA and the MADEP, in consultation with the OMSAP discussed below, shall provide the MWRA with comments on this scope of work. Within ninety (90) days after receipt of these comments, MWRA shall submit a revised scope of work for review by OMSAP, and for approval by EPA and the MADEP. After receipt of the revised scope of work, EPA and the MADEP will determine whether implementation of the food web model is warranted. The* *food web model shall: (a) include phytoplankton, zooplankton, planktivorous fish and marine mammals including endangered whale species, (b) allow an evaluation of the strength and likelihood of potential stressors that may alter the food web, (c) be based on results of ongoing monitoring, special studies of plankton (phytoplankton and zooplankton) dynamics and any other current or historical research in Cape Cod Bay. The MWRA may choose to fulfill the obligations described in this paragraph by ensuring that these items are performed by another entity.*

In response to this requirement the MWRA distributed a copy of the full scope of work (Hunt *et al.* 1999) for review by OMSAP, IAAC, and PIAC membership. The scope of work is available on the Internet at http://www.mwra.state.ma.us/harbor/enquad/pdf/99-09_enquad_report.pdf).

The scope of work addresses the key concerns listed in the permit: (a) to include phytoplankton, zooplankton, planktivorous fish and marine mammals, (b) to allow an evaluation of the strength and likelihood of potential stressors that may alter the food web, (c) to be based on results of ongoing monitoring, special studies of plankton (phytoplankton and zooplankton) dynamics and any other current or historical research in Cape Cod Bay. Towards this end, this scope of work focuses on the key factors that affect the seasonal abundance for important prey species of the endangered species that inhabit Cape Cod Bay but focuses on the food web of the right whale as it is of greatest concern in Cape Cod Bay.

The process that the MWRA would follow to guide model development is shown in Figure 1-1 (from Hunt *et al.* 1999). The framework would be implemented in an incremental manner with decision points for process review, including recommended decision criteria for stopping or continuing, suggested end points, and definition or redefinition of subsequent steps. This incremental approach is required to ensure the relevance of the model effort, to consider its predictive skill, and to define the appropriate modeling framework.

The first activity to be conducted under the scope of work calls for a revisit of the major impact evaluations in the SEIS (EPA 1988) and in the later EPA/NMFS endangered species consultation. Three major Federal technical reviews of the MWRA project determined that upgrading treatment and relocating the effluent discharge would have no unacceptable consequences for Massachusetts and Cape Cod Bays (EPA 1988), and would not jeopardize endangered marine mammals within the Bays (EPA 1993, NMFS 1993). The first task includes review of the assumptions used in the assessments and new data that have become available since the assessments were completed. This review and reassessment was designed to address two questions that are included in the scope of work: "Will environmental conditions worsen as a result of the outfall relocation?" and if so "Is such change likely to harm whales?"

The logic of the scope of work maintains that if the answer to the first question is no, then the value of food web modeling is questionable. However, continued monitoring in its present form would be indicated and would continue as required under the permit. An evaluation of whether or not to modify the present monitoring program could be made as long as specific questions that can and should be addressed by a monitoring program can be defined relative to the endangered species. If the evidence addressing the first question indicates that adverse impacts are likely, then an evaluation of the potential harm to the endangered species would be made to further understand the potential for impact. If the answer to the second question is no, then continued monitoring, including additional studies mandated by the contingency plan to address any adverse environmental impacts that occur, would be indicated. If available information is equivocal, further research to define significant linkages must be conducted as part of basic research on the endangered species and should proceed as part of a larger effort to understand man's impact to these species.

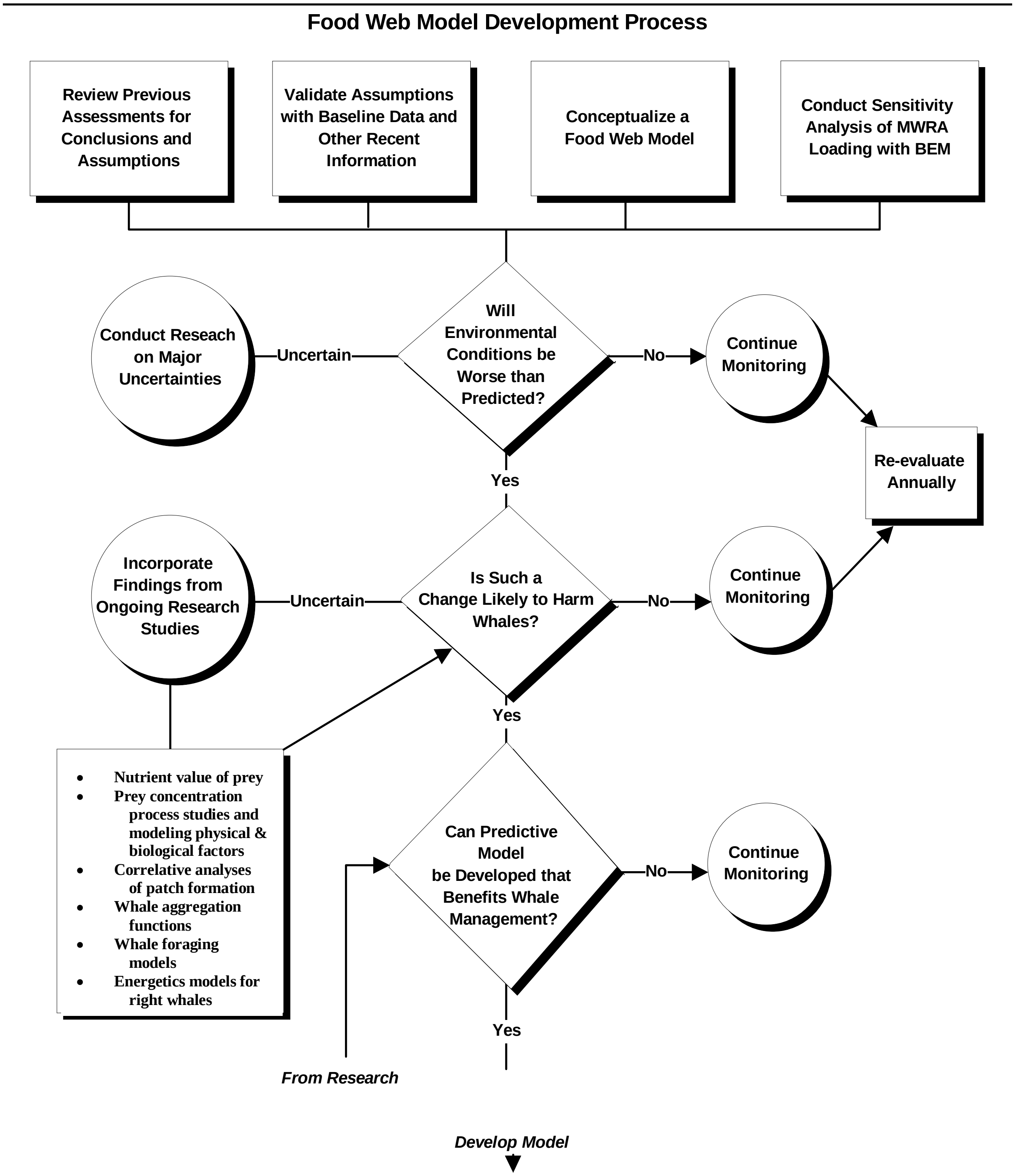

Figure 1-1. Food Web Model Development Process – Initial Assessments.

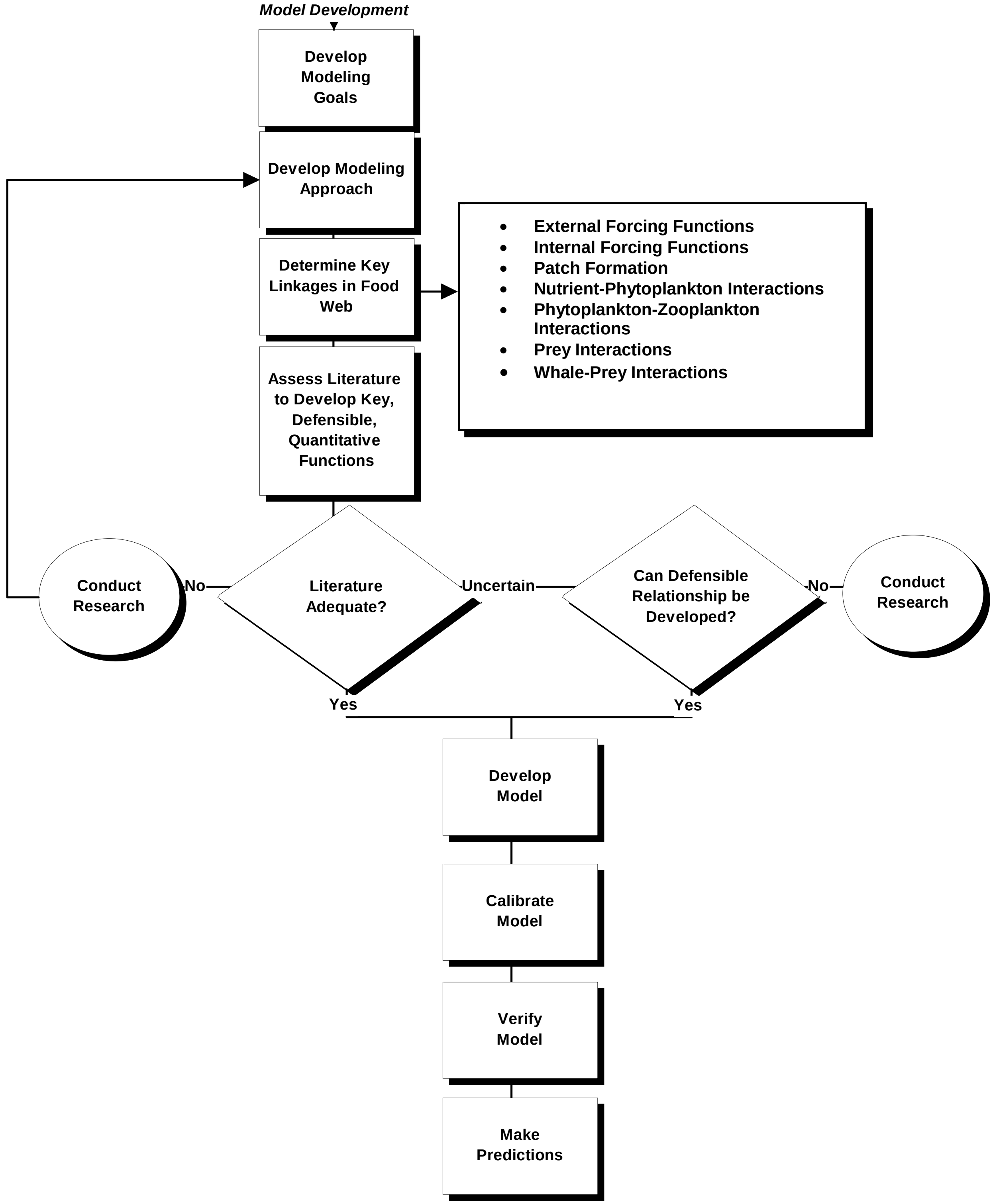

Figure 1-2. (continued).

Moreover, if this reassessment leads to the conclusion that impacts of relocating the outfall discharge are greater than predicted in previous reviews, and that whales are likely to be impacted by the change, development of the food web model would proceed. Such an effort could require substantial research into fundamental processes that control the various linkages between the nutrients, phytoplankton, prey species, and whales. The steps required to complete the modeling are described in Hunt *et al*. (1999).

1.2 Purpose

This report directly addresses three of the four activities defined in the MWRA food web model scope of work under Task 1 (the entire first row of Figure 1-1). The fourth activity, to conceptualize a food web model, has already been addressed in Kelly (*et al*. 1998) and is only briefly summarized here in the context of the overall model development questions.

The report is designed to address issues central to a FWM and to place current understanding of the Massachusetts Bay system, including recent nutrient loading data from the MWRA effluent at Deer Island, into context of the historical assessments. It also provides information by which the key assumptions made in the historical assessment documents can be evaluated for accuracy and updated. Moreover, some of the data provided to the OMSAP on December 18, 1998 represent new research and modeling sensitivity results that are relevant to the sources of nutrients sources to Massachusetts Bay and potential system wide responses that may result from the outfall relocation. In addition, these new BEM modeling outputs address water quality responses in the Bays relative to extreme hypothetical changes in the nutrient input by the MWRA outfall (present and future) from a local and regional perspective. It also enables calculation of nutrient mass balances inclusive of transport across the Bay's ocean boundaries. This review provides the opportunity to capture this new understanding into one document and to place the information into the context of the food web modeling issues. Finally, although each of the above assessments conclude that food web modeling in not warranted in the context of the MWRA outfall, a later section of this report reviews the state of the art in food web modeling. This was included because of the general need within the public, regulatory, and scientific community to address current understanding of the attributes, capability, advantages, limitations and constraints of food web modeling in general and more specifically in relation to the endangered species in Massachusetts and Cape Cod Bays. Thus, the report includes general reviews of the state of food web modeling from the two major modeling approaches presently available: foodweb analysis and network analysis.

1.3 Report organization

The comparison of historical assumptions relative to potential impact of the MWRA outfall in Massachusetts Bay to baseline data is discussed in Section 2. A discussion of the sensitivity analysis and nutrient mass balance information developed using the BEM model is presented in Section 3. Section 4 includes the reviews of food web modeling approaches and feasibility relative to the right whale occurrence issue. Section 5 discusses food web modeling in relation to the outfall and our current understanding of the Massachusetts and Cape Cod Bays including recommendations for further activities. References cited are included in Section 6.

2. COMPARISON OF ASSESSMENT AND BASELINE INFORMATION

Ecological assessments of potential impact from the relocation of the Massachusetts Water Resource Authority's treated sewage outfall into Massachusetts Bay were completed in the late 1980's and early 1990's (EPA 1988, EPA 1993, NMFS 1993). The assessments were conducted with the best available environmental data at the time of the assessments. Since publication of these assessments, the MWRA and others have developed a substantial database and understanding of the ecological functioning and transport mechanisms operating in Massachusetts and Cape Cod Bays. In addition, more sophisticated water quality modeling of the ecosystem (HydroQual and Normandeau 1995) and effluent plume dilution and dynamics have been undertaken (Signell *et al.* 1996). These assessments have been consistent in finding that that impact from the MWRA outfall in Massachusetts Bay will be limited and confined to an area very near the outfall (EPA 1988; 1993). The biological opinion (NMFS 1993) also concluded that the relocated discharge would not jeopardize the continued existence of endangered species. In spite of the lack of direct evidence or quantitative predictions of adverse impact to the food web of endangered species, concerns that the discharge may have an adverse effect on the food web of endangered species remained (October 27, 1998 OMSAP Meeting Minutes). The species of most concern is the northern right whale that visits the Bays seasonally. To address this concern, it was suggested that food web modeling could provide predictions of impact to these animals from the relocated MWRA outfall.

Before conducting any modeling efforts, which require substantial time and information to complete successfully, the MWRA felt that a review of the recent monitoring data and information should be conducted to determine whether or not a modeling effort was warranted. Comparison of the new data to the assumptions made in the assessments enables examination of the continuing validity of the prior conclusions before the outfall becomes operational in 2000-2001. To conduct this comparison, the major environmental assessments associated with the outfall along with other relevant documents (see Table 2-1) were reviewed. Key factors in these reports were used to develop the comparisons and draw conclusions. The assumptions were summarized and compared with recent monitoring data to determine the continuing validity of the previous conclusions. In addition, the more recent hydrodynamic modeling and expected plume dilution model results (Signell *et al.* 1996) were examined. An assessment of whether the historical estimates and predictions remained accurate was then made.

Table 2-1. Documents reviewed for comparisons

EPA SEIS for the Outfall (EPA 1988)
EPA Biological Assessment (EPA 1993)
NMFS Biological Opinion (NMFS 1993)
Cape Cod Commission Review of Biological Assessment (BCC 1993)
Dilution Transport model (Signell *et al.*1996)
MWRA Toxics Review (Mitchell *et al.*1997)
Conceptual Food Web Model (Kelly *et al.*1998)
MWRA Zooplankton Retrospective (Lemieux *et al* 1998)
MWRA Water Column Baseline Monitoring Data 1992-1998 (Kelly and Turner 1995a; Kelly and Turner 1995b; Murray *et al.* 1997; Cibik *et al.*1998a; Libby *et al.* 1999)
MWRA Phytoplankton review (Cibik *et al.*1998b)
Humpback whale disappearance 1986 (Jahoda and Ryer 1988)
"Predicting Right Whale Distributions Workshop" summary (Clapham 1998)

2.1 Review of assumptions in the SEIS, BA and NMFS biological opinion

The key assumptions and information used in EPA (1998), EPA (1993), and NMFS (1993) relative to the environment in Massachusetts Bay and MWRA inputs are summarized in this section. The assumptions addressed in this report focus on nutrient related issues, as these are central to the food web modeling discussion. Toxic contaminant related issues are not specifically addressed in this report. However, the assumptions included in the SEIS (EPA 1988) have been shown to be overly conservative (Shea and Kelly 1992). Additionally, monitoring of the secondary effluent has shown that contaminant concentrations in the MWRA sewage effluent have decreased substantially with the cessation of sludge discharge (12/91), the advent of improved primary treatment (1/95), two batteries of secondary treatment (7/97 and 2/98), and treatment of the Nut Island sewage flow at the more efficient Deer Island treatment plant (7/98). This has resulted in a substantial decrease in the input of contaminants to Boston Harbor and Massachusetts Bay from the Deer Island outfall (see for example Figure 2-1). Calculations of contaminant concentrations after initial dilution show that marine water quality criteria will not be exceeded in the receiving waters (Graf and Bigornia-Vitale 1999; Sung and Higgins 1998; Butler *et al.* 1997; Hunt *et al.* 1995). Therefore, these toxic compounds are not expected to have adverse impact on the receiving environment and are not addressed further in this review.

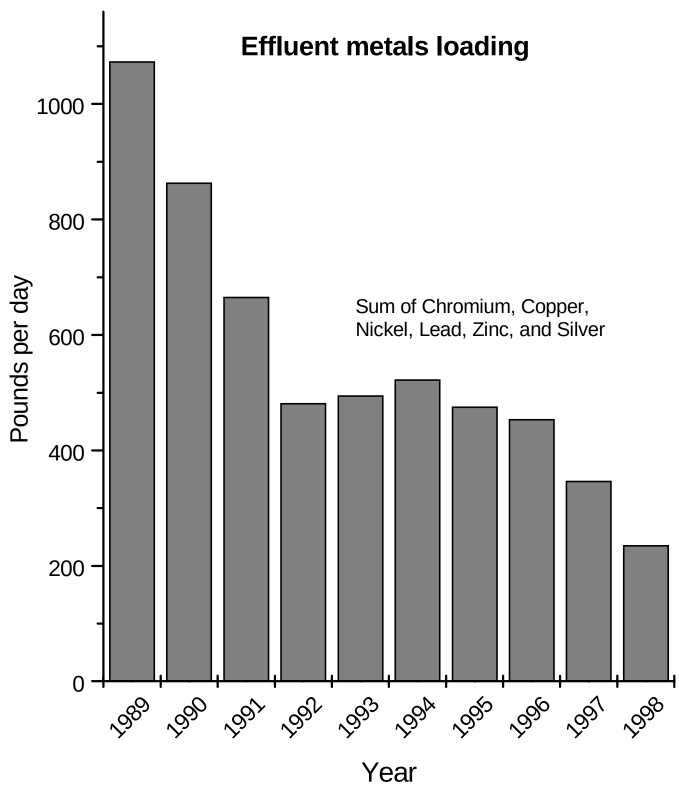

Figure 2-1. Comparisons of the summed annual loading of chromium, copper, nickel, lead, silver and zinc in the Deer Island effluent from 1989 through 1998 (figure courtesy of the MWRA).

Note also that the EPA's SEIS was prepared under the assumption that approximately three years of primary treated effluent would be discharged at the new outfall site. Due to the delay in the completion of the outfall tunnel, only secondary effluent will be discharged. Thus, only the potential effects of secondary treated effluent are relevant to this reassessment. The key nutrient-related areas of concern and related assumption(s) and conclusions in EPA (1998), EPA (1993), and NMFS (1993) are summarized in Table 2-2.

Table 2-2. Summary of key assumptions included in previous environmental assessments related to the MWRA outfall in Massachusetts Bay.

Area of concern	Assumption/finding
Nutrient loading	1. Nitrogen loading under secondary treatment will be 12,300 metric tons per year (EPA 1988, 1993; NMFS 1993) 2. Annual nutrient loading at the new outfall location will be equal to that at the Deer Island outfall 3. Nutrient removal during secondary treatment and retention of nitrogen in Boston Harbor from present outfall location were not explicitly considered in the SEIS (EPA 1988 and D. Tomey. EPA Region I, personal communication, May 1999)
Nutrient levels in the receiving waters	1. *Nearfield*: New outfall will have low and localized influence on nutrient concentrations in the nearfield (EPA 1988) 2. *Farfield*: Farfield nitrogen concentrations not likely to change (EPA 1988) 3. *Harbor*: Nitrogen concentrations in Boston Harbor would be lower after transfer of effluent offshore (EPA 1988) 4. Relocation should not significantly increase effluent derived dissolved inorganic nutrients but may moderately increase particulate under primary treatment (NMFS 1993)
Plume dilution/Nutrient transport	1. Total nitrogen concentrations would be within natural variability of the system at plume dilutions >200:1 (EPA 1993) 2. Farfield phytoplankton communities will be subjected to nitrogen concentrations that are similar to those experienced under existing conditions (NMFS 1993)
Area of enrichment/area of impact	1. Area of plume influence under primary treatment was 85 km^2 based on 2-D transport modeling (EPA 1993) 2. Enriched area was ~4 km^2 under secondary discharge and viewed as a changed condition without excess growth of phytoplankton (EPA 1988)
Phytoplankton biomass changes	1. Marginal decreases in phytoplankton biomass in outer harbor and coast to Situate
Phytoplankton species response	1. Small localized nutrient inputs were not likely to cause widespread changes in phytoplankton species, abundance, or productivity (EPA 1988) 2. Species shifts were not expected
Nuisance algal species response and the outfall as an attraction for endangered species or prey of endangered species	1. Low total nutrients not likely to alter the existence and frequency of the occurrence (EPA 1988, 1993) 2. Proposed discharge will produce conditions in Massachusetts Bay similar to those from existing outfall (NMFS 1993) 3. The potential for increased red tide toxicity is small (NMFS 1993)
Zooplankton Species response	1. Small localized nutrient inputs are not likely to cause widespread changes in zooplankton species, abundance, or productivity (EPA 1988)
Dissolved Oxygen suppression	1. Worst case scenario (during water column stratification) indicated no more than a 0.1 mg DO/L suppression under secondary treatment (EPA 1988) 2. Suppression was within existing 6-8 mg DO/L range in Massachusetts Bay

2.2 Summary of findings from the baseline monitoring and other evaluations

2.2.1 Nutrient loading

The MWRA continually monitors nutrient concentrations in effluent from their treatment facilities. These data are used to calculate the annual loading of nutrient from the MWRA effluent, which are reported to regulators and public annually. Based on this data, nitrogen loading to Massachusetts Bay from the MWRA effluents decreased 12% from 1996 to 1998-99 (Table 2-3). The decrease is due to the advent of one bank of secondary treatment in August of 1997 and a second battery in February 1998. The third and final battery is scheduled for completion in March 2000. In addition, treatment of Nut Island sewage flow on Deer Island began in July of 1998. Thus, 1999 provides the first year of data under full secondary treatment of sewage at Deer Island.

Table 2-3. Summary of the annual loading of nitrogen to Boston Harbor from the MWRA treatment plant effluent compared to loading assumed in previous reports.

	Estimated loadings (mT/yr)			Measured loading (mT/yr)		
	SEIS[1]	BEM[2]		1996	1998	1999
Type of treatment	Primary plus Secondary	Primary	Secondary	Primary	1-2 Batteries of secondary	2 Batteries of secondary
Total Nitrogen	12,300	11,120	8,148	12,727	10,834	11,169
NH_4	N/A	6,028	6,150	6,610	8,135	8,299
$NO_2 + NO_3$	N/A	333	461	575	344	489

[1] EPA 1988

[2] HydroQual and Normandeau 1995

The 1998-99 annual nitrogen load is ~11%less than the load the used in SEIS (EPA 1988), Biological Assessment (EPA 1993), and Biological Opinion (NMFS 1993) to determine that no substantive impacts are expected from the relocation of the outfall. These loadings are, however, 35% higher than the loading assumed by the Bays Eutrophication Model to compare the relative effects of outfall relocation and level of effluent treatment. Recently the BEM was rerun to explore the effects of increased loading, and the results can be found in the sensitivity tests of Section 3.1.

Primary treatment and secondary treatment processes typically each remove about 15% of the influent nitrogen. Nitrogen removal by secondary treatment is evident in Table 2-3 as a reduction in the annual nitrogen loading between 1996 and 1998-99 (all other factors are assumed equal). The nitrogen loading is expected to further decrease when the third battery of secondary treatment is on line and more of the plant flow, especially during peak flow conditions, can be treated to the secondary level. Secondary treatment removes particulate material by settling (sludge formation) and also by mineralization to dissolved inorganic nitrogen. The proportion of ammonia in the effluent consequently has changed from about 50% to approximately 75% (Table 2-3). By design, little of this is nitrified in this treatment plant.

The loading estimates in the various environmental assessments for the outfall further assumed that all of the nutrients discharged at the Deer Island outfall would enter Massachusetts Bay. Recent studies (Kelly 1997, 1998) have documented that 10 to 15% of the nutrients discharged at Deer Island are retained within Boston Harbor. Therefore, most of the nutrients discharged by the MWRA at Deer Island are transported out of the Harbor and are already entering the Massachusetts Bay system. Thus, relocation would increase the nitrogen load to the Bay by 10-15%. This increase would counteract the decrease caused by the added efficiency of the nitrogen removal at the treatment plant (Sung and Higgins 1998; Butler *et al.* 1997; Hunt *et al.* 1995). Therefore, only very small changes in the nutrient loading to the Bays from the Deer Island treatment facility will result from the relocation.

Questions have also been raised about the relative contribution of nitrogen from the MWRA effluent compared to boundary inputs to the Massachusetts Bays system. Becker (1992) briefly examined this aspect of the nutrient mass balance for Massachusetts Bay and found that the nitrogen input to the Bays was likely dominated by the boundary inputs. In response to the question, this concept was further developed through the calibrated BEM model (HydroQual and Normandeau 1995, HydroQual 2000). A mass balance developed using the BEM (see Section 3.2 of this report for details) shows that the Deer Island effluent contributes a small fraction (~3%) of the total nitrogen entering the system.

2.2.2 Nutrient levels in the receiving waters

Baseline measurements in the water column of Massachusetts and Cape Cod Bays have extensively documented spatial and temporal variability in nutrient concentrations and plankton responses (Kelly and Turner 1995a; Kelly and Turner 1995b; Murray *et al.* 1997; Cibik *et al.* 1998a, 1998b; Lemieux *et al.* 1998; Libby *et al.* 1999). Strong seasonal differences are evident in the data as are seasonally dependent surface to bottom gradients in the water column. The baseline-monitoring program has also documented strong horizontal gradients emanating from Boston Harbor (Kelly *et al.* 1996) into the western portion of the nearfield (i.e. extending toward the site of the future outfall). Model computations of the nutrient distribution (HydroQual 2000 and Section 3.1 of this report) clearly show that increases in nutrient concentrations above the background variability will be confined to the nearfield areas. Moreover, the most recent modeling shows that nitrogen levels in Boston Harbor will decrease with the transfer of the effluent discharge offshore. However, the elevated nutrient levels in the coastal region (Boston Harbor southward towards Plymouth) will be relatively unchanged with the transfer (see Section 3.1 for detail). Thus, the most extensive water quality modeling to date for Massachusetts Bay predicts that relatively little change will occur in the spatial pattern of nutrient concentration when the effluent is moved offshore.

The hydrodynamic model of Signell *et al.* (1996) provides further enlightenment with respect to the expected dilution of the MWRA effluent in Massachusetts Bay in response to the outfall relocation. These investigators developed a 3-dimensional hydrodynamic model of the Massachusetts Bay and Boston Harbor system with a relatively fine grid spacing of 1 km. The model has been found to reproduce the major hydrodynamic features of the Massachusetts Bays, especially in the western areas where the new outfall is located. These hydrodynamic features include the evolution of the seasonal pycnocline, mean flow pattern, and strength of the subtidal current fluctuations. As a result the model is well suited to study average dilution characteristics of the effluent at the new location. The model was used to develop projections of effluent plume dilution under various environmental conditions. Animations of the model outputs can be found at http://crusty.er.usgs.gov.

Representative concentration contours are presented in Figure 2-2and Figure 2-3. The figures show effluent concentration contours along a transect extending from Boston Harbor at the left through the new outfall in Massachusetts Bay and then towards Stellwagen Bank. The contours of effluent concentration are 0.125% effluent; the red color corresponds to > 2% effluent, orange to 1-2% effluent, yellow to 0.5-1% effluent, and white to <0.5% effluent. The first arrow along the top of each figure shows the location of the current outfall at Deer Island; the second arrow indicates the location of the new outfall. The upper panel in each figure represents the effluent concentration based on the current discharge location; the second panel represents the conditions for the new outfall. The model computations clearly demonstrate demonstrate the expected reduction in the relative amount of effluent that will be found in Boston Harbor. Under the stratified conditions represented in Figure 2-2 the plume is clearly confined to deeper waters below the pycnocline. The horizontal extent of the plume under average conditions is clearly limited to within a few kilometers of the diffuser. Similarly, the plume under unstratified conditions (Figure 2-3) is limited in horizontal extent, but does extend to the ocean surface.

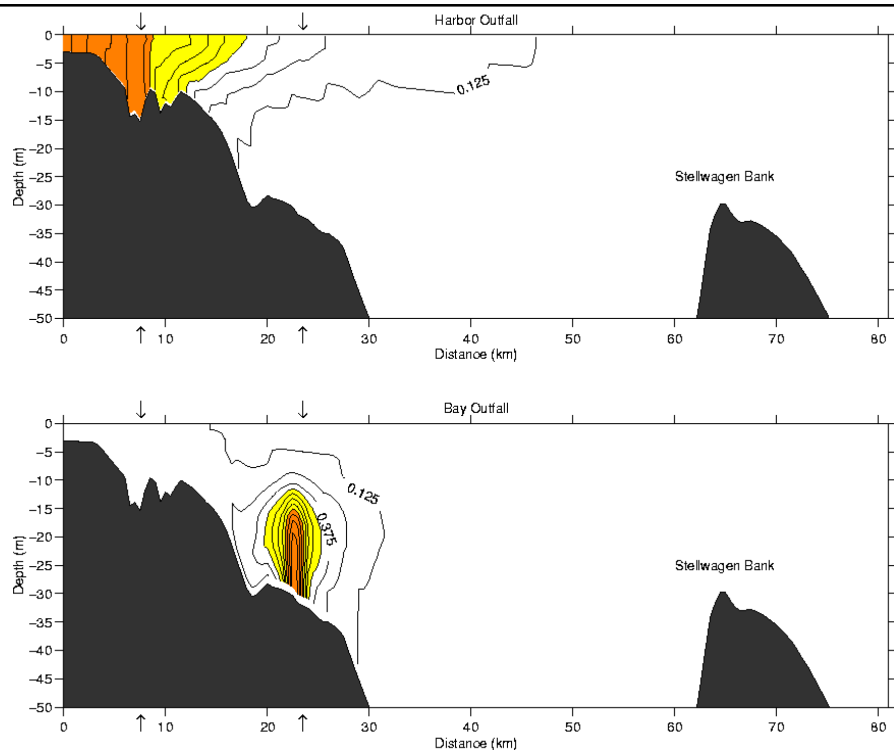

Figure 2-2. Distribution of the MWRA effluent concentration in Boston Harbor and Massachusetts Bay from the ECOMsi 3-D model under stratified conditions.

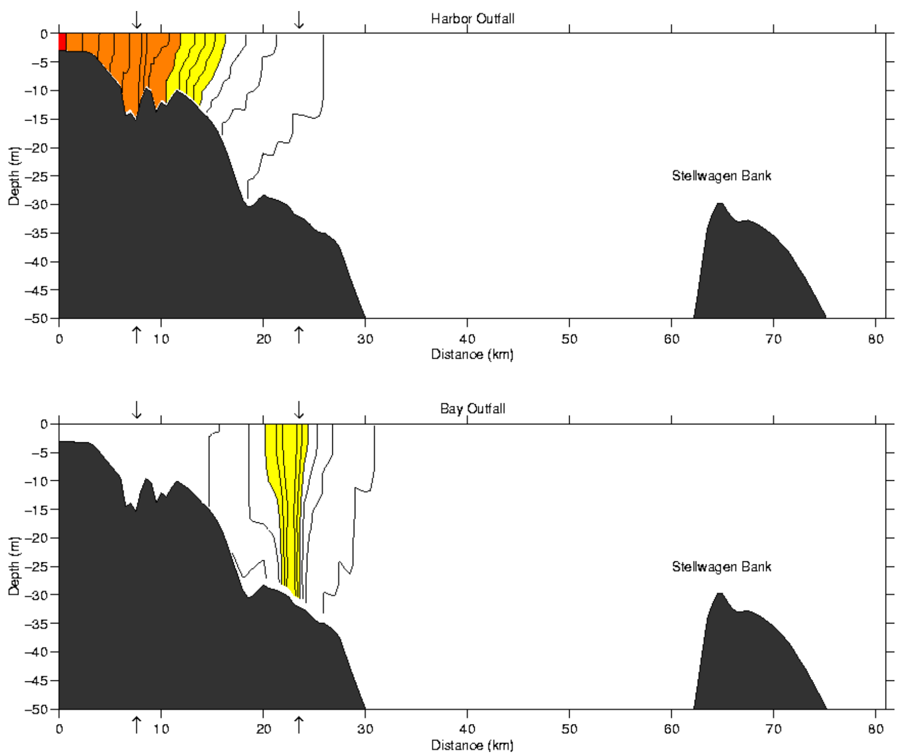

Figure 2-3. Distribution of the MWRA effluent concentration in Boston Harbor and Massachusetts Bay from the ECOMsi 3-D model under non-stratified conditions (from Signell *et al.* 1996).

2.2.3 Area of enrichment/area of impact

The 3-D model output described in Section 2.2.2 was also used to estimate the area in the Bays that would be under measurable influence from the discharge. This analysis indicated that the area that would be influenced is only 7 km^2, which is some 12 times less than predicted for primary treated effluents under the SEIS (EPA 1993). This is also <0.2percent of the combined area of Massachusetts and Cape Cod bays.

The BEM model was also used to evaluate the relative contribution of effluent to the total nitrogen entering the Massachusetts Bay system including that entering across the boundary with the Gulf of Maine. This is discussed in detail in Section 3.2 of this report. The results show that the relative contribution of the outfall is substantial close to the outfall, up to about 40%, but less than 10% in Cape Cod Bay. Thus, nitrogen levels in Cape Cod Bay are substantially more affected by transport from the boundary than by the MWRA effluent.

Additional evidence for reduced impact at the outfall is the dramatic reduction in the total solids discharged by the MWRA as a result of the facilities improvements (Figure 2-4). Annual discharge of total solids has decreased from 165 tons per day to just over 40 tons per day following removal of sludge discharge in 1991 and secondary treatment start up in 1997.

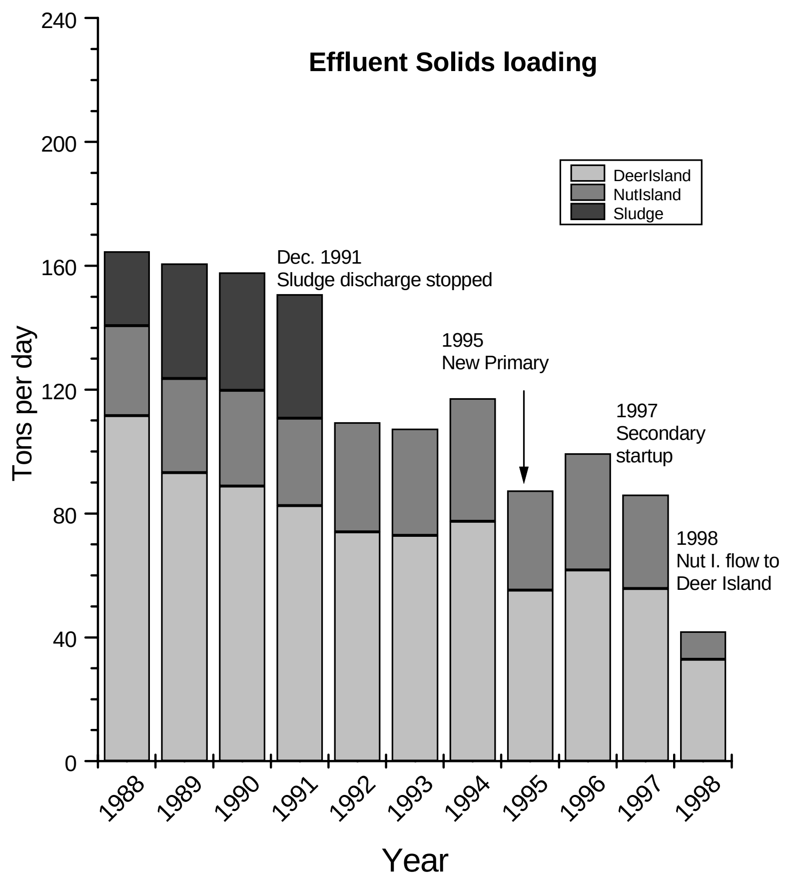

Figure 2-4. Average daily discharge of solids by the MWRA from 1988 through 1998.

2.2.4 Phytoplankton

Biomass changes: Chlorophyll, a measure of phytoplankton biomass, exhibits a dynamic range in Massachusetts Bay at several spatial and temporal scales. Seven years of baseline monitoring have shown that spatial distribution in Massachusetts and Cape Cod Bays exhibit seasonally dependent vertical gradients in the water column and a consistent offshore gradient. This offshore gradient emanates from the more productive waters of Boston Harbor and extends into western Massachusetts Bay including the western side of the nearfield. These gradients can change in intensity and spatial extent on a daily to weekly basis. A gradient also extends southward from the Harbor along the coast off Scituate MA and further southward towards Plymouth. The intensity of the gradient is seasonally dependent and responds to physical forcing functions such as those related to weather. The baseline data also show large variability in the seasonal average chlorophyll biomass (both within a year and within season across years), as well as variability in the annual average (see Libby *et al.* 1999 for the most recent compilation of baseline data).

Phytoplankton species response: As summarized in Cibik *et al.* (1998b), phytoplankton species in the Bays are also highly variable at weekly, monthly, seasonal, and annual scales. Spatial variability within season is driven by the interactions between nutrient availability and light. Typically the seasonal abundance of the numerically dominant phytoplankton species will vary by as much as three orders of magnitude. Variability within one group of phytoplankters (e.g., dinoflagellates, diatoms, microflagellates, etc) can be as much as one order of magnitude (a factor of 10). Dominance by various species groups also varies seasonally and can range from dominance by the small microflagellates in the summer to a mixture of diatoms in the winter and fall. Dinoflagellates display infrequent aperiodic blooms of generally low abundance. As observed for chlorophyll, Boston Harbor and the near coastal waters along the shore south of the Harbor tend to have the highest cell abundance, which is as expected given the generally higher nutrient levels in these regions.

Given the relatively small changes in nutrient distributions that are likely in Massachusetts Bay (see above and Section 3.1) following the transfer of the effluent offshore, it is not expected that baywide changes in phytoplankton species abundance or community structure will occur after the outfall is operational. This expectation is further supported by the most recent modeling results (Section 3.2) that show major influence on the nitrogen concentrations from the effluent will be confined to the nearfield and that only about one-third of the nutrients in the nearfield will emanate from the MWRA outfall. Moreover, the fact that the nutrient field in the farfield areas is driven more by the inflow of nutrients at the boundary of the Massachusetts Bay, it is unlikely that substantive changes in the phytoplankton abundance and species composition would be caused by the outfall relocation.

The only caveat to this projection is the increase in ammonia in the secondary effluent relative to primary effluent even though the total nitrogen concentration decreases (Table 2-3). Because ammonia is more quickly consumed by phytoplankton than inorganic or organic nitrogen forms, this could lead to greater stimulation of growth closer to the outfall, but only during those periods when the effluent reaches the photic zone. Note that the increase in ammonia is greater than that assumed in early projection runs of the BEM, and the Model Evaluation Group may recommend that the assumed loads be revised for forthcoming model runs.

Regardless, the high natural variability in the natural abundance of the various plankton species makes detection of more than major shifts in plankton composition and abundance problematic. Such major shifts are not expected to be caused by the outfall relocation given the relatively small changes in the nutrient fields and trapping of the plume below the pycnocline during the late spring to early fall period (see Figure 2-2). If major shifts do occur, they are likely to result from factors and circumstances that are outside of the control of the MWRA.

2.2.5 Nuisance algal species

The MWRA monitoring program has documented that nuisance species can bloom in the waters of Massachusetts and Cape Cod Bays. Specifically, major blooms of *Phaeocystis pouchetii* were observed in the spring of 1992 and spring of 1997 and an *Alexandrium tamarense* event was documented in 1993 (Anderson

1997) in coastal areas of Massachusetts Bay. Blooms of *Alexandrium* have not occurred in offshore waters since that time. Paralytic shellfish poisoning (PSP) outbreaks in shellfishing areas have not been recorded since 1993 in the Bays (Don Anderson, WHOI, personal communication June 1999). Cells of *Alexandrium tamarense* have been noted sporadically (31 of 1156 samples collected by the MWRA from 1992 through 1998) throughout the baseline period. Except for the event in 1993, when cells in one MWRA sample reached about 170 per liter (well below the level considered problematic for this species (D. Anderson, personal communication July 1999)), the *Alexandrium* abundance has been very low. When detected under the MWRA monitoring program, it has generally been < 5 cells per liter (26 of the 31 samples).

Alexandrium is known to arise from two sources in Massachusetts Bay. Locally, the species arise from germination of benthic cysts (this mechanism is generally confined nutrient rich coastal ponds). Once established, nutrient availability, coastal currents and winds regulate the bloom's distribution and abundance (see summary in Cibik *et al.* (1998b). Advection of the species into the Massachusetts system from the Gulf of Maine is the other mechanism leading to blooms in Massachusetts Bay. This transport mechanism is strongly influenced by wind conditions, which can either drive the surface waters of the Gulf of Maine into or away from Massachusetts Bay, depending on the direction and duration of the wind.

Note that other phytoplankton species of concern in the Bays such as *Pseudonitzia multiseries,* which can cause domoic acid poisoning, have not been recorded in bloom abundance nor has this diatom approached levels of concern (>500,000 cells/L). Similarly, low but persistent abundance of *Ceratium* spp, the dinoflagellate responsible for the anoxic bottom water event in the New York Bight in the 1970's, has been observed in the later years of the monitoring program (Libby *et al.* 1999). Observed levels are at least two orders of magnitude lower than associated with the anoxic event in the New York Bight.

Available information indicates that after the outfall is relocated little change in the species abundance, distribution, or community composition can be expected relative to that measured during the baseline period. This expectation is primarily based on the same rationale presented for biomass changes: 1) no net change in nutrient loading to the Bays from the MWRA outfall relocation, 2) only small local changes in the nutrient concentrations in the nearfield, 3) dominance of nutrient input by import into the Massachusetts Bay system across the boundary with the Gulf of Maine, 4) no change in farfield nutrient concentrations from the outfall relocation, and 5) isolation of the discharge plume from surface waters during the stratified period.

2.2.6 Zooplankton species response

As discussed for the phytoplankton, the MWRA baseline-monitoring program has documented temporal and spatial variability in zooplankton abundance and distribution in the Massachusetts Bays system. The data, including a retrospective analysis (Lemieux *et al.* 1998), indicate similarity of the nearfield and offshore farfield communities and a similarity of the Harbor and coastal communities. The data gathered and analyzed during the MWRA outfall monitoring baseline period do not change or alter the conclusions from the SEIS (EPA 1988) and federal endangered species assessments. That is, the expectation of small and localized changes in the nutrient concentrations and concomitant lack of change in phytoplankton communities will result in neither substantive nor readily detectable changes in the zooplankton community composition and species abundance.

Very high resolution mapping (10s of cm scale) of the spatial variability of physical and selected biological variables in Massachusetts Bay in March of 1998 (Davis and Gallager 1998) found characteristic spatial distributions for phytoplankton and zooplankton taxa and that the taxa had clear correlative affinities for different water types. The characteristic spatial (distance) correlation scales for most of the plankton taxa dropped off above 2 to 4 km whereas the spatial scales of the physical parameters, fluorescence, or particle fields (as measured by beam attenuation) were larger. The data are thought to indicate that there is some small-scale taxa-specific patchiness in the system. The data also indicate that colder fresher surface water from the Cape Ann area contributes to the formation of Cape Cod Bay water and tends to dilute the plankton in the northern and western parts of Massachusetts Bay. Lastly the data suggest that local heating in Cape Cod Bay may be important in initiating blooms in Cape Cod Bay in the spring. While much work remains, this high resolution mapping effort has begun to shed some light on the interrelationships between selected zooplankton and the physical properties of the waters of Cape Cod Bay.). Important to the issues of the present report is the observation that water derived from the boundaries of the Massachusetts Bay system substantially affects the plankton distributions and levels in the system.

Similar results were found in a March 1999 survey (Davis and Gallager 2000), although the route of water and plankton to Cape Cod Bay was found to be from offshore in northeast Massachusetts Bay. The study suggests that the spatial scale for changes in the distribution of planktonic taxa were <2 km and that correlative length scales for zooplankton were about 20 km. The latter allows statistical testing of zooplankton abundance between the nearfield and farfield as spatial independence of stations this far apart is demonstrated. This finding supports the results of a statistical treatment (Ellis *et al*. 2000) of the major offshore species (i.e., sum of the adult plus copepodite forms of *Calanus*, *Pseudocalanus*, *Centropages typicus*, and *Oithona*). This latter study indicates that there is no difference in the abundance of these zooplankton between the nearfield and farfield during the January to May period across the 1992 through 1999 baseline period.

Studies of zooplankton patches have suggested that threshold values on the order of 4,000 *Calanus*/m^3 are required for efficient right whale feeding (Clapham 1998). Anecdotal evidence suggests that right whales in Cape Cod feed in dense patches of zooplankton but are also known to feed in the area in the absence of these patches (Mayo as cited in Clapham 1998). In 1998, the MWRA increased the number of stations sampled for zooplankton in Cape Cod Bay in the winter from 2 to 4. The results have documented that a larger range in zooplankton abundance was detected in this system with more spatial sampling. Very high resolution sampling using a towed video plankton recorder (VPR) has also documented fine scale spatial processes that likely affect the distribution of nutrients, phytoplankton and zooplankton in the right whale feeding area of Cape Cod Bay (Davis and Gallager 1998, 2000). Factors causing the formation of these fine scale features and there importance to the whale food resources are as yet unclear, although correlative analysis to the areas the whales are often observed suggest they may be linked (Mayo in Clapham 1998).

The baseline data do not refute, but rather support the conclusions drawn in the outfall SEIS (EPA 1988) that the outfall would only have a limited local affect on the zooplankton in the greater Massachusetts Bay system. The information that the new outfall will not alter the nutrient regime in Cape Cod Bay suggests that forces larger than the outfall will continue to drive the zooplankton dynamics in this system (see Davis and Gallager 1998, 2000). More importantly, information on the importance of the physical processes relative to the distribution of the zooplankton is just beginning to be developed.

Lastly, the MWRA effluent will not affect physical processes such as regional stratification and the general circulation of Massachusetts Bay. This is most easily shown by the rapidity with which the plume dilutes under stratified and unstratified conditions (Sections 2.2.2).

2.2.7 Dissolved oxygen suppression

Very small (0.1 mg/L) suppression of dissolved oxygen in the deeper waters of Massachusetts Bay due to outfall relocation was modeled during the SEIS process. Moreover, measured DO levels in the deeper waters were found to range from 6 to 8 mg/L in the summer. The baseline-monitoring program has documented that this remains true. However, the baseline monitoring has identified that DO concentrations may decrease to 4 - 5 mg/L in some areas in the late summer (Kelly and Turner 1995b). This level is not considered harmful to marine organisms (EPA 1999) and does not approach hypoxic conditions, although it falls below the Massachusetts State marine water quality standard of 6.0 mg/L. The monitoring program has also documented a systematic rate of decrease in the bottom water oxygen levels once water column stratification sets up and water temperatures increase in the June to July time frame.

After the SEIS was completed, more sophisticated modeling using the BEM (HydroQual and Normandeau 1995, HydroQual 2000) was developed. It reproduces the major DO trends in the system for 1992 through 1994, and projects improved DO in Boston Harbor and small localized DO increases in the summer in the nearfield following outfall relocation. This same model does not indicate changes in DO in other farfield areas such as Stellwagen Basin and Cape Cod Bay. As a sensitivity test the nutrient loading at the outfall site was assumed to have doubled, with the result that a small decrease in DO was computed in the farfield areas. Such an increase would be analogous to a doubling of the population in the area served by the MWRA treatment plant, which incidentally is unlikely given current local population trends.

As in the sections above, the baseline monitoring and modeling data point to a system that has a well described seasonal DO response. This response appears to be driven regionally by the nutrient input at the boundaries and which is slightly exacerbated near the new outfall by the effluent input. The best available water quality model indicates improvements in the Harbor and only small DO changes in the nearfield when the MWRA secondary treated effluent is transferred from the mouth of Boston Harbor farther into Massachusetts Bay.

2.3 Comparison and conclusions

The results summarized in Section 2.2 clearly indicate that the assumptions and conclusion drawn in the SEIS (EPA 1988) and subsequently revisited in the Biological Assessment (EPA 1993) and Biological Opinion (NMFS 1993) remain valid today. Irrespective of the topic examined, the recent studies and baseline data indicate that the relocation of the outfall will have little if any impact in areas outside of the immediate nearfield. In fact, the assumptions and assessments within the SEIS have been shown in this current reassessment to be conservative in the areas of nutrient loading, nutrient concentrations expected in the nearfield receiving waters, the area of enrichment/area of impact that could be detected, phytoplankton biomass changes, and dissolved oxygen suppression.

Evidence presented in the following sections also indicates that the nutrient transport into the Bays from the offshore boundary areas will exert greater influence on farfield nutrient distributions than will the outfall. By extrapolation of our basic understanding of the ecological interactions between and among nutrients, phytoplankton and zooplankton, the boundary is thus more likely to influence the overall plankton species composition and abundance in the Bays and especially the farfield areas such as Cape Cod Bay and Stellwagen Bank.

The results from the monitoring program and model computations are not as easily extrapolated to the response in plankton species, especially nuisance algae. However, several lines of reasoning lead to the conclusion that the outfall relocation will not exert any more impact than it presently does on the Massachusetts Bay system. These include the information that nutrient loading at the new outfall will not be different than is presently discharged at the mouth of Boston Harbor and that this discharge is already exported to Massachusetts Bay, that more refined modeling projection indicate the area of impact will be smaller, and that the discharge will be trapped below the pycnocline in the stratified period. If any effects occur they, as stated in the EPA SEIS, will be localized and of limited magnitude.

From the above information and discussion, the question "will environmental conditions worsen as a result of the outfall relocation?" is answered in the negative. Because no changes are likely, the question "is such change likely to harm whales?" must also be answered as no. Regardless, it is instructive to pursue whether application of food web models could provide incremental or substantial improvement in our ability to predict the occurrence of right whales in Massachusetts Bay and whether the outfall will have impact on the occurrence. To address these issues, the results of the BEM sensitivity modeling conducted as part of this reassessment is first presented and considered. A summary of the conceptual food web model developed by Kelly *et al.* (1998) is then presented and discussed in light of the findings presented in the above sections. This section is followed by a review of food web modeling approaches.

3. RECENT MODEL RESULTS

One utility of a calibrated mathematical model of a water body is the ability to project the impacts of possible management scenarios on future water quality. In the case of the Massachusetts Bays system, such a model was developed with funding provided by the MWRA. The model was built on a state-of-the-art model having wide application to other regions. This water quality model, known as the Bays Eutrophication Model (BEM), was initially calibrated against two extensive data sets. One of the data sets used for model calibration included data collected as part of the first year (1992) of operation of the Harbor Outfall Monitoring Program (HOM). The details of the model framework and its calibration have been previously reported (HydroQual and Normandeau 1995). More recently the BEM was compared against an additional two years of data (1993-1994) from the ongoing HOM program (HydroQual 2000). The results of this effort indicate that the BEM captures the principal processes that interrelate primary production and dissolved oxygen to Bay-wide circulation, water column temperature and stratification, nutrients, and light. While the model does not reproduce species-specific phytoplankton blooms that occasionally occur within the Massachusetts Bays system, for example, the *Asterionellopsis glacialis* bloom that occurred in the fall of 1993, the BEM does reproduce a number of the spatial and temporal features of phytoplankton biomass and primary productivity observed in Boston Harbor, Massachusetts Bay, and Cape Cod Bay.

Accordingly, the model efficacy was considered sufficient to use the model in an exploratory analysis of the sensitivity of nitrogen loading on key ecological measures of the Massachusetts Bay ecosystem. This exploration was conducted as one component of the (see Section 1) development of a scope of work for a food web model for the Massachusetts Bays system (Hunt *et al.* 1999).

3.1 Sensitivity modeling

For this analysis, the BEM was used to perform a series of sensitivity runs wherein the magnitude of nutrient loading from the MWRA wastewater treatment facilities on Deer Island was manipulated. Combinations of three levels of nutrient loading at either of two locations were modeled. The two locations were (1) the present points of discharge to the waters of Boston Harbor from the Nut Island and the Deer Island Treatment Plants, and (2) the new diffuser location in northwestern Massachusetts Bay. These model runs were conducted to elicit the relative effect of major changes in the amount of nutrient discharged by the MWRA on key components of the Massachusetts Bay ecosystem. These components included total nitrogen (TN), dissolved inorganic nitrogen (DIN), phytoplankton biomass (chlorophyll), and dissolved oxygen (DO) in both Massachusetts and Cape Cod Bays. The responses were considered by indirect inference as representing the possible impacts on food web dynamics of the system, and which therefore may have consequences for important prey species of endangered species in the Massachusetts Bays ecosystem.

The series of sensitivity runs performed as part of this analysis used the 1992 calibration (HydroQual and Normandeau 1995) as the base condition or “1X” loading (assuming 11,150 mtons of nitrogen per yr). The response in the Bays was compared to the response from the 1X input to determine the sensitivity of the system to increased and decreased nutrient loading and the two discharge locations. In 1992 the wastewater treatment facilities operated by the MWRA provided primary treatment only. A total of six sensitivity runs were conducted. Note that the volume of the water discharged as effluent was not changed during these runs. The sensitivity runs included:

(1) current outfall location, zero organic carbon and nutrients (total nitrogen, total phosphorus, and dissolved inorganic silica) in the MWRA effluent,
(2) current outfall location, current (1992 calibration) carbon and nutrients in the MWRA effluent,
(3) current outfall location, twice the current (1992 calibration) carbon and nutrients in the MWRA effluent,
(4) future outfall location, zero organic carbon and nutrients in the MWRA effluent,

(5) future outfall location, current (1992 calibration) carbon and nutrients in the MWRA effluent, and
(6) future outfall location, twice the current (1992 calibration) carbon and nutrients in the MWRA effluent.

All other carbon and nutrient inputs, e.g., non-MWRA treatment facilities, combined sewer overflows, storm sewer, riverine, groundwater, and atmospheric deposition, and environmental conditions, e.g., temperature, light, boundary conditions, extinction coefficients, etc., were the same as for the 1992 calibration. The only exceptions were for the future outfall location (FOL) sensitivity runs, wherein the hydrodynamic model was rerun with the freshwater associated with the current outfall locations (COL) at Nut and Deer Islands relocated to the future outfall location.

Computations from each of the sensitivity runs were compared against the computations from the base condition through spatial mapping of concentration intervals. The figures discussed below represent five-day averages of model results. The first set of sensitivity comparisons is presented for surface phytoplankton biomass, as indicated by chlorophyll-a (chl-a), for mid-April (Figure 3-1). This date corresponds to the time period during which vertical stratification begins to occur within the water column. Therefore, this date also corresponds to the last period of time (until the turnover of the water column in mid- to late-October) wherein nutrients discharged at the future outfall location reach the surface waters of the Bay. In Massachusetts Bay, after mid-April, the water column of the Bay becomes vertically stratified due to differential warming of surface waters by energy from sunlight. Nutrients discharged via the diffusers at the future outfall site are projected to remain trapped below the pycnocline when the water column is stratified, thus would be unavailable for uptake by phytoplankton in the surface waters of the Bay.

As seen in Figure 3-1, the nutrients discharged by the MWRA, at their present discharge locations at Nut Island and Deer Island, do have an impact on chlorophyll levels in Boston Harbor and the nearshore regions of northwestern Massachusetts Bay. Relative to the zero nutrient (0X) sensitivity run the 1X nutrient loading under conditions of primary treatment from the MWRA treatment facilities result in (Figure 3-1a) chlorophyll concentrations that are 1.5-2 μg/L greater in Boston Harbor and alongshore as far south as Scituate. Concentrations are 0.5-1 μg/L greater under the 1X loading as far south as Gurnet Point, near Plymouth (Figure 3-1b). Nutrients from the MWRA discharge, however, do not appear to increase the concentrations of chlorophyll-a in Cape Cod Bay in moving from 0X to 1X loading. In contrast, doubling the MWRA nutrient loading is projected to stimulate additional phytoplankton growth including the southwest portion of Cape Cod Bay (Figure 3-1c). However, the increases in chlorophyll-a concentrations in southwest Cape Cod Bay are relatively small, 0.5-1 μg/L. Perhaps more significant are the sensitivity projections that show while the relocation of the MWRA outfall to Massachusetts Bay reduces chlorophyll-a concentrations in Boston Harbor, there is little or no impact on the levels of chlorophyll-a that develop in Massachusetts Bay and Cape Cod Bay (Figure 3-1d-f).

Projection computations for Total Nitrogen (Figure 3-2) show similar spatial profiles as were observed for chlorophyll. The elimination of nitrogen associated with the MWRA effluent discharge at the current outfall location significantly reduced the concentrations of TN in Boston Harbor (Figure 3-2a). Residual concentrations of TN are due to inputs from CSO and storm sewers draining to the harbor, as well as inputs from the Charles River. The addition of the existing (1992) TN loads at the current outfall location increases the concentrations of TN in the Harbor to more than 0.4 mg N/L and increases the concentrations of TN to approximately 0.25 mg N/L to Gurnet Point, near Plymouth Harbor (Figure 3-2b). Doubling the TN in the MWRA effluent results in further increases in TN in the harbor and the along shore region of northwest Massachusetts Bay to between 0.3 and 0.35 mg N/L as far south as Humarock (Figure 3-2c). The concentrations of TN are also computed to increase by approximately 0.05 mg N/L in the southern and

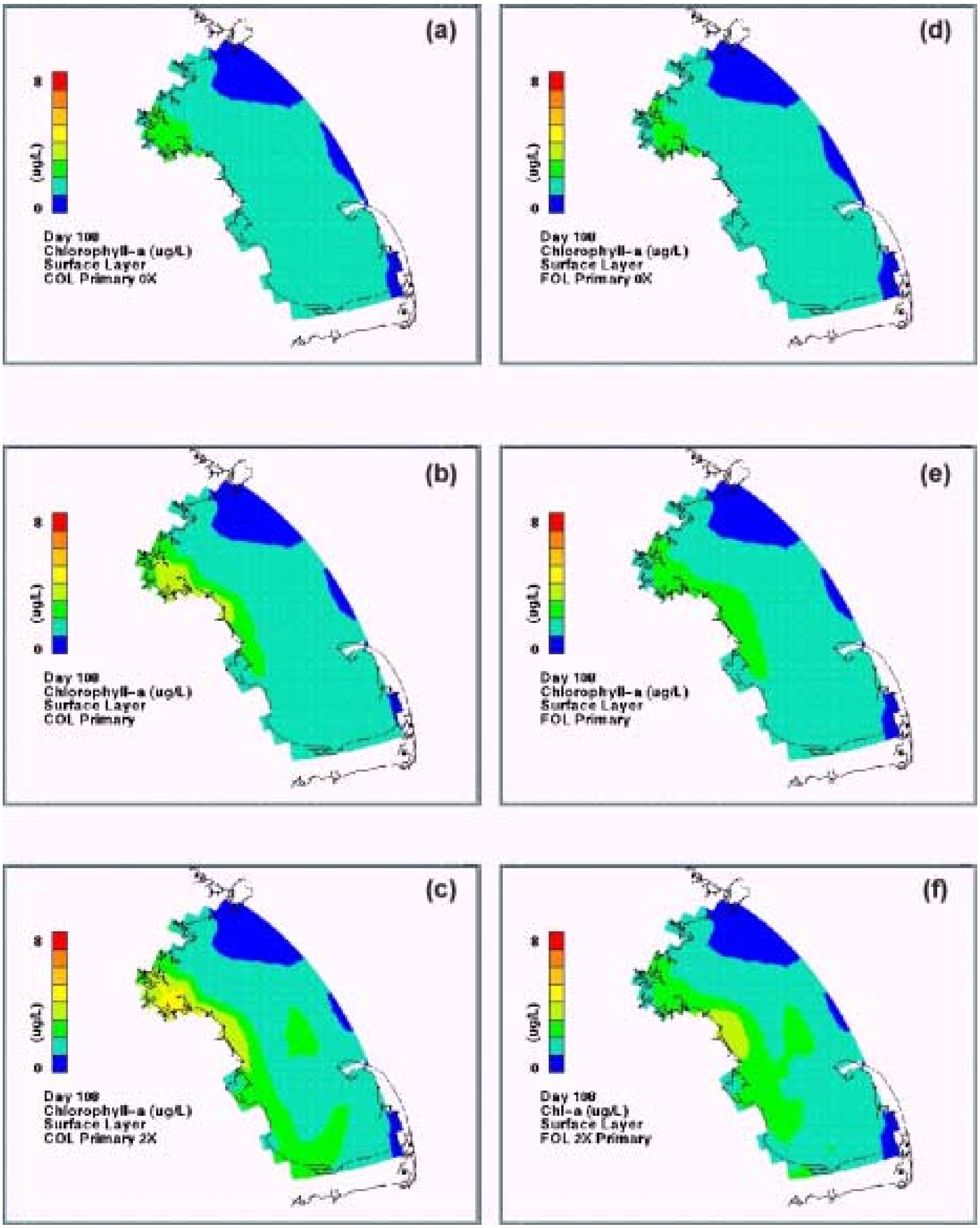

Figure 3-1. Spatial representation of modeled late April chlorophyll in surface waters of Massachusetts Bay in response to 0X, 1X, and 2X nutrient loading and shift in discharge location from the current location (COL) and future outfall locations (FOL).

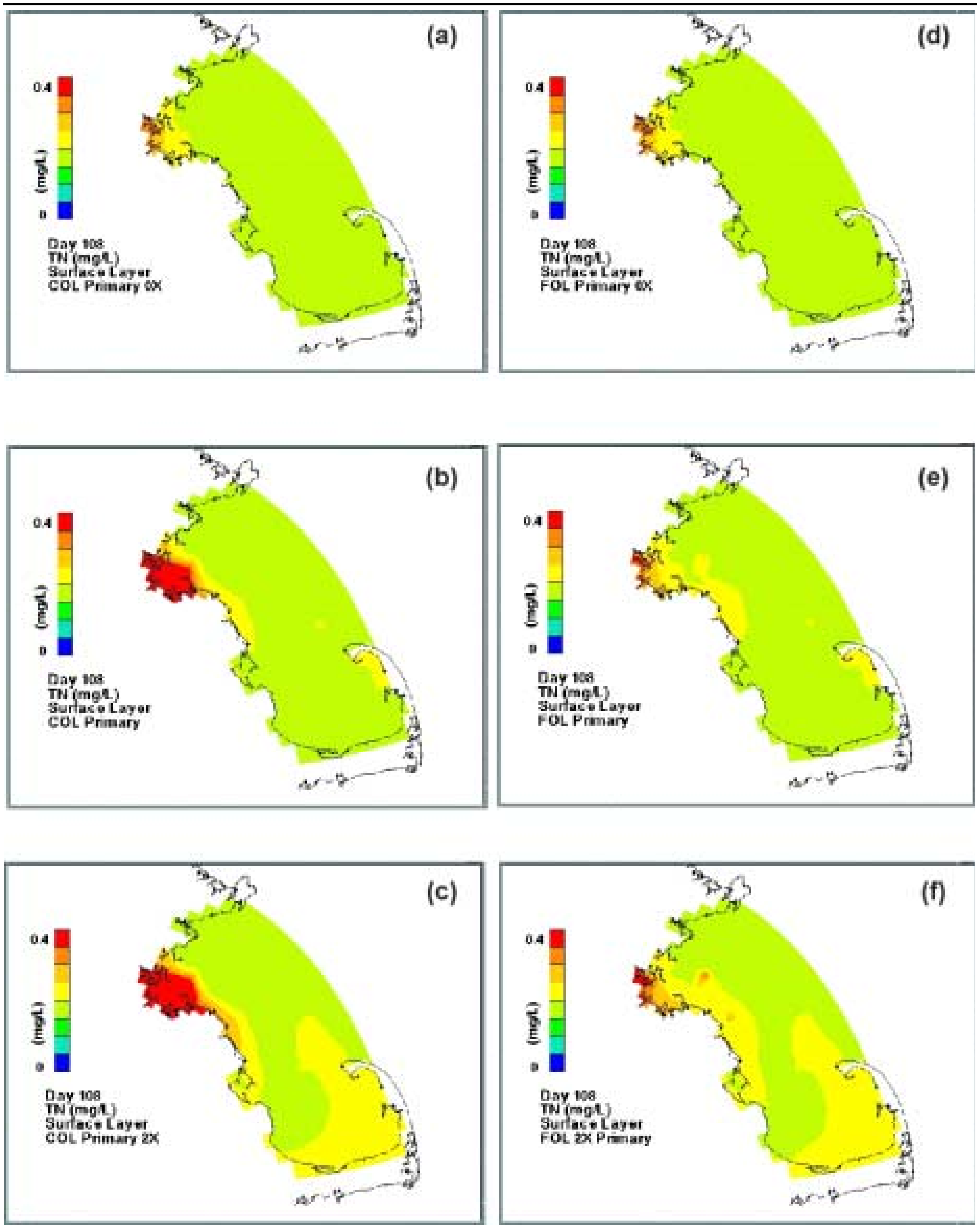

Figure 3-2. Spatial representation of modeled late April total nitrogen in surface waters of Massachusetts Bay in response to 0X, 1X, and 2X nutrient loading and shift in discharge location from the current location (COL) and future outfall locations (FOL).

southeastern portions of Cape Cod Bay. It is interesting to note, however, that concentrations of TN did not increase in southwest Cape Cod Bay, just to the south of Plymouth during this five-day period. Further examination of the model results show that during this period a wedge of water with low dissolved organic nitrogen (DON) concentrations pushed in between Massachusetts Bay and Cape Cod Bay causing the discontinuity in the TN contours. DON comprises approximately half of the TN concentration during this time. As was also observed for the chlorophyll projection results, the movement of the MWRA effluent to the future outfall location site did not result in significantly different spatial profiles of TN (Figure 3-2e-f). The only significant differences between the COL and FOL results are that concentrations of TN are reduced in Boston Harbor and are increased slightly more offshore in the western portions of Massachusetts Bay, as well as in the vicinity of the FOL. Computed concentrations of TN are virtually the same in Cape Cod Bay for both the FOL base condition Figure 3-2e) and the load doubling or 2X run (Figure 3-2f), as compared to the COL base condition (Figure 3-2b) and the 2X (Figure 3-2c) runs.

Similar results were also computed by BEM for DIN (Figure 3-3). In general, increasing the MWRA nutrients load at the COL increases DIN concentrations in Boston Harbor and the along shore areas of northwestern Massachusetts Bay (Figure 3-3a-c). Doubling the MWRA nutrient loading results in an increase in DIN concentrations in the southeastern and eastern portions of Cape Cod Bay between Nobscusset Point and Provincetown (Figure 3-3c). Moving the outfall to the FOL location results in an increase in surface DIN in the immediate vicinity of the outfall (Figure 3-3e-f). In addition, the effluent plume is moved further offshore. The concentrations of DIN in Cape Cod Bay for the baseline run scenarios (Figure 3-3b, e) and the 2X scenarios (Figure 3-3c, f) appear to be similar. One key feature of the model computations that can be observed in Figure 3-3 is the relatively large influx of DIN from the Gulf of Maine boundary. This influx of nitrogen is a significant source of nitrogen to the Massachusetts Bays system, as will be discussed subsequently.

The discharge of organic carbon and nutrients by the MWRA also influences the concentrations of dissolved oxygen in the bottom waters of the Harbor and the Bays. Model computations for late October indicate that increasing inputs of organic carbon and nutrients at the COL lead to decreasing levels of bottom water dissolved oxygen both in the inner portions of Boston Harbor, the immediate area just outside of Boston Harbor, and the central and southwestern portions of Cape Cod Bay (Figure 3-4a-c). Changes in bottom water dissolved oxygen in Boston Harbor and its immediate vicinity are due to organic carbon oxidation and nutrient driven algal respiration and sediment oxygen demand. Changes in Cape Cod Bay are principally due to algal respiration and sediment oxygen demand, a small component of which may be associated with nutrients from the MWRA discharge (see below). The principal differences between the COL and the FOL locations are increases in dissolved oxygen levels in Boston Harbor that are associated with moving the outfall to Massachusetts Bay and a slight decrease in the immediate vicinity of the future outfall location under the FOL 2X projection (Figure 3-4f). Note that the 1X projection for the FOL does not demonstrate substantial changes in the spatial extent of the bottom water DO in Massachusetts Bay relative to the COL.

In summary, it appears from model computations that moving the MWRA wastewater effluent into Massachusetts Bay from Boston Harbor should not have a significant impact on the nutrient fields or water quality of Massachusetts Bay or Cape Cod Bay. Therefore, this result also suggests that the outfall relocation should have little or no impact on the prey food supply for endangered species of the Massachusetts Bays system.

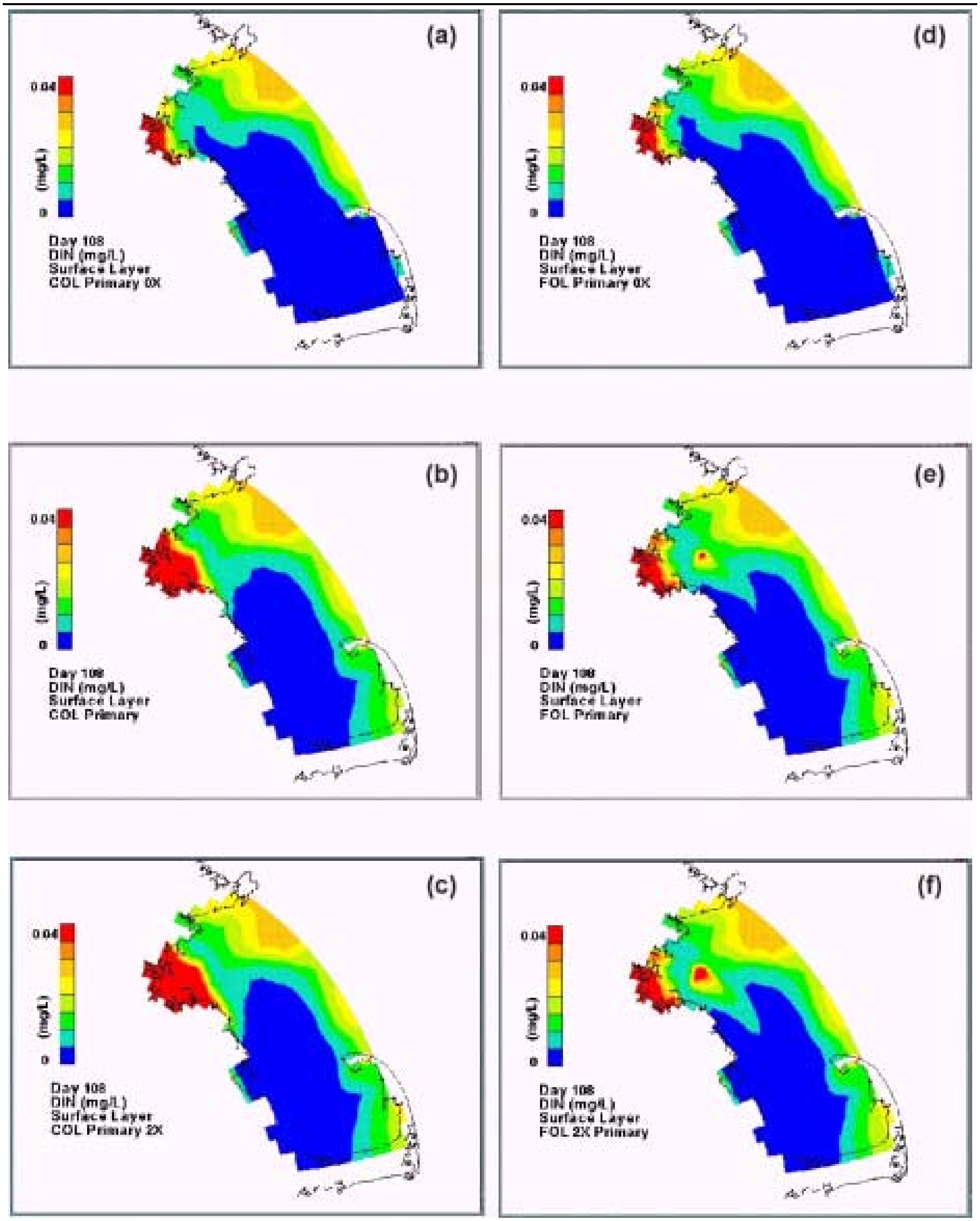

Figure 3-3. Spatial representation of modeled late April total dissolved inorganic nitrogen in surface waters of Massachusetts Bay in response to 0X, 1X, and 2X nutrient loading and shift in discharge location from the current location (COL) and future outfall locations (FOL).

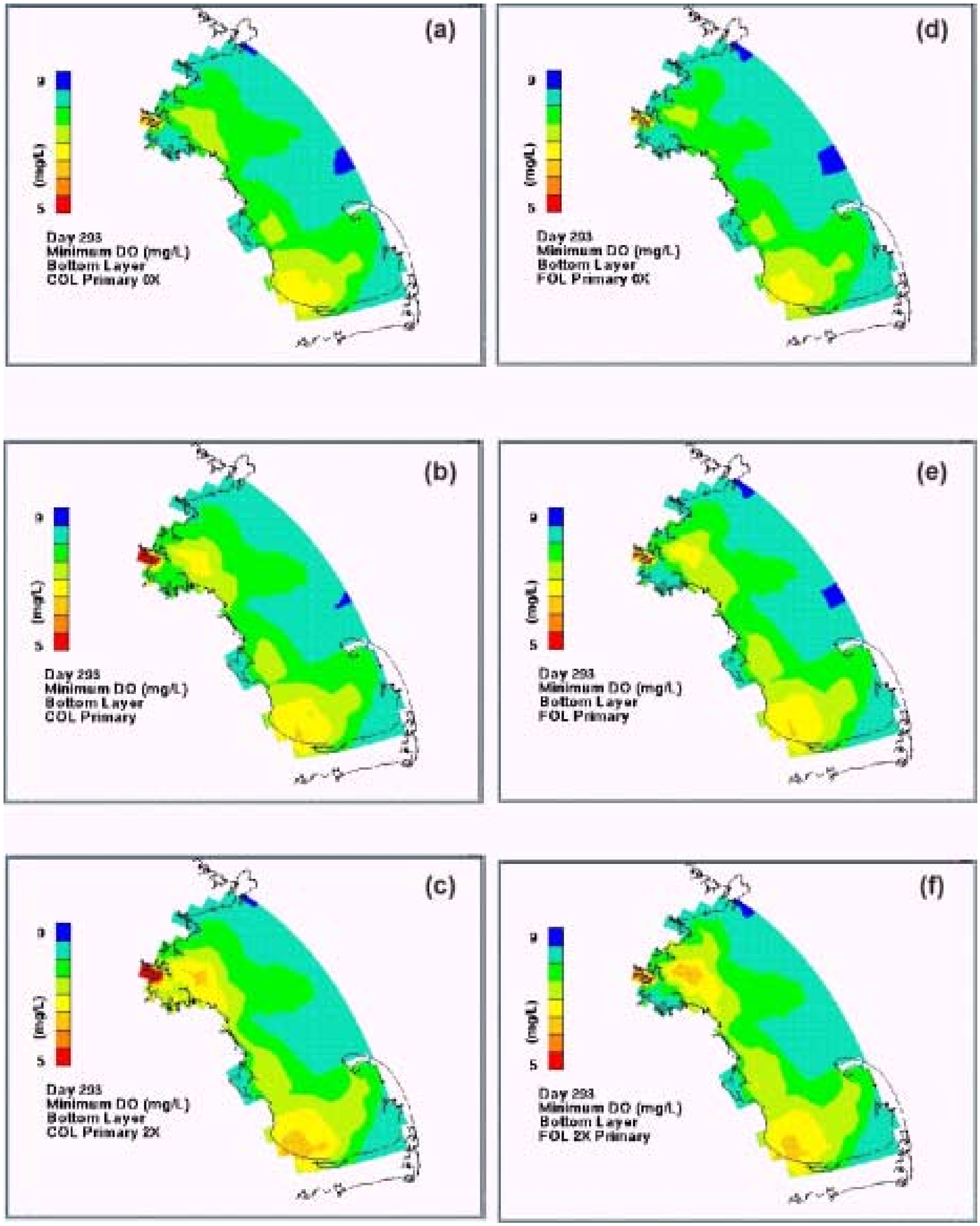

Figure 3-4. Spatial representation of modeled late October dissolved oxygen in bottom waters of Massachusetts Bay in response to 0X, 1X, and 2X nutrient loading and shift in discharge location from the current location (COL) and future outfall locations (FOL).

3.2 Modeling relative to the nutrient status of Massachusetts and Cape Cod Bays

From the sensitivity analysis performed using the BEM (Section 3.1) it was demonstrated that the relocation of the MWRA's effluent from Boston Harbor to northwestern Massachusetts Bay would have little impact on the water quality of Massachusetts Bay and Cape Cod Bay. Based on model projections using the BEM, it was also observed that Massachusetts Bay and Cape Cod Bay are more sensitive to the magnitude of nutrient inputs, rather than the absolute location of the discharge. Moreover, model computations also indicated that a substantial change (i.e., a doubling) in nutrient loading would be required before perceivable changes in phytoplankton biomass would occur in portions of Cape Cod Bay. To understand the reasons for these results from the model computations, additional analyses were performed using the water quality model. These included mass balance calculations based on the model and spatial and temporal aspects of MWRA nitrogen inputs versus nitrogen inputs from the Gulf of Maine. In the first instance, a mass balance for all sources and sinks for nitrogen over an annual cycle was performed for the entire Massachusetts Bays system (Figure 3-5).

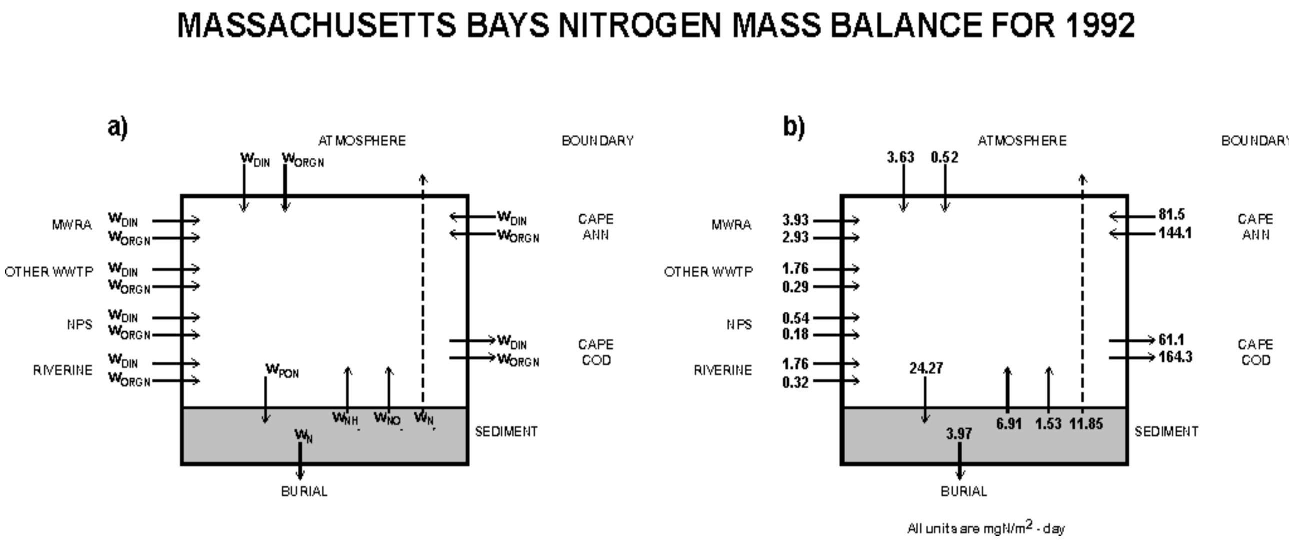

Figure 3-5. Free body diagram showing all of the sources and sinks in mg Nm^{-2}d^{-1} for total nitrogen for the Massachusetts Bays system.

Nitrogen inputs to the system included: the MWRA effluent; discharges from other waste water treatment plants (WWTP); nonpoint source (NPS) inputs, including combined sewer overflows, storm sewers and groundwater; riverine sources (Charles River, etc.), atmospheric deposition directly to the surface waters of the Bays; fluxes of ammonium and nitrate nitrogen from the bottom sediments of the Bays; and the influx of nitrogen from the Gulf of Maine along the northern boundary of the model. Loss terms associated with nitrogen included: burial of refractory organic nitrogen to the deeper portions of the bottom sediments of the Bays; efflux of nitrogen gas (resulting from nitrification and denitrification in the bottom sediments) to the atmosphere; and the advection of nitrogen from the Bays to the Gulf of Maine along the southern boundary of the model at the northern tip of Cape Cod. Using the results from the 1992 base case conditions, the BEM computes that over an annual cycle only 3 percent of the total nitrogen entering the Massachusetts Bays system is derived from the MWRA inputs. The model also indicates that approximately 93 percent of the nitrogen entering the Massachusetts Bays system is associated with inflowing waters from the Gulf of Maine. These computations are consistent with the observation that the majority of waters within the Massachusetts Bays system appear to be more representative of oligotrophic conditions than eutrophic conditions. Only Boston Harbor and perhaps the northwestern portions of Massachusetts Bay off Boston Harbor appear to be relatively enriched with respect to nutrients and phytoplankton biomass.

While the mass balance analysis presented above summarizes nutrient inputs to the entire Massachusetts Bays system over an annual cycle, it does not provide insight into the spatial and temporal aspects of the MWRA nitrogen inputs versus those nitrogen inputs from the Gulf of Maine. Therefore, the BEM was used once more to gain additional insights into the spatial and temporal distribution of nitrogen from these two sources. To perform this analysis, the water quality model was run treating nitrogen as a conservative variable tracer (i.e., the biology in the model was turned off). The model was run for the outfall location at Deer Island (COL) and the new outfall site (FOL). One run in each set was completed using the combined loading from the boundary and the MWRA outfalls (no other sources were allowed), once using only the observed 1992 TN loading from the MWRA, and once using only the 1992 boundary TN concentrations used in the 1992 calibration. The assumed effluent and boundary concentrations are shown in Figure 3-6. The effluent load ranges from about 14 to 27 mg/l with slightly higher concentrations in the late spring to early winter. The boundary loading shows relatively constant input concentrations (0.3 mg/L) in the bottom waters but a clear seasonal loading in the surface waters. Summer time concentrations are as low as 0.1 mg/L the winter concentrations are between 0.25 and 0.3 mg/L.

Temporal distributions of total nitrogen concentrations from the COL runs are presented in Figure 3-7 for five model segments (stations) within the Massachusetts Bay water quality model domain. Three of these model segments were selected in the vicinity of Boston Harbor and two in Cape Cod Bay. The same station series for the future outfall site is shown in Figure 3-8. The first model segment is located just southeast of the current outfall at Deer Island and represents nearfield station N10 which is known to be influenced by the outflow of Boston Harbor water (Kelly 1997).

Comparison of the surface layer concentrations of TN at this location (Figure 3-7a) indicates that during the summer months when the boundary concentrations of TN are low, the MWRA discharge may contribute up to 50 percent of the TN in the immediate vicinity of Boston Harbor. However, this percentage decreases dramatically with distance from Boston Harbor. A model segment located in the vicinity of the future outfall site (represented by Station N21) shows that the MWRA contribution to TN in this portion of Massachusetts Bay may be, on average, only 30 percent (Figure 3-7b). Occasional maximum values of 40 percent are observed which corresponds to periods when the prevailing wind-driven circulation advects pulses of Boston Harbor water to this location. The spatial influence in Massachusetts Bay of the MWRA loading is diminished even further (Figure 3-7) with distance from Boston Harbor (represented by Station N04 at northeast corner of the nearfield area). Examination of the model computations for TN at two locations in Cape Cod Bay (Figure 3-7d-e) suggests that the boundary is far more important to the concentrations of TN in Cape Cod Bay than is the MWRA effluent. These model computations show that under the assumption that it is conservative, nitrogen entering the Massachusetts Bays system from the Gulf of Maine may contribute between 80 and 90 percent of the Cape Cod Bay TN that can be ascribed to the MWRA effluent plus the boundary input.

The same station series for the new outfall location (Figure 3-8) demonstrates two attributes of the relocation. The first is the substantial decrease in the amount of total nitrogen contributed by the outfall at locations closer to Boston Harbor and at Station N21 near the eastern end of the new diffuser. The second is the unchanged total nitrogen contribution from the outfall to Cape Cod Bay. The percent contribution from the outfall is consistently <10% as shown in Figure 3-9. Note that the percent contribution in the western Cape Cod Bay is slightly higher than in eastern Cape Cod Bay.

These model computations, together with the sensitivity analysis presented in Section 3.1, suggest that nutrients discharged by the MWRA both at the current outfall location and at the future outfall location have limited spatial effects on primary production in the Massachusetts Bays system. These computations also suggest that the discharge of nutrients by the MWRA at the future outfall site should have little or no impact on prey food sources favored by endangered species within the Massachusetts Bays system.

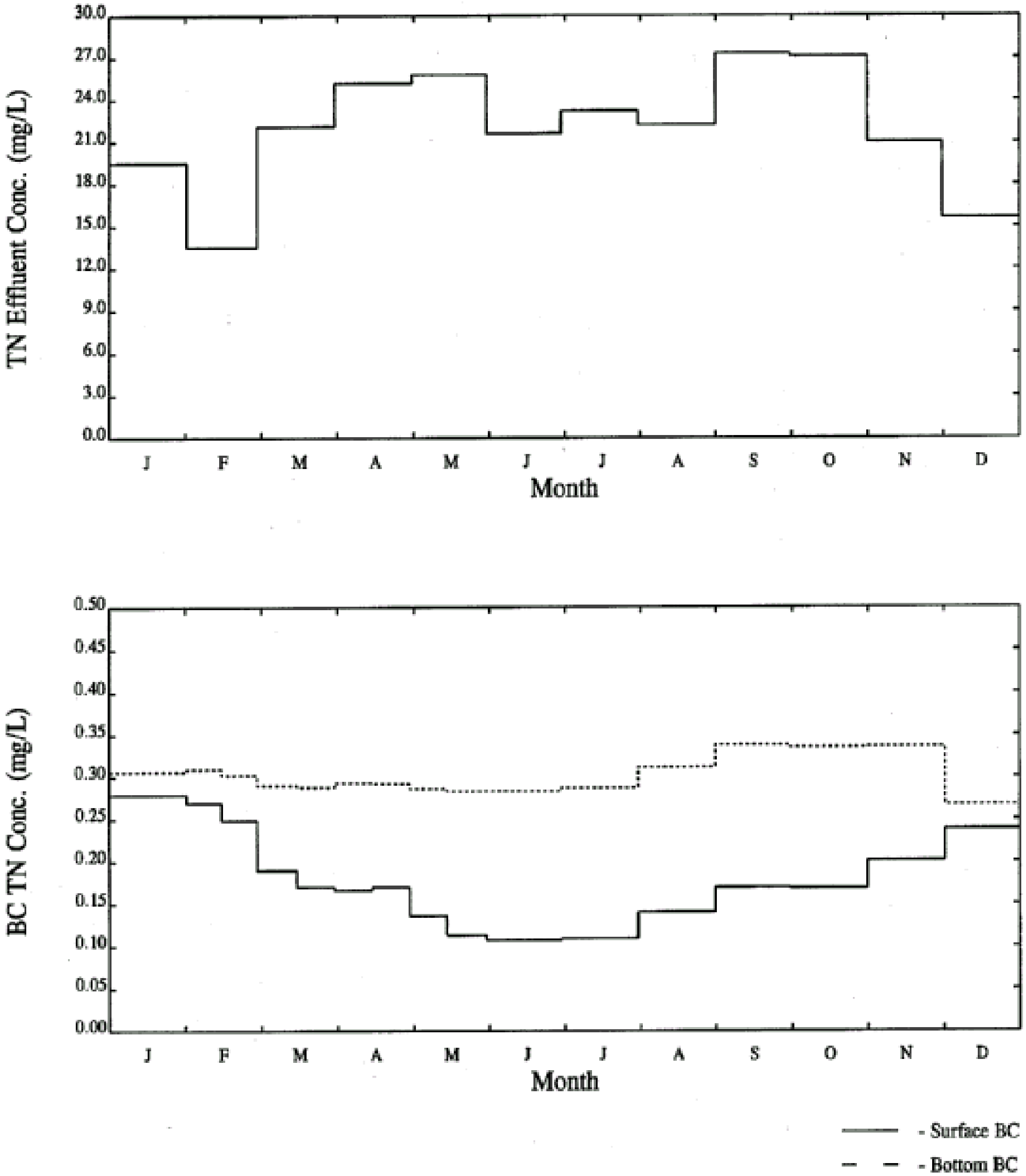

Figure 3-6. Computed temporal total nitrogen concentration in Massachusetts Bay for 1992 based on nitrogen input from only the MWRA effluent (added at the current outfall location) and from the surface and bottom waters at the boundary with the Gulf of Maine.

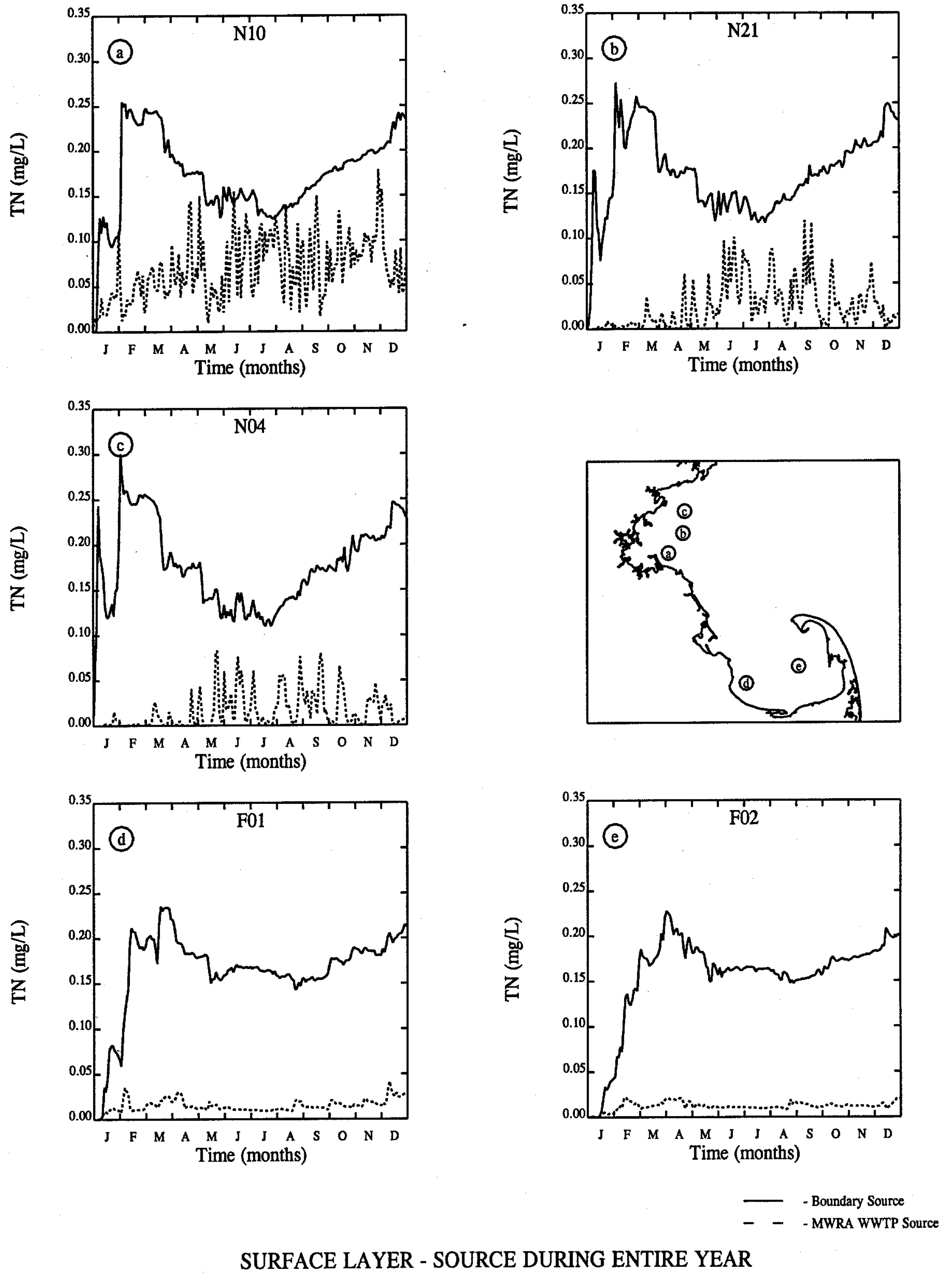

Figure 3-7. Computed temporal concentrations of total nitrogen in the surface layer of Massachusetts Bay for 1992 based on nitrogen inputs from *only* the MWRA effluent (added at the *current* outfall location) and from *only* the Boundary with the Gulf of Maine.

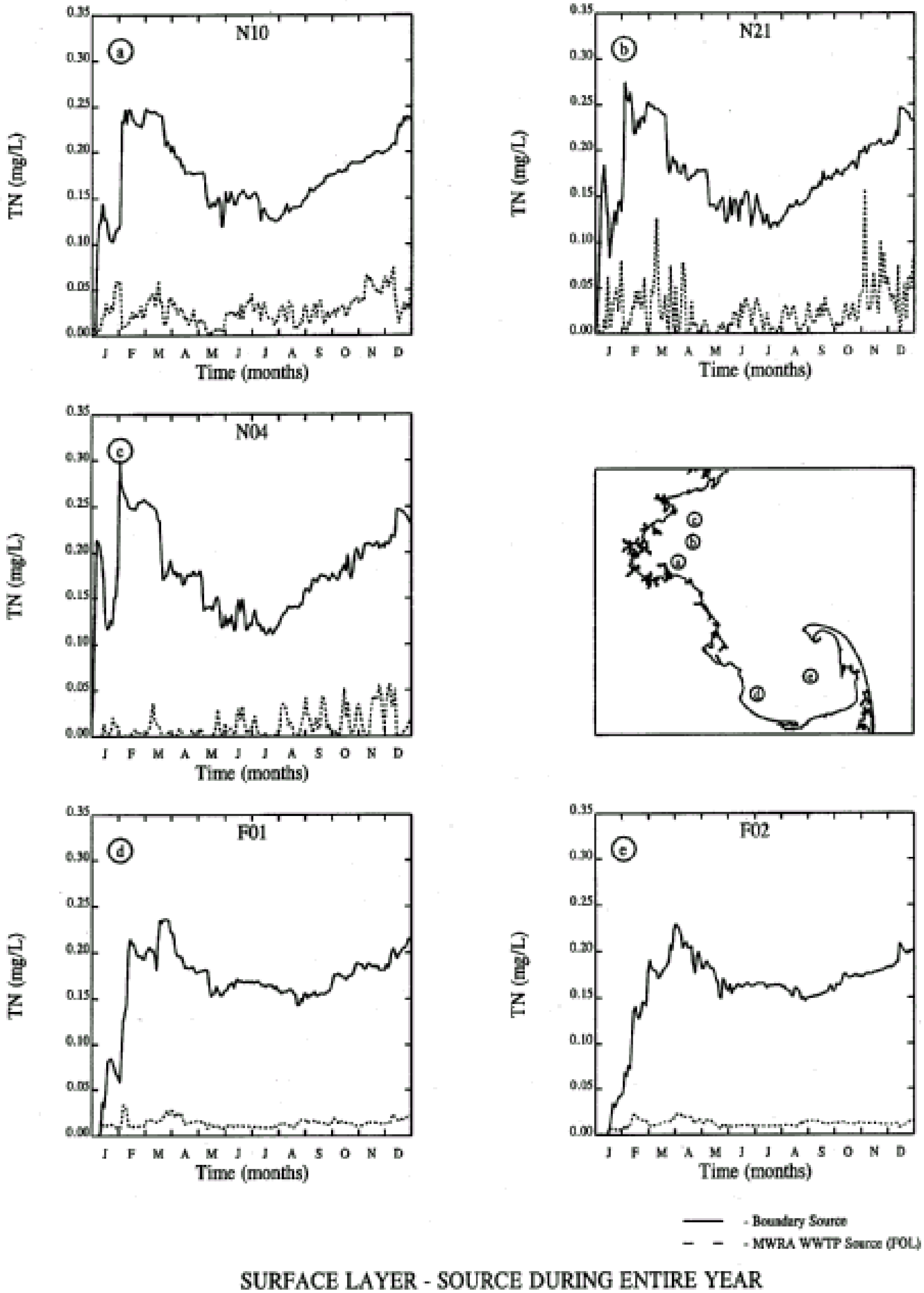

Figure 3-8. Computed temporal concentrations of total nitrogen in the surface layer of Massachusetts Bay for 1992 based on nitrogen inputs from *only* the MWRA effluent (added at the *new* outfall location) and from *only* the Boundary with the Gulf of Maine.

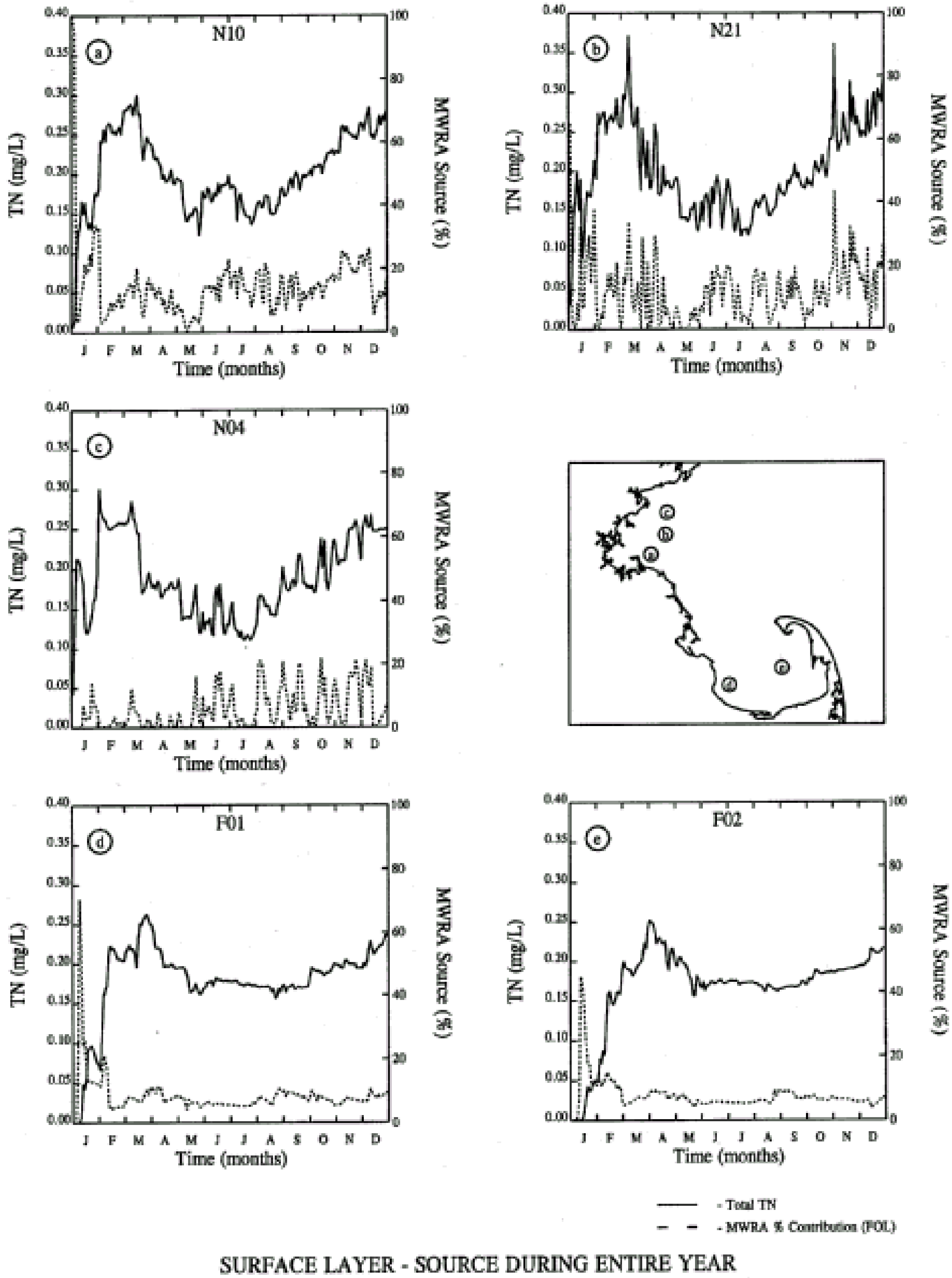

Figure 3-9. Computed temporal concentration of total nitrogen in the surface layer of Massachusetts Bay for 1992 based on *combined* nitrogen inputs from the MWRA effluent at the *new* outfall location and the Boundary with the Gulf of Maine, and the *percent* contribution by the outfall.

3.3 Food web model conceptualization

Kelly *et al.* (1998) (http://www.mwra.state.ma.us/harbor/enquad/pdf/98-04_enquad_report.pdf) developed a conceptual food web model for endangered species in Cape Cod Bay. This concise conceptualization described known information about the food web in Cape Cod Bay relative to right whales and provided suggestions on how a food web model might be structured. The conceptual food web focused on nutrient-related issues and starts with the recognition that two lipid-rich copepod species (*Calanus finmarchicus* and *Pseudocalanus* spp.) are the preferred prey of the right whales in Cape Cod Bay. Also identified were significant trophic linkages in the system, potential food transfer pathways in the Bay, and species that compete for the right whale food. The conceptualization recognized that physical and biogeochemical factors shape food web dynamics and that the interplay among and between the biological and physical components of the system is poorly known and not predictable on an annual basis. Other considerations include the importance of biological species that can disrupt feeding of right whales (e.g., *Phaeocystis*, which can clog the baleen of the whales) or compete with the whales for the food resources (e.g., planktonic invertebrates and planktivorous fish). Scale issues that influence the biological process operating in the system were also identified as important factors to consider in any food web approach. For example, water advection from outside the Bays can carry populations of food resources into the Bays and can bear on the question of timing relative to the development of an adequate resource for the whales in the winter-spring period. The considerations indicated smaller scale factors such as winds, tides, and local weather could bear on whether or not conditions were right for development of zooplankton patches.

The conceptualization concluded in part "fundamental research on the whale food web leading up to the right whale should encompass the following:

> Identify environmental and biological features that create patches of prey acceptable for whale feeding. We believe this topic dictates a fine-scale, high-resolution sampling of patch dynamics and will strongly involve physical factors, many of which are not likely to be affected by moving the outfall. Advanced understanding of molding factors will not guarantee predictability of biological response and food web dynamics to a distant change."

In comments appended to the Kelly *et al.* (1998) report, Dr. Robert Kenney of the University of Rhode Island challenged "the assumption that prey availability for right whales in Cape Cod Bay is exclusively or primarily a function of the abundance and productivity of zooplankton within the Bay." He indicated "If that were the case, then a model of the local food web might reasonably be expected to have some predictive power concerning right whale occurrence. However, it is not that simple. Local zooplankton production is only one of the factors affecting prey availability for right whales, which in turn is only one of several factors controlling right whale occurrence." He further argues that the habitat in Cape Cod Bay must be placed "into larger context with more emphasis on the relative importance of all factors influencing prey availability." and that "Prey availability is then a function of both zooplankton abundance and the entire suite of mechanisms, primarily physical/hydrographic mechanisms, which control how the zooplankton are aggregated into those patches." He concludes that the "local process of nutrient concentration $\rightarrow$ phytoplankton productivity $\rightarrow$ zooplankton productivity is not likely to predict the occurrence of feeding right whales". He further concludes from model predictions of no change in nutrients away from the outfall that

> "one might predict in advance that the results of such a model would be that there would be no change in the occurrence of right whale prey, given the results of previous mixing/dilution models which essentially predict no detectable change in nutrients outside of the mixing zone near the diffusers. The physical factors which are more likely controlling right whale prey availability are much less likely to be impacted by the change in the outfall location …"

These conclusions were drawn in advance of the sensitivity modeling described in Section 3.1. It is clear from the arguments made by Kelly *et al.* (1998) and by Dr. Kenney, the new sensitivity analysis computations, the advanced dilution modeling, and data showing no change in nutrient loading to the system (Sections 2.2.1 and 2.2.2) that nutrient levels in Massachusetts Bay cannot be expected to be different than they are presently when the discharge is transferred. In the absence of such a change, any supposition that the outfall relocation will have detrimental effects on the occurrence of right whales from increased nutrient loads is unfounded.

On the contrary, the data could be used to argue that any reduction in nutrient loading could result in a slightly diminished capacity to produce phytoplankton and zooplankton, thereby leading to inadvertent decreases in food resources for the zooplankton that contribute to the whale food chain. As is will be seen in the subsequent sections, factors other than the zooplankton in the Bays can affect the food chain in the Bays and potentially the occurrence of the right whales.

The considerations of Kelly *et al.* (1998) and Kenney consistently point to factors outside of the nutrient loading from the MWRA outfall as influential in determining the occurrence of the right whales in Massachusetts and Cape Cod Bays. An example of such factors is described in Jahoda and Ryer (1988). That paper discusses the dramatic disappearance of the humpback whales from the Stellwagen Bank area in 1986 and concomitant appearance of right whales in this region. Observationally, the large population of humpbacks that appeared in the region consistently from the early 1970's through the spring 1986 departed the Stellwagen area by the end of May. This population apparently congregated in an area in south channel 60 miles east of Chatham. Concomitant with the disappearance of the humpback whales in the late spring was the appearance in June and July of many planktivorous (plankton feeding) right whales. These animals apparently stayed throughout the summer months in the Stellwagen area. According to Jahoda and Ryer, this event was followed by many observations of planktivorous animals that were observed feeding throughout the summer and early fall. These included basking sharks and sei whales (reported as surface feeding on zooplankton). As crowning culmination to a summer of exciting whale observations, a rare blue whale, the first observed in the area in 50 years, was observed in October and described to have been behaving as if it was feeding on zooplankton. The summer of 1987 was also described as being limited in humpback whale activity following a normal appearance in the spring. Plankton feeders, especially sei and right whales, were numerous in 1986 but not in 1987. Moreover, few whales of any type were observed in the area after mid August of 1987.

The paper also describes potential factors that contributed to this set of events. Essentially the paper argues that the disappearance of the humpbacks and appearance of the right whales in 1986 is related to the disappearance of the sand lance population that occurred over several years leading up to the 1986 event. Observational data suggested the after 1984/1985 the humpback whale activities modified from a more carefree playful behavior to one more focused on feeding. Factors affecting the sand lance population include mackerel, which feed on early stages of sand lance. NMFS data was reported to show a decline in sand lance abundance from 1972 through 1985. The increasing numbers of humpbacks and apparent declining food resource was ascribed to the increased feeding focus by the humpbacks in late summer of 1984 and 1985. Moreover, the sand lance population was reported to have crashed by the time the humpbacks arrived in early 1986. Thus, their primary food source was not available in sufficient numbers to support the population and the humpbacks moved on to better feeding areas.

Within the food web, sand lance are secondary consumers of zooplankton that are primary consumers in the area. Jahoda and Ryer argue that the loss of sand lance (the causes of this are not described) is believed to have reduced grazing pressure on the zooplankton in 1986, thereby allowing the zooplankton to increase to attractive levels for the planktivorous animals. As a result of the sand lance disappearance, the other planktivorous feeders were able to take over the sand lance's niche and flourish at least in 1986. The factors that kept the numbers and length of planktivorous feeders in the Bay for only a short time in 1987 are not discussed. Speculation from the above conceptual model would suggest the zooplankton populations did not grow to levels that could sustain the populations observed in 1986.

How do these observations link to the conceptual model of Kelly *et al.* (1998) and comments from Kenney? Clearly factors outside of the issue of nutrients from a specific discharge were at play, much as indicated by Kenney. One can assume that nutrient loading to the system was consistent and distant from Stellwagen Bank. Nutrient concentrations and input across the Stellwagen Bank area was likely relatively consistent over time; therefore, the nutrient supply in this time period was probably similar among years. Changes in phytoplankton biomass and species composition may have changed, but are not known definitely to have occurred. The long-term zooplankton data set reviewed in Lemieux *et al.* (1998) would suggest that the populations of zooplankton in this area have been generally constant although temporal resolution is limited in the data sets and events could be missed. The overriding sense that this event conveys is that external factors affected not only the occurrence of the whales but the basic processes within the entire ecosystem. This information leads to the question "could the event have been modeled and predicted *a priori* on the basis of fundamental ecological understanding and food web modeling?" The answer in part lies in the deliberations of a workshop convened in November of 1998 to examine issues around predicting right whale distribution (Clapham 1998).

This workshop specifically set out to examine the question of "whether it was possible to predict right whale distribution from environmental data, and to do this with sufficient reliability to be of use in improving research and management on this species". The workshop described the current state of knowledge as having broad understanding of the factors that affect the distribution of this species but a lack of detailed knowledge. The workshop also reiterated that it would be extremely valuable to NMFS to be able to model the distribution of right whales. The review of factors influencing the distributions identifies variability in habitat structure that results in an aggregation (concentrations) of zooplankton prey species as a key factor. These include bathymetry, density structure in the water column, current patterns and the behavior patterns and tendency of the prey species towards aggregation. The workshop further recognized that key information on the energetic nutritional value of the various whale prey species is lacking, that issues of scale are critical to the ability to predict distribution of this species, and knowledge of the underlying ecological processes is required. A first order requirement for predictive capability was identified as the ability to predict the locations of aggregates of the right whales themselves. The workshop also identified a series of retrospective studies that would help to improve our understanding and predictive capability. These included investigations of the overall abundance of *Calanus* relative to right whale distribution, expansion of right whale habitat characterization, environmental and reproductive studies, and planktivore interspecies competition studies. Studies of thermal fronts and tracking of the animals at several levels were also identified as important to improving the knowledge base that could lead to a predictive distribution model. Importantly, two modeling elements were identified. These include a right whale foraging model that incorporated decision making at the individual animal level and a reliable energetics model for the right whales for understanding prey choice, resource thresholds, energy budgets and the decisions making process in relation to the right whale management and recovery plan. Fundamental data collection to support an energetics model development included nutritional value studies of the various prey species and more information on the diet of whales.

It is evident from the workshop's proceedings (Clapham 1998) that much knowledge and information must be developed before the modeling process can be effectively completed. The overall sense of the workshop report is towards understanding the larger scale issues. This affirms the discussions in Kelly *et al.* (1998) and that are evident in the events of 1986/1987 described by Jahoda and Ryer (1988). Moreover, nowhere in the document is a broad based food web model development suggested or is development of models that focus solely on Massachusetts and Cape Cod Bays or any one specific habitat advocated. Thus, this panel of experts focused on the larger scale issues as being critical to our ability to develop models that predict right whale distributions in key habitat areas.

The availability of a conceptual model, identification of the importance of factors outside of Massachusetts Bay system, and questions about food web modeling in general led to further exploration of food web modeling, specifically information on the efficacy of these models for making predictions about the appearance or occurrence of the whales in the Massachusetts Bay. More specifically, the concerns were directed at the connection between nutrients that would come from the MWRA outfall and the whale occurrence.

Accordingly, expert food web modelers' opinions were sought from science and modeling communities outside of the Massachusetts Bay region. A search for scientists conducting food web modeling identified two experts who were willing to answer the challenge. These experts were requested to provide reviews of food web modeling in general and as related to the right whale questions in specific. These reviews are presented the next section.

4. A REVIEW OF FOOD WEB MODELING APPROACHES

The study of nature has always been a balance between the accumulation of direct observations on the ways that real systems behave and the creation of theoretical constructs or models that explain *how* reality is working. The former activity is called empiricism or phenomenology; the latter endeavor is the creation of theory or modeling. If one is charged with the management of a natural system, it stands to reason that the chances of success are increased when both activities are combined. When it comes to ecological communities, it is the interactions among the elements or compartments of the system that become important in the extreme. Thus, the issue relative to interactions of the MWRA outfall with the endangered species is how best to approach webs or networks of interacting species such that some level of correspondence between the action and the species of interest can be made. The systematic analysis of exactly what is observed to transpire in networks of interaction (empiricism) has come to be known as Network Analysis. Attempts to describe mathematically *how* the connections in the foodweb depend upon the constituent populations and their physical environment has classically been considered foodweb modeling.

Increasingly, the object in ecological protection and remediation has become the study of particular impacts in the context of how they affect the entire ecosystem. The tools that managers can employ to conduct whole-systems analysis, however, remain quite limited in number. By far and away the most popular approach has been to been to invoke theoretical constructs and create simulation models of the system in question. Unfortunately, as soon as the ecological community being studied grows beyond a few components or includes any reasonable degree of nonlinearity, such mechanistic simulations often become numerically unstable or produce biologically unfeasible output (e.g., negative or unrealistically high population levels) and thus fail as predictors of future conditions. To overcome these limits, higher dimensional dynamical models are often developed that depend on the state of scientific knowledge to provide reasonable results or ability to develop constructs that represent the processes in the system. The precise representation of these processes is often limited by the state of scientific knowledge. As such, realism may be sacrificed until science proceeds to fill in the information gaps.

Such difficulties are not encountered with dynamic models, however, until midway through the procedure when particular dynamic forms are introduced. The initial steps, the identification of which compartments comprise the system and how these elements are connected with one another, appear sound enough. The possibility thus remains that these underlying structures, or networks, as empirically described, might of themselves contain significant clues to how an ecosystem is functioning. Such is the conjecture of at least two schools of ecological study, which fall under the rubric of *Trophodynamic Modeling* and *Network Analysis*. These approaches comprise rise the major present day thinking in terms of foodweb modeling. These newer approaches are much more advanced than concepts used to model foodwebs a decade ago.

Trophodynamic Modeling and Network Analysis, while endeavoring to describe and explain the many and varied ecological interactions that occur within ecosystems, differ in their approach and implementation. Both attempt ultimately to understand how the trophic levels interplay and transfer energy from the lower to higher trophic levels. Both utilize information on ecological compartments (species information) and the linkages between and among these species. Figure 4-1 provides an example of the major trophic levels that comprise a foodweb, but note that within each compartment there can be many species, thus many linkages that form the foodweb. Classically a foodweb is comprised of a series of food chains that connect large and carnivorous animals to their ultimate plant food and occasionally the nutrients that the plants rely upon for growth. As such the view of a foodweb is of predator prey relationships (e.g., who eats whom) and the tendency of the interactions is along a defined pathway. Such concepts are used to evaluate ecological properties such as bulk biomass abundance, population structure, effects of nutrient inputs, or spatial structure of the foodweb. Each additional level of species inclusion and interaction leads to a more complex set of interactions, and thus require more complex models.

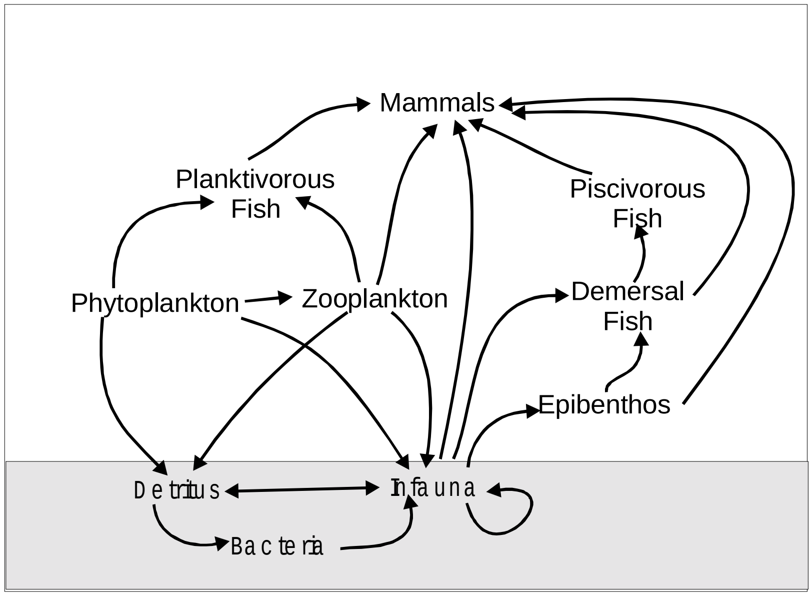

Figure 4-1. Schematic representation of the major trophic levels in a typical foodweb.

Recent approaches to foodweb modeling have incorporated the dynamic aspects of the interactions. As such various levels of theoretical assumption or observational information on the dynamics are utilized. As explained in Section 4.1.1, foodweb analysis discussed in this report represents the dynamic interactions between and among the species that comprise a foodweb. The approach is referred to as a trophodynamical model throughout section 4.1.

Network analysis also uses the connections among ecosystem components. However, network analysis differs from the foodweb approach in that it attempts to examine changes among different steady states and not necessarily with the properties of a given steady state. As such Network Analysis assumes that systems are "perfectly mixed" as part of the input-output functions and is an analysis of empirical data for clues on how the system is currently functioning. Network analysis can evaluate such things as the magnitude of a particular exchange (i.e., between food web compartments), or attribute how much of the energy flow of one trophic level comes from an earlier part of the host population, or examine how the size and trophic organization of the system can be quantified. Thus, Network Analysis, while a static approach (i.e., change from one to a different steady state), enables evaluation of how two compartments affect the other over all possible pathways of interaction. As such the approach requires extensive data and extensive definition of the ecological compartments. That is, network analysis requires cataloging all compartments in the system of interest.

Trophodynamic Modeling and *Network Analysis* are discussed in greater detail in the sections that follow. The sections were developed by expert investigators who were challenged to evaluate the status of current approaches to the analysis of complicated ecosystems in general, and specifically relative to the issue of right whale occurrence in Massachusetts Bay. They present general issues relative to trophodynamical models and network analysis, and consider the relevance of these approaches to the question of whether relocation of the MWRA sewage outfall can affect the endangered species. The premise is that change in the occurrence of northern right whales in Cape Cod Bay and Massachusetts Bay is the effect of concern. In this section "Massachusetts Bay" is used in its wider sense to include Cape Cod Bay.

The sections are organized around five basic areas:

⇒ what the particular approach is (definition),
⇒ the state of the modeling approach,
⇒ what science does and does not understand about the approach in general,
⇒ what we get from the approach, and
⇒ major advantages and shortcomings of the approach.

4.1 Considerations of the status of trophodynamical modeling

This section provides a brief overview and summary of the current status and general issues related to food web modeling from the view of trophodynamical modeling. Note that the question of how sewage outfalls might affect right whales in Massachusetts Bay is a problem in *perturbation theory*. In particular, it can be formulated as a *press perturbation*, which is an experiment of the form:

1. Measure the quantities of interest,
2. Change some parameters in the system, and hold them fixed at the new values,
3. After the system has reached a new steady state (which may include an element of variability), measure the quantities of interest again.

Item 2 is the *perturbation*, the difference between 3 and 1 is the *response*. In the problem at hand, the quantity of interest is right whale abundance, and the perturbation is moving the location of the sewage outfall. The food web modeling considered below focuses to a degree on the approaches used to understand responses to these perturbations.

4.1.1 Definition of food web modeling

To develop this definition, several terms are first offered. An ecological *community* is a set of species that live together within a given area. In the scientific literature, a community is usually taken to be a closed system, but the concept can easily be extended to allow import or export of organisms, and this extension has received considerable attention of late (Polis and Power 1999). A *trophospecies* is a set of biological species that are sufficiently similar in their trophic functioning (how it functions both as predator and as prey) to be aggregated together for the purpose of a particular modeling approach applied to a particular problem. Unless otherwise indicated, the term "species" is to used to mean "trophospecies" throughout the following section. A *food web* is a specification of which species eat which in an ecological community. It is a simple topological specification of the presence or absence of a feeding relationship between each pair of trophospecies. An example can be found at the beginning of Section 3.3

One can conclude practically nothing about a system from a food web in itself. The food web is a skeletal framework to which one can add additional information (such as dietary proportions, process rates, and so on) to produce a model that tells one something. There are two main classes of models that researchers have associated with food webs: *static* models and *dynamic* models. Static models describe mass balance within a food web, at a particular steady state (equilibrium). This steady-state assumption means that neither the biomass of a trophospecies, nor the mass transfer between trophic levels are assumed to vary with time. Typically they generate a large number of descriptive statistics [see for instance, Ecopath (Christensen and Pauly 1992), and network analysis (Ulanowicz 1986a,b)] that characterise flows within the system. A detailed explanation of network analysis, one type of static food web analysis, is contained in Section 4.2.

However, the management utility of static food web modeling approaches are sometimes unclear, for weak linkages can be important for the system, and strong linkages can be unimportant (for instance, Paine 1980). More to the point for the problem at hand, static models tell us nothing about the responses to perturbations, as perturbations are dynamical processes by their very nature. Perturbations are concerned not with properties of a given steady state, but with a change from one steady state to a different steady state. Therefore, only trophodynamical models are considered in the remainder of Section 4.1 "trophodynamical modeling". Note, however, that a mass balance for a food web is often a useful preliminary step toward constructing a dynamical model.

A trophodynamic model is a dynamical system (in the sense of mathematics: a system of coupled differential or difference equations that simulate [or mimic] natural processes) associated with a food web. Food web models tend to be *trophodynamical* in spirit (i.e. these models are trophodynamical in the sense that they assume that trophic interactions among species are the dominant *biological* interactions). That is, they are coupled systems of population dynamical models, for all the species in the system. They can include age structure, size structure, spatial structure, and extrinsic forcing functions, including importation or exportation of organisms. They generally do not include genetic variables or explicit dynamical equations for geophysical processes, although there is no reason in principle why a food web model could not be coupled to a geophysical model of some sort.

4.1.2 The state of trophodynamical modeling

As detailed in Section 4.1.3, trophodynamical modeling is still in its infancy. It is not an established management tool, but rather a vital and lively topic of fundamental scientific research. However, that does not mean that it has nothing to contribute to management problems. Indeed, it is rife with potential contributions because many management problems are "multispecies" by their very nature, and because traditional modeling approaches that use single-species models or predator-prey models have been perceived to be ineffective, especially in dealing with crises such as collapsing fish stocks. Trophodynamical modeling is thus quickly being adopted by a number of agencies charged with management decisions. For example, the Norwegian government has been pursuing MULTSPEC, a food web model for the Northeast Atlantic for about 5 years. Likewise the Canadian Department of Fisheries and Oceans is in the process of a large-scale, systematic data mining process as a basis for trophodynamical modeling of the Northwest Atlantic. A food web model has also been put forward by one group of researchers as a contribution toward understanding the decline of Alaskan marine mammals.

All of this work is being done in the spirit of exploration and elucidation. These efforts yield one form of information that must be considered together with other forms of information in addressing non-routine management problems. The models are not at a stage that can be included as part of an algorithm that produces a management decision as output. The closest instance in which "trophodynamical modeling" was taken as the basis for a management decision involved the process that the South African government used in deciding whether or not to cull Cape fur seals in the Benguela ecosystem as a means of increasing yields in the hake fishery. A workshop of international experts convened in Cape Town in 1991 recognised that traditional 1- or 2-species modeling was not adequate, and recommended instead a five-compartment model for the system --- still pretty modest as a food web model (which would tend to aim for more taxonomic inclusiveness and resolution). Work on this model was commissioned, and the resulting report concluded that any gains from a cull would be minimal, and that, moreover, a cull could actually result in *reduced* yields to the fishery (Punt and Butterworth 1995). After consultation with a panel of international experts convened for the task, the South African government accepted the report's conclusions and opted against a cull.

As well, a protocol for the scientific evaluation of proposals to cull marine mammals, prepared recently for UNEP (United Nations Environmental Programme) by a Scientific Advisory Committee convened for the purpose, contains a significant trophodynamical modeling component (Anonymous 1999).

4.1.3 Present understanding of trophodynamical modeling capabilities

Several modeling types and elements of modeling are considered below. These include bulk abundance models, population structure models, models that incorporate nutrients, and spatial domain issues relative to food web models. The definition and understanding of these types of models are considered below.

4.1.3.1 Bulk abundance models.

A *bulk abundance food web model* is a model in which the state variables are total population number or biomass densities, averaged over the total area occupied by the system. Such a model can have extrinsic forcing functions. These models are well understood. The steps in constructing a bulk abundance model are as follows.

1. *Develop a conceptual formulation of the model.* In this stage, as sketched in Section 4.1.1 species to be included in the model are identified (preferably all known species relevant to the problem statement would be included). Aggregated trophospecies are then chosen. The state variables of the model include the population densities of those trophospecies, averaged over the area occupied by the system.

2. *Decide on the functional form of the model*. Since a food web model is trophodynamical in nature, this means choosing functional forms (structure) for the density-dependencies of the multispecies numerical responses and functional responses (May 1981; Yodzis 1989, 1994a). For instance, consider a predator's "functional response", which is its consumption rate of prey, as a function of prey abundance. The consumption rate might increase linearly as prey abundance increases, or it might saturate as prey abundance increases. Other complications may, or may not (this is a decision that has to be made by the modeller), be present. For instance, if a predator has more than one prey species, which is typically the case, then its consumption rate of prey species A may depend not only on the abundance of prey species A, but also on the abundance of some other prey species, say B (because, for instance, if the predator spends more time feeding on B, it may have less time available to feed on A). This kind of effect, which is terrifically difficult to measure in the field, can influence quite significantly the conclusions drawn from the model. Many modeling exercises simply assume a functional form (usually as linear as possible) with no justification or discussion whatever. This is not a sound procedure. Ideally, the functional forms from data are desired. In practice this is extremely difficult. Therefore, modellers often follow a procedure that constructs several different models, with a range of functional forms. The results are then analysed to test the influence of functional form on model output.

   Another approach to the issue of functional form is to consider *small* perturbations, which can be treated by linearizing the model (Yodzis 1994b, 1998a). This reduces the problem of specifying functional forms to the problem of specifying partial derivatives of functions at a point. That is instead of having to specify *infinitely many numbers* in the form of a function, specification of only *one* number in the form of a derivative (for each function and each state variable in the model) is required. This local type of model is, in principle, more reliable than a global one, because it requires less information as input to produce an output. The disadvantage that has to be traded off is an inability to handle large perturbations. For example, this approach would not be particularly useful for investigating the recovery of highly depleted cod stocks in the northwest Atlantic. However, it might be appropriate for looking at the influence of effluent outflows on populations in Massachusetts Bay, since the quantities reaching such a location are so far from the source that the perturbation *is* a small one.

3. *Calibrate the model*. Parameter estimation can come from empirical observation, theoretical argumentation (such as the use of allometric scaling; Yodzis and Innes 1992), or least-squares fits

of solutions to time series data. All of these methodologies are well understood. Nevertheless, parameter estimation is a substantial problem in models that contain many parameters, as food web models do. Inevitably, many parameter estimates will contain a substantial amount of uncertainty. Indeed, many of these parameters do not *possess* definite values at all, but rather they fluctuate over time. The variability can either be an expression of our own ignorance of parameter values or an expression of natural variability in those values. Regardless, one can assign probability distributions rather than fixed numerical values to some parameters. In this case the model, rather than predicting a definite answer, will provide a probability distribution for the answer. Typically one is interested in one "tail" of the distribution, for instance, the probability that a relocation of the outfall in Massachusetts Bay will result in, say, a 50% reduction in right whale abundance in the Bay. However, looking at the whole probability distribution frequently holds surprises that need also to be weighed in the decision-making process (Yodzis 1998a). For instance, one might very well find a significant probability that relocation of the outfall will result in an increase in right whale abundance rather than a decrease.

In an uncertain science such as ecology, exact predictions can seldom be made and investigators should not acknowledge this. In reality, modeling can realistically only provide probabilistic information. Management agencies, and the general public, in their turn, will have to accept that this is the best scientific information modellers can provide, and be prepared to make decisions on that basis.

The minimum data requirements for a bulk biomass model are:

A. Dietary proportions for all species,

B. Population biomass for all species (either time series, or putative equilibrium values for a period of time during which biomass are considered to have been relatively constant),

C. Typical adult body masses for all species,

D. Extrinsic influences such as harvest rates or nutrient inflow rates.

These data are often difficult to develop and require active research to be fully understood.

4.1.3.2 Models with population structure

Population structure such as age- or size-structure can easily be accommodated in a model, and might be required for some of the larger species, which may undergo considerable ontogenetic shifts in diet. However, any inclusion of population structure adds further data requirements: in the list of minimal data required for a bulk biomass model, we would then have to replace everywhere “for all species” with “for all subclasses (such as age- or size-classes) of all species”. Population structure should probably be included for any target species (those of particular interest) that undergo significant ontogenetic shifts in diet, but there is some evidence that including population structure for ancillary species has little effect on predictions of press perturbations for target species (Yodzis 1998b).

4.1.3.3 Models with nutrient recycling

Nutrient recycling can also be incorporated in a straightforward manner. For instance, Moloney and Field (1991a) have developed size-based plankton models that incorporate both carbon and nitrogen flows, and applied them successfully to several systems in the southern oceans (Moloney and Field 1991b).

4.1.3.4 Models with spatial structure

A major shortcoming of current trophodynamical modeling is the inability to handle spatial structure. Very little work has been done to incorporate spatial structure into food web models. There are both practical and scientific difficulties here. A food web model with spatial structure would be tremendously complicated, and require a huge amount of data. There are also significant unresolved issues in the underlying science. For example, animals have a variety of strategies of movement, of which we know little; and the role of physical processes such as turbulence is only partly understood. Even at the level of single populations, there are major gaps in our understanding. For instance, plankton patchiness, particularly blooms of toxic species, and the factors enabling patch development are a tremendously important aspect of food webs, yet are an inadequately understood aspect of nature.

4.1.4 Utility trophodynamical modeling

In a complex ecosystem (such as Massachusetts Bay), many processes and interactions are simultaneously occurring, and each is being influenced by some of the others. For example, if substantial amounts of additional nutrients are injected (which is not projected from the MWRA outfall relocation), changes in phytoplankton may be triggered, which can trigger changes in zooplankton. This in turn can trigger other changes in phytoplankton, which can trigger changes in planktivorous fishes (triggering in turn changes in zooplankton, hence also in phytoplankton), and so on. In trophodynamical modeling, all of these many processes, all going on at the same time, are combined to produce a result for, say, zooplankton abundance. Thus, the main utility of trophodynamical modeling is to address perturbation experiments in multispecies settings. For all its inadequacies (Sections 4.1.3 and 4.1.5), trophodynamical modeling is the only way scientists have of doing this.

4.1.5 The major advantages and shortcomings of trophodynamical modeling

Trophodynamical modeling has three major advantages but four major shortcomings. The advantages include: (1) it encourages development of a very good conceptual understanding of the system, (2) it can be used to address many questions that may arise because a whole system is encompassed, not just a few species of interest, and (3) it is the only existing technique to try to answer certain questions. The shortcomings include: (1) it requires a very good conceptual understanding of the system, (2) it requires substantial amounts of data, (3), it currently does not address spatial issues adequately, (4) it is, for the most part, untested; and there are inherent difficulties in verifying a food web model. These attributes are described in more detail below.

4.1.5.1 Advantages

Conceptual understanding: Even at the relatively crude level of the topological food web that is to underlie a model, decisions must be made that may affect the model’s outcome significantly, and yet which require much information and understanding. These decisions are of three kinds: spatial extent, taxonomic inclusiveness, and degree of population structure.

> <u>Spatial extent</u>: The spatial extent to be included in the model is the first conceptual decision that must be made. This may include simple spatial structure and some import/export. Critical to this is the ability to set scales that come as close as possible to having a closed system.
>
> <u>Taxonomic inclusiveness and resolution:</u> The second conceptual decision is which biological species to include in the model and which of those species to aggregate together into trophospecies (taxonomic resolution). For instance, if one were going to take the Cape Cod Bay model of Kelly *et al.* (1998) further, one would have to decide which copepod species to lump together. As far as right whales are concerned, this might come down to a high lipid content category, and a low lipid

content category. Decisions on how much detail to include in the "Other Copepod Predators" category must be made as well. A number of these taxa might need to be treated as separate trophospecies. Furthermore, still more species might need to be added to the food web, for example, predators of the "Other Copepod Predators". Some species of interest might not sensibly be viewed as part of the food web at all. For instance, since they range very widely and spend only a small part of the year in Massachusetts Bay, perhaps right whales should be viewed as a "tourist species" from the standpoint of the Massachusetts Bay food web. That is, they may not affect the food web significantly, or they may not be significantly affected by it.

Population structure: The third conceptual decision that must be made is the amount of population structure to include for each trophospecies in the system. Age classes might be required for some of the larger animals, whose diet may change significantly as they grow. For some taxa, size classes might be a more sensible approach than classes based on species membership (see for instance, the plankton work of Moloney and Field 1991a, 1991b).

Ideally, these decisions should be made on the basis purely of the science of the system and of the individual species in it. Of course, one wants to make these decisions in the context of whether each added complication will actually affect the predictions of the model, and this may be difficult to know *a priori*. One could argue that the best starting point is to build a model with the highest possible degree of taxonomic inclusiveness and resolution --- the model can always be simplified later and one can at that point see how much difference it makes. In practice, many of these decisions will be constrained by the available data and by the feasibility of obtaining appropriate additional data. Most food web projects start by gathering together all available data, and determining what one can conclude on that basis. However, this approach can be dangerous: if we force implementation of a food web model before building up an adequate data base, we run a higher risk of getting unreliable answers.

The second advantage is that trophodynamical modeling encompasses a whole system, not just a few species of interest. Thus it can be used to address many questions that may arise. Once a food web model is developed and in hand, many questions about the system can be posed, including ones that may not have been conceived of when the model was constructed. For instance, should toxins become an issue, they could be added to a food web model and thus deal with biomagnification.

The third advantage is that this approach may be the only way to address certain issues. Simply put, there are problems that cannot be addressed scientifically in any other way.

4.1.5.2 Shortcomings

Conceptual understanding. The first shortcoming, the need for a high degree of conceptual understanding in order to produce a credible food web model, has already been discussed in detail in Section 4.1.5.1 as an advantage. This requirement can be viewed as advantageous in that putting together a food web model forces us to substantially increase our understanding of the system and provides a framework for doing so. However, it can be viewed as disadvantageous in that the required effort is enormous. Producing a credible food web model for a complex system such as Massachusetts Bay is a tremendous undertaking, which would require years of effort.

Data requirements: Food web models require a tremendous amount of data. Frequently, this circumstance will render trophodynamical modeling impracticable. If good time series data are available for all population abundances, these can be used to help in parameter estimation. Allometric relationships can be used to estimate some physiological rates (Yodzis and Innes 1992, Yodzis 1998a). As well, some of our ignorance can be addressed through Monte Carlo analysis (Yodzis 1998a). In this approach, poorly known parameters are assigned probability distributions, using biological reasoning to constrain those distributions.

However, parameters treated in this way add greatly to the uncertainty of the final result, so this should be done only as a last resort and it cannot be done with too many parameters.

Spatial structure: Very little research has been done with respect to spatial issues in a food web context. This need not necessarily involve a spatially explicit model. For instance, if a set of conditions (which would presumably include mean density) that favoured the formation of plankton patches were known, this could be input to a bulk abundance model as an extrinsic factor. However, such knowledge is extremely limited.

Verification: Verification of modeling approaches requires a long-term observational monitoring program, which can be quite expensive to implement. For this reason, trophodynamical modeling is at present largely untested. Currently, the Norwegians feel that MULTSPEC (Borgstad et. al. 1997) has been performing well, but far more experience with the approach is required before its predictions can be used with confidence (Tore Haug, personal communication to P. Yodzis, 1999). Formulation and testing of a food web model is a desirable long-term goal for any complex managed system, but one needs to be realistic about the difficulties confronting the approach in the short term.

4.2 Network Analysis

This section provides a brief overview of general issues related to network analysis as related to food webs. Network Analysis contrasts with trophodynamic modeling in that it deals with changes between different steady states whereas the trophodynamic models represent the dynamic interactions between and among the species that comprise a foodweb.

4.2.1 What is network analysis?

Foodweb analysis in the more classic sense primarily deals with the qualitative question, "Who eats whom?" The answers to this question can be depicted pictorially as a set of boxes connected together by directed arrows (see for example Figure 4-1). Quite often, the diagrams are highly connected into a complicated web -- whence the moniker, Food*web* Analysis or when dynamics are involved, foodweb modeling. The foodwebs of various ecosystems have been assembled and their topological attributes, such as connectivity, path length and hierarchical structure have been assessed to see if any patterns emerge (Briand 1985, Cohen *et al.* 1986, Paine 1988). This exercise, which does not involve any dynamical assumptions (nor any flow magnitudes) has been referred to as Foodweb *Analysis*.

Network Analysis, by comparison, is a quantitative expansion of the underlying question to encompass, "Who eats whom, *and at what rate*?" That is, attached to each arrow connecting ecosystem components is a number that quantifies the magnitude of that particular exchange. Network Analysis requires far more information than does Foodweb Analysis, but its quantitative dimension allows it to address far more sophisticated issues. Moreover, if empirically snapshots of a system are compared at two different times, the assumption is that some forms of physical dynamics have transpired to yield differences. Hence, comparison of the states of a system at two different times can be made without making any assumptions about *mathematical* dynamics. Network Analysis allows such comparisons, and if a time- series of ecological “snapshots” is available, a "motion picture" of the *physical* dynamics is represented. Whence, such time series analysis can be used in Network Analysis to describe the physical dynamics, but in post facto manner.

4.2.2 Present understanding of ecological Network Analysis

As currently comprised, Network Analysis addresses four major questions about ecosystem functioning:

(1) "With what magnitude do any two ecosystem compartments affect each other over all possible pathways?" (Input- Output Analysis.)

(2) "At which trophic levels does each compartment feed?" (Trophic Analysis.)

(3) "How much of each flow is comprised of material that was earlier part of the same host population?" (Recycle Analysis.)

(4) "How can the size and trophic organization of the system be quantified?" and/or "What is the contribution of each component to the magnitude of the overall structure?" (Whole System Status.)

The response to each question issue is considered next.

Input- Output Analysis: In creating quantified networks, one is cataloguing all of the direct trophic interactions in the system. Can one use this structure to estimate what the indirect consequences might be? For example, a Killer Whale entering a coastal embayment feeds only on the larger fishes it finds there. It ingests no appreciable amount of microscopic plant life, or phytoplankton. The prey fish, however, feed in turn on smaller fish, some of which feed on still smaller animals (mostly invertebrates), many of which ingest phytoplankton. In assessing the importance of phytoplankton to the whale, the ecosystem manager might well ask, "What fraction of the material comprising the whale's intake once was embodied as phytoplankton (as opposed to, say macrophytes [large plants] or detritus [nonliving organic material])?" Alternatively, one could pose the obverse question, "Of the carbon fixed by phytoplankton, how much eventually is ingested by the whale?" Still further, one might want to know whether some particular small fish is a contributor or a competitor to the whale.

Questions similar to these have been posed decades ago by economists, who analyzed networks of material and cash flows to assess how the economy was functioning. Wassily Leontief (1951), in particular, was awarded the Nobel Prize in Economics for developing what has come to be known as "Input- Output Analysis", a set of matrix algebraic operations designed to quantify how much of each economic process is necessary to meet a given final demand. Bruce Hannon (1973) suggested the employment of Leontief's methods in ecosystems, and Szyrmer and Ulanowicz (1987) modified those matrix calculations so that they could address ecologically more meaningful questions, such as the first two in the preceding paragraph. Later, Ulanowicz and Puccia (1990) expanded the analysis to include either positive or negative interactions, and thereby distinguish between indirect competitions and enhancements.

As an example of how the results of I-O analysis might be employed, Baird and Ulanowicz (1989) studied the "indirect diets" of two carnivorous fishes in Chesapeake Bay, the striped bass (*Morone saxatillis*) and the bluefish (*Pomotatus saltatrix*). Superficially, the two predators seem to be heavy competitors within the same niche. A perusal of their indirect diets revealed, however, that the striped bass acquired most of its resources ultimately from the microscopic plants (phytoplankton), whereas the bluefish depended via indirect routes mostly on dead organic matter on the Bay bottom (much of which has its origins outside the system.)

Trophic Analysis: One major outgrowth of Input- Output Analysis was the observation by Levine (1980) that the matrix methods of Leontief could be altered slightly to compute the effective trophic level at which each predator feeds. That is, many predators feed at more that a single trophic level. When striped bass, for example, ingests Bay anchovy, it is feeding mostly at the fourth trophic level. (Most material flows to the Bay anchovy from the phytoplankton via very small hard- bodied crustaceans.) When it eats a small blue crab (*Callenectes sapidus*), however, it is feeding at the end of a complicated subweb that includes pathways of length four or five. Input- Output Analysis permits one to weight and average all pathways of all lengths in the pyramid of life that supports each predator to ascertain the effective level at which the animal is feeding. This effective trophic level often is an indicator of how well that species is faring in the context of the given ecosystem. For example, when a tidal marsh creek off Crystal River in Florida was disturbed, the average trophic rank of the predator stingray fell from 3.83 to 3.69 as a result.

Another advantage of the matrix algebra used in Input- Output Analysis is that it can be applied in a stepwise fashion to identify exactly how much each predator feeds at each integral trophic level. Doing so, for example, for the Bluefish in Chesapeake Bay reveals that this species feeds 10% at the third trophic level, 22% at the fourth, 67% at the fifth and 1% at higher levels (Baird and Ulanowicz 1989). It becomes possible, therefore, to apportion the activity of the bluefish to abstract levels 3, 4 and 5 according to these ratios (Ulanowicz and Kemp 1979). One may do so in a way that accounts for all material in circulation (Ulanowicz 1995), so that one in effect transforms a complicated foodweb (Figure 4-2) into an equivalent foodchain, as in Figure 4-3.

The simplified food chain often can be used to diagnose the results of system perturbation. Usually, whenever the entire system is impacted, this equivalent chain is shortened and/or the efficiencies with which material is transferred along the chain drop in response (Ulanowicz 1996).

***Recycling Analysis*:** It has been hypothesized that the presence of feedback loops or circuits of material recycle are indicative of the active controls in ecosystems (Forrester 1987). The problem with very complicated foodwebs, however, is that cyclic pathways often abound in them. In fact, there is always the possibility that the number of simple cycles (those without any repetition of elements) in an ecosystem network could grow enormously high, due to the combinatorics involved. Fortunately, the relatively sparse connectivities of most trophic networks (ca. 20%) and the relative absence of any cycles that do not include dead material keep the number of simple cycles within readily countable proportions. Baird and Ulanowicz's network of Chesapeake Bay, for example, contains only 67 simple cycles. Ulanowicz (1983) has developed an efficient form of "backtracking" algorithm to identify and remove all simple cycles from an ecosystem network.

Knowing exactly where and to what magnitude recycling is taking place (or, alternatively, not occurring) in a network can yield important clues as to how the ecosystem is functioning. (Some ecologists have considered it nonsensical to speak about the concept of function within an ecosystem that possesses no obvious external purpose.) Figure 4-4, for example, shows the aggregate of all recycling flows that occur in the Chesapeake network of Figure 4-2. The reader will notice immediately that recycle is split into two domains, one comprising most of the members of planktonic (floating in the water column) foodweb, and the other comprised of deposit- feeding, bottom- dwelling organisms and carnivorous fishes. Conspicuously *not* engaging in any recycle are the bottom- dwelling filter feeders, such as oysters and other bivalves, and the filter- feeding fish, such as Bay Anchovy and Menhaden. These components are seen rather to *function* as bridges that extract material and energy from one domain of control (the planktonic) and inject it into another (the deposit feeders and carnivorous fishes.)

Similarly, two members of the microbial community, the free- floating bacteria and the microscopic flagellates (very small bacteria- eating organisms with tiny tails for propulsion), which are thought in the open ocean to help recycle nutrients, engage in *no* recycle in Chesapeake Bay. Rather they are serving to shunt carbon out of the ecosystem.

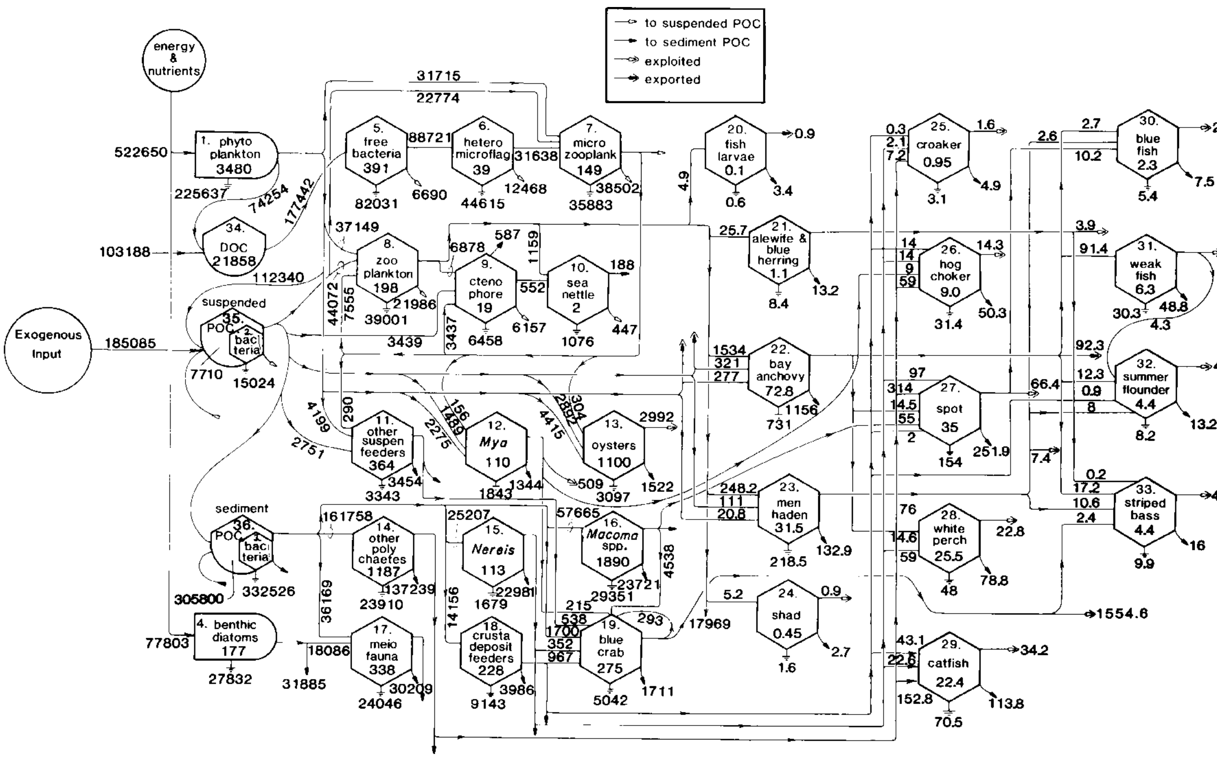

Figure 4-2. Schematic representation of the annual carbon flows among the 34 principal components of the Chesapeake mesohaline ecosystem. Carbon standing stocks are indicated within the compartments in mg/m^2, and the indicated carbon flows are in mg/m^2/yr.

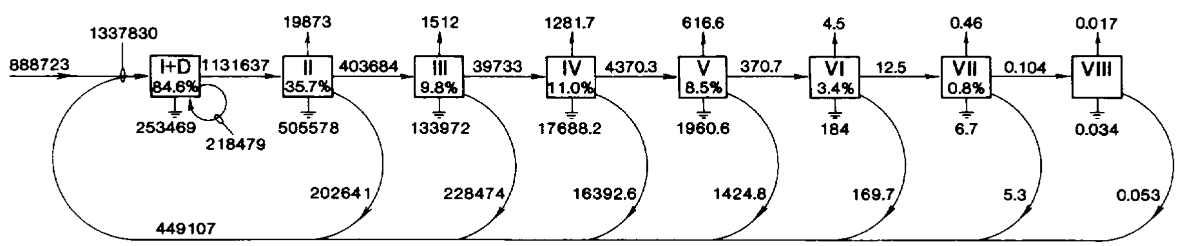

Figure 4-3. The trophic chain corresponding to the network in Figure 4-2 with primary producers and detritus merged. The percentages in the boxes represent annual trophic efficiencies.

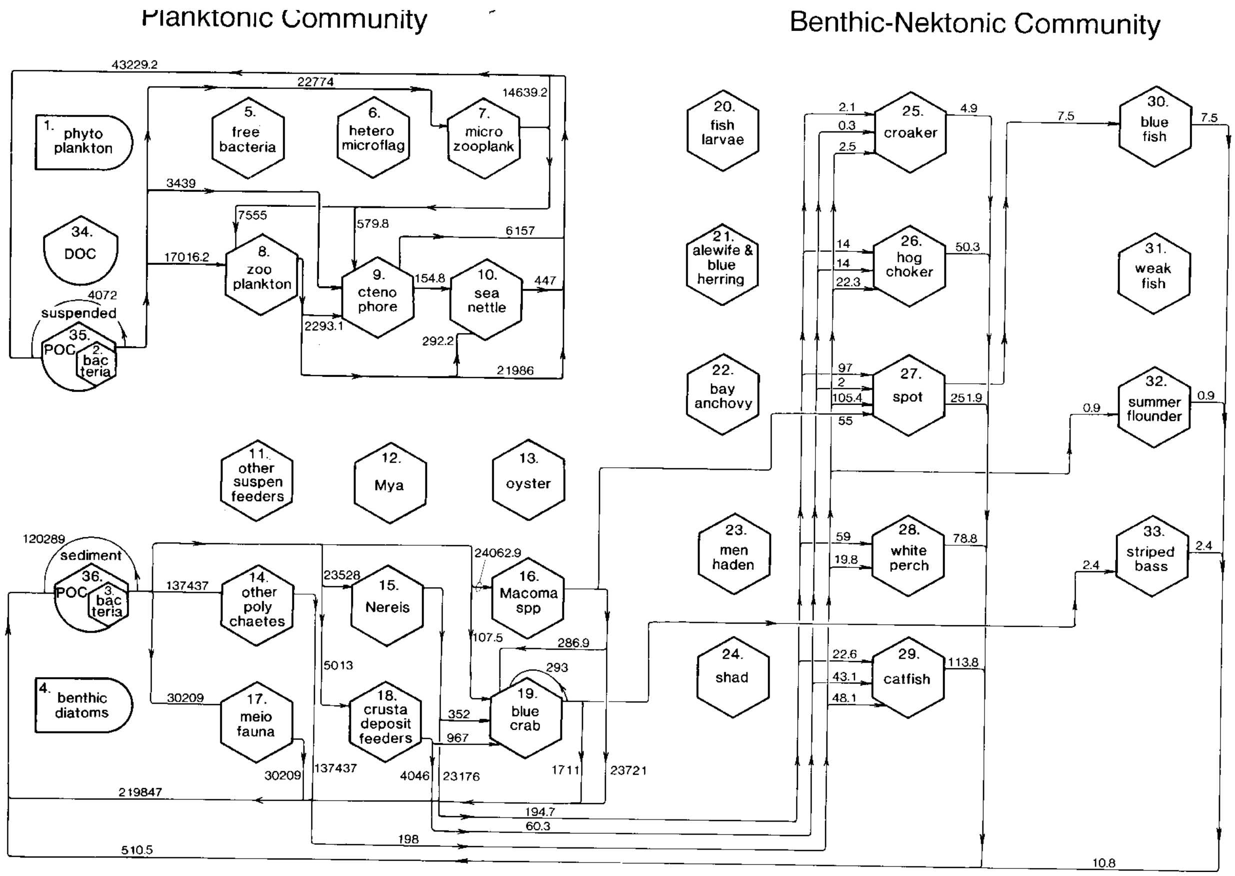

Figure 4-4. The composite cycling of carbon that occurs in the flow diagram in Figure 4-2. Units of flow and numbering of the compartments are the same as in Figure 4-2.

Whole System Status: If ecosystems are to be treated as wholes, it is imperative that measures be developed that can indicate their current status. In the network depiction of ecosystems, the major attributes are that of system size (or activity level) and its degree of organization. The first property can be gauged quite succinctly simply by aggregating the magnitudes of all the processes occurring in the system. The system characteristic "organization" is a bit more difficult to quantify, but the quantity "average mutual information" from Information Theory appears to capture all the pertinent aspects of how the system is put together (Rutledge *et al.* 1976, Ulanowicz 1986a). The product of these two has been termed the system's "ascendancy" and has been hypothesized to be the leading indicator of an ecosystem's growth and development (Ulanowicz 1986a).

If estimates of an ecosystem trophic flow network are available under two different conditions, the relative measures of the system ascendancy can be invoked to quantify any change. For example, whenever an ecosystem is significantly perturbed, one expects the opposite of growth and development to be evident, and the ascendancy can be expected to decrease (Ulanowicz 1996). Thus, changes in ascendancy can be used to quantify the degree of impact upon an ecosystem in response to disturbance. Elsewhere, systems that undergo eutrophication (excessive enrichment) display a characteristic combination of changes in the factors of the ascendancy that distinguishes that negative impact from the more benign process of simple enrichment (Ulanowicz 1986b).

Ecosystems often are characterized metaphorically as being "healthy" or possessing "integrity." With the advent of system ascendancy and related measures, such as its complement, system "overhead," it now becomes possible to address the issue of system health in quantitative fashion (Mageau *et al.* 1995). Thus, it may soon be possible to say whether the response of an ecosystem to a disturbance has put it outside the domain of healthy functioning.

Another useful feature of the ascendancy is that with it one may readily quantify the part- whole relationship. That is, the individual contributions of a particular ecosystem component to overall system performance are easy to isolate. Suppose, for example, that a manager has available two networks, one representing the ecosystem of a coastal embayment, and the other that of the coastal shelf. Suppose further that a species of whale is common to both ecosystems. One may then compare the relative contributions of the whale to each ecosystem to ascertain how it is faring in each context (Monaco 1995).

How sensitive the system ascendancy is to changes in component stocks or individual transfers is a direct measure of the importance of that biomass or flow to the functioning of the entire system. Ulanowicz and Baird (1999), for example, estimated separate networks for the carbon, nitrogen and phosphorus flows occurring in the Chesapeake ecosystem. They then used the sensitivities of the ascendancy to the stocks of each chemical element in any particular taxon to determine whether C, N or P was limiting to each stock. That is, they used the ascendancy in place of Liebig's Law of the Minimum. Liebig’s Law cannot be used, however, to determine which *source* of the limiting element is most important to each taxon. They were able to use the sensitivities of the ascendancy to each individual flow into a taxon to determine which one is limiting.

Finally, if one has estimates of an ecosystem network as it changes over space or time, one may use higher dimensional versions of the ascendancy to quantify the status of the ecosystem over that particular spatial or temporal domain. In particular, one may use the measure to identify the location or time where systems "bottlenecks" may have arisen (Ulanowicz, 1999). In summary, given enough data, one could use the system ascendancy to single out those taxa and those regions of time, space that are of greatest strategic importance to the performance of the entire ecosystem.

4.2.3 Data requirements for Network Analysis

At the outset of any network analysis, it is necessary to choose a "currency" or medium that is common to all network components with which to quantify the flows and stocks. Usually, this medium is a chemical element common to all life forms, such as carbon (C), nitrogen (N) or phosphorus (P), but it could as well be some form of energy. Within any one network, all magnitudes should be expressed in terms of the same medium. It may be, however, that one wishes to compare the kinetics of different chemical species, e.g., C, N, and P. In such case, one estimates separate, parallel networks for each currency and expresses all concentrations in the units of common mass (e.g., mg/l.) (See Ulanowicz and Baird 1999.)

Once a currency has been selected, it is necessary to measure or estimate ALL the flows of this medium among all the components of the system. This set of flows can be distinguished as four separate types: (1) Inputs to the system from the outside, (2) Exchanges within the system, (3) Exports of useable medium to other systems, and (4) Dissipation of medium to its lowest energy form (e.g., N_2 in the case of nitrogen). In addition, for some analyses it is necessary to know the densities or stocks of each ecosystem component expressed in the terms of the selected currency.

It quickly becomes obvious that Network Analysis requires copious data. Often, the number of flows to be estimated runs into the hundreds or even thousands. It almost never happens that all these magnitudes have been measured. The situation is far from hopeless, however. Primary production figures (e.g., how much carbon each unit of plant material fixes per unit time) are usually available in the literature for most ecosystems. Furthermore, most systems have been studied enough that, with relatively little effort, one can assemble estimates of the densities of the animal compartments (one of the necessary items). Once these biomass densities are known, one may usually consult published tables of physiological constants that pertain to most organisms in the system. For example, suppose a density of a certain forage fish has been estimated by fisheries biologists to be, on average, 140 mg ash-free dry weight per square meter. Usually, one-half of the ash-free dry weight consists of carbon, so the density of the fish is set at 70 mg carbon per square meter. Furthermore, one discovers that an individual fish consumes 0.1 of its weight per day. Whence, the demand of the population will be for 7 mg-carbon per sq. meter per day. Looking at the physiological tables (e.g., Joergensen *et al.* 1991), one finds that the same species of fish respires about 2.8% of its weight per day, or 2 mg-carbon per sq. meter per day. The residual 5-mg- carbon per sq. meter per day is available for consumption by the predators of that fish population.

The 7-mg of demand must be apportioned among the various food sources of the fish. Thus it becomes necessary to know the dietary proportions of the species (within the current ecosystem). It may be known, for example, that 70% of the intake consists of zooplankton, 10% of particulate detritus (dead particles of organic remains floating in the water), and 20% of living phytoplankton. (When dietary proportions remain unknown, one sometimes resorts to the assumption that intake is proportional to the stocks of prey available.) Whence, one estimates exchanges of 4.9, 0.7, and 1.4 mg-carbon per sq. meter per day flowing to the fish from the compartments for zooplankton, particulate detritus and phytoplankton, respectively. Proceeding in this manner, one matches the demands of the predators with the available productions of their respective prey, usually with the aid of a spreadsheet algorithm, such as LOTUS or EXCEL. Excess production is usually considered to flow to some pool of detritus, whereas any surfeit of demand is usually an indication that the estimates of stock levels need to be reexamined. Some investigators prefer to conduct this tedious balancing act by hand so as to be able to control all the assumptions made. Others choose to employ one of several algorithms (e.g., see www.ecopath.org) that will automatically balance the network, given the biomass estimates, physiological constants and dietary fractions. The appearance of such balancing software has radically increased the ease with which networks can be assembled, and the number of networks appearing in the literature has increased apace (Christensen and Pauly 1993) including NETWRK at www.cbl.umces.edu/~ulan/ntwk/network.html, which executes all four of the analyses described above.

4.3 Summary of food web modeling approaches

The food web assessment approaches described above each have limits and advantages. It is clear from the discussions that these approaches require much planning and forethought to effectively execute. They are clearly not at the stage of producing information that can be translated directly into management decisions; rather the outputs must be used with other information. Importantly, both approaches appear to be most effective when the responses being examined are substantive and in response to a clear perturbation that shifts the condition from one state to another. That is, subtle changes in a system will not be effectively modeled under these approaches under current ecological understanding and data limitations.

These reviews also identify several factors in common relative to the evaluation or prediction of the connection between the occurrence of right whales in the Massachusetts Bay system and the new MWRA outfall. The first common element is the development of a clear and complete conceptualization of the issue and the ecological components that interact to affect the issue of concern. The second is an expectation of a measurable perturbation of the system that can be captured by the model, i.e., the system needs to move to a new steady state in the case of the foodweb analysis approach. The third common factor is the need to include all species relevant to the problem in the model (i.e., not only the prey of a specific species but other predators of primary prey of the species of interest). An understanding of the functional role of each species is also important and requires a through justification and definition for inclusion in the modeling approach. The fourth common factor is a requirement for accurate estimates of the biomass (or other measure of flows and stocks) on all of the species interacting in the food web and in many cases the population structure of the important species. These are difficult to find and to develop. Poor data on these factors can cause substantial errors in the computations if not estimated well. The fifth area is identification of appropriate scales to include in the model (e.g., the ability to close the system to the maximum extent possible). This can substantially affect the ability of any model computations if not properly defined. In both approaches, the availability of the appropriate types of data at the correct scales of interest is preferred over limited data. The availability of data is often limited and difficult to develop with any degree of accuracy. Thus, data requirements are extensive and can be limiting.

Last, the approaches presented above are clearly exploratory research tools and will remain as such until science can address the qualitative aspects with quantitative information. The reviews indicate that the ability of the approaches to examine small shifts in a system is problematic, especially given the data demands for biomass and rate requirements. Thus in systems with small to no expected perturbation from a single source such as the MWRA outfall, the ability of an approach to quantitatively predict subtle changes is questionable.

5. FOOD WEB MODELING IN RELATION TO THE OUTFALL

This review and reassessment was designed to address two questions that are included in the right whale food web scope of work (Hunt *et al.* 1999) submitted to EPA for review and comment. The section is organized around the two questions: "Will environmental conditions worsen as a result of the outfall relocation?" and if so "Is such change likely to harm whales?"

5.1 Will environmental conditions worsen as a result of the outfall relocation?

To address this question several approaches were pursued. The first was a review of the recent monitoring data to determine if conditions were different than assumed under the EPA SEIS (EPA 1998), the Biological Assessment (EPA (1993) and Biological Opinion (NMFS 1993). The second was to compare dilution fields and expectations based on more recent 3-D modeling. The third approach was to perform sensitivity and mass balance modeling using the calibrated BEM model to determine expectations for changes in nutrient fields and plankton biomass as measured by chlorophyll. These approaches are used in this section to evaluate whether or not food web modeling would provide a clearer picture of expected changes than can be derived from the available data and assessment approaches. These lines of evidence indicate the following:

1. Present nitrogen loading from the MWRA treatment plants is less than assumed in 1988.

2. The Deer Island effluent contributes a small fraction (~3%) of the total nitrogen entering the system.

3. Nitrogen entering at the boundaries of Massachusetts Bay exerts more influence on the total nitrogen concentrations in the farfield areas than the effluent discharge does.

4. BEM and 3-D hydrodynamic model results demonstrate that nutrient concentrations above the background variability will be confined to a small area near the outfall.

5. Elevated nutrient levels in the coastal region (from Boston Harbor southward towards Plymouth) will be unchanged or slightly lower with transfer of the effluent discharge location to Massachusetts Bay.

6. BEM model results predict little change in spatial or temporal patterns of nutrient concentrations in Cape Cod Bay relative to the current and future effluent discharge locations.

7. 3-D hydrodynamic model computations estimate the area in Massachusetts and Cape Cod Bays that would be under measurable influence from the discharge is small (only 7 km^2 which is <0.2percent of the combined area of Massachusetts and Cape Cod bays).

8. 3-D hydrodynamic model computations predict that the effluent nutrient concentrations will be diluted to 200:1 within a few kilometers of the outfall diffuser, and thus will be indistinguishable from background.

9. Change in the nutrient fields in Massachusetts Bay will be highly localized and have little to no impact on the phytoplankton and zooplankton species distributions and communities in the Bay.

10. Nutrient levels in Massachusetts and Cape Cod Bays will not be enriched to levels that promote the growth of nuisance species such as the “red tide” organism *Alexandrium*..

11. BEM computations project small increases in the DO in bottom waters of the nearfield in the summer.

These results are similar those found in the various ecological assessments completed for the new the MWRA outfall and often indicate the conclusions and projections in these reports were conservative. Thus, the new data from the monitoring program indicate that the environmental conditions in Massachusetts Bay will not be worse than projected and in fact will likely show even less change that previously thought.

5.2 Is such change likely to harm whales?

Based on the review above, model predictions indicate adverse changes to the ecology and functioning of the Massachusetts Bay system will not occur as a result of the outfall relocation. The fact that ecological impact may be less and have less spatial extent than projected in the various environmental assessments further argues that no net change will occur in the system after relocation.

The newer information indicates any changes that occur as a result of the relocation of the MWRA outfall will be confined to locations very near the outfall. The major farfield area affected will be Boston Harbor where the effects from nitrogen loading are expected to lessen. As a result, chlorophyll levels in the harbor are expected to decrease and dissolved oxygen levels in the inner harbor to rebound to high concentrations. Planktonic communities (either biomass or species distributions) in Massachusetts Bay are not expected to change as a result of the relocation. Plankton communities in the Cape Cod Bay and Stellwagen Bank areas are also not expected to change as result of the relocation. Thus, in the food supply shifts (either species or abundance) of the right whale are not expected. This species responds to many factors and conditions; most of these are external to the Bays. Therefore, because the nutrient inputs, concentration, and distribution, and plankton distributions will not change with the relocation, it is unreasonable to assume that detrimental effects on the occurrence of the whales will occur.

Moreover, the development of a food web model that endeavors to link the outfall discharge to the occurrence of right whales in the Bays would likely be an exercise in futility. The futility arises from several factors. The first is that these food web models are most effective when addressing measurable perturbations in a system, and such perturbations are not expected to result from outfall relocation. The second is the requirement that the food web models have complete and accurate species-by-species biomass information. This set of data is difficult to obtain and its accuracy cannot be easily ascertained. The third is uncertainty in the overall importance of the Bays to the energetics of the whales (i.e., inability to close the food web model domain). The fourth is that food web model development at a local or habitat specific scale is unwarranted given the importance of external factors that affect the distribution of the whales. As identified in a 1998 workshop convened to address knowledge of right whale distribution and predictability of the whale distribution (Clapham 1998), much research must be conducted to understand the factors that affect the population and its distribution. It is clear from the discussions and conclusions of this workshop that federal research dollars must be made available to address the fundamental questions raised. These questions must be addressed before predictive models can be developed.

The recommendations in Clapham (1998) provide a clear set of research and modeling directions related to the right whale and its occurrence in not only Massachusetts Bay, but over its entire range. Thus, funding of the key research and modeling needs identified from the workshop, which are more likely to fill the integrated long-term, large-scale research demanded for the overall management of right whales, is recommended. Moreover, the clear large scale spatial issues related to the protection and management of this species points to the need for broader agency involvement (federal, regional, and state levels) to effectively address the pressing issue of the salvation of the northern right whale population.

5.3 Applicability of food web modeling to the whales and outfall

The evaluation in this document show that under their present status, available food web modeling approaches are not likely to identify subtle changes in the ecosystem. Thus their successful application to the issue of right whale occurrence in the Bays is highly questionable given the factors that must be addressed before the modeling could go forward with confidence. The reviews above do not however, exclude the approach as a research tool, rather they point to the uncertainties and lack of knowledge on basic food web modeling elements. Thus, they are most effectively used today, as research tools not a management tool. Given the reviews it is instructive to summarize the data requirements and needs of the modeling approaches. This is provided in the next section.

5.4 Data needs for food web modeling

The modeling reviews above clearly indicate several data needs for effective food web modeling. These include:

1. Species by species biomass estimates
2. Population biomass estimates
3. Adult species body weight all species
4. Prey or trophic interaction data
5. Dietary information for species of interest
6. Dietary information by life stage
7. Harvest rates by species
8. Nutrient linkages
9. Whale ranges
10. Aggregation dynamics for whales (and other species)
11. Physical and biological factors resulting in prey aggregation (patch development)
12. Spatial structure (physical and biological)
13. Prey and whale energetics data

Given the outcome of this review, it is recommended that food web modeling not be pursued by the MWRA. Rather, the summary and recommendations in Clapham (1998) provide a clear set of research and modeling directions related to the right whale and its occurrence in not only Massachusetts Bay but over its entire range. Thus, funding of the key research and modeling elements from that workshop are more likely to fill the integrated long-term, large-scale research demanded for the overall management of right whales.

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