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# Ecosystem Indicators and an Environmental Score Card for the Lake Champlain Basin Program

Lake Champlain Basin Program

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# Ecosystem Indicators and an Environmental Score Card for the Lake Champlain Basin Program

Final Report to the
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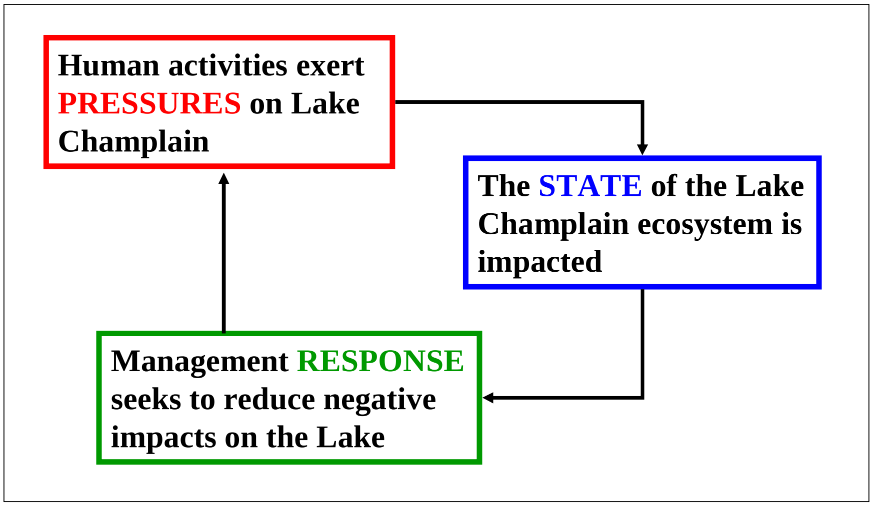


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# EXECUTIVE SUMMARY

The goal of this project was to develop a comprehensive suite of ecological indicators that are scientifically defensible and consistent with the goals and objectives of *Opportunities for Action*. Successful ecosystem management must be based on a comprehensive monitoring data base, but these data must be collected within a framework that allows the information to be used in decision-making. One of the most widely used frameworks to assist decision-makers is the Pressure-State-Response (PSR) framework initially developed by the Organisation for Economic Co-operation and Development. This framework is based upon the notion that human activities exert **pressures** on the environment that result in changes in the **state** of the environment and its natural resources. When these changes are detrimental, they trigger a societal **response** in the form of management policies and actions that are designed to reduce the environmental pressures and bring the state of the resources back to the desired condition. These relationships can be captured in simple cause and effects diagrams. When ecological indicators are selected for a monitoring program using this framework, they can subsequently be used as performance measures for a comprehensive management or action plan. A thoughtfully-selected set of indicators can both track environmental conditions and provide information to judge the effectiveness of the management actions that are implemented over time.

Because the complexity of the Lake Champlain ecosystem, like any ecosystem, is too great to be captured in a single PSR diagram, we selected indicators using a series of PSR diagrams that draw upon the goals in *Opportunities for Action*, the Lake Champlain Basin Program's Comprehensive Pollution Prevention, Control, and Restoration Plan, and that collectively represent the larger lake and basin ecosystem. Each PSR diagram addresses an issue area central to the health and management of Lake Champlain and represents a logical grouping of components of the ecosystem. The issue areas we selected were phosphorus pollution, fecal coliform bacteria, mercury toxicity, the sport fish community, the plankton in the pelagic food web, water chestnut and recreation. About 65 indicators were recommended across all seven of these issue areas.

Each indicator we have recommended is either ecologically relevant, meaning it relates to essential components of the ecosystem and responds to relevant stressors, or socially relevant, meaning it is linked to the concerns of the public and the government. It is also measurable, statistically sound (to the extent that we could evaluate this), and interpretable. To make an indicator interpretable, we must define an acceptable level for it. Without such information, an indicator cannot be used to evaluate whether the ecosystem is in a healthy or desired condition or in an unhealthy or undesired condition. Although current limitations in our ecological and social understanding can make defining these levels challenging, the desired ecosystem state, expressed through the goals of the management plan, can help define these levels. Sometimes acceptable levels can be defined using ecological thresholds, while in other cases they involve social preferences and policy decisions. For those data that have been collected for several years, we used modeling scenarios, sensitivity analysis and other techniques to describe the natural variability range, characterize real differences, and to quantify uncertainty.

To characterize public preferences, we used social norm curves and stated choice analysis, techniques used extensively in marketing and socio-economic analyses, to assess preferences for alternative states of the Lake Champlain ecosystem. We described these states by a set of ecosystem characteristics (or attributes) selected based on *Opportunities for Action* and focus group meetings. The characteristics included intensity and duration of algae blooms, number of days of beach closures, fish consumption advisories, land use distribution in the Lake Champlain basin, and the spread of water chestnut in the lake.

The levels of these characteristics were varied systematically to create alternative descriptions of Lake Champlain. A series of paired comparisons of alternative states of the Lake Champlain ecosystem were created and incorporated into a survey. Survey respondents had to make trade-offs among various characteristics and chose the alternative that they preferred. The choices of all respondents were then aggregated and used to estimate preferences about ecosystem attributes using a logistic regression model. These preferences were used to suggest acceptable levels of some of our indicators.

For phosphorus, we also developed a dynamic mass-balance model using the *Stella* modeling software. The model is designed to be run on a lake segment by lake segment basis. Preliminary models were developed for the Main Lake, Shelburne Bay, and Missisquoi Bay lake segments. The model tracks phosphorus through components of the Lake Champlain ecosystem, allows changes in state variables relative to changes in phosphorus inputs, and is linked to simple measures of management efforts and recreational use and enjoyment of the lake. The model shows which factors have the greatest influence on phosphorus concentration in the lake, and which factors are most responsive to management manipulation. The linkages in both the PSR diagrams and the phosphorus model are also our working hypotheses of the important relationships in the Lake Champlain ecosystem. As further data are collected, these hypotheses can be tested and further refined in future studies.

Recommendations are also provided for an environmental scorecard for the Lake Champlain Basin Program. This scorecard is meant to become a key part of the public education efforts associated with the program. It should present the indicators within the pressure-state-response framework to reinforce cause and effects relationships in the ecosystem, and clearly articulate the condition, or health, of the ecosystem, and progress towards the management goals and objectives outlined in *Opportunities for Action*. Individual scoresheets can be used to summarize trends in indicators in the issue areas, and to summarize important issues in the various lake segments.

Through our analyses, we discovered that the current monitoring efforts on Lake Champlain are fragmented, and sometimes without clear objectives. Data were often hard to track down, difficult to get, and in a form that required enormous effort in order to extract, summarize and use for analysis. We believe that there would be substantial benefits in examining these programs in light of what they are accomplishing, both individually and in combination. The result of such an examination might be a streamlined monitoring program with significantly greater impact.

Our proposed indicator suite includes a variety of parameters that are currently being measured by partner agencies in the Lake Champlain Basin Program, but also many parameters that are not currently being measured. These include biological and ecological parameters, social measures and economic measures. Clearly economic considerations are essential in the decision-making process, and capturing the links between ecological goods and services and their economic values through a series of economic indicators would add significantly to the overall indicators program for Lake Champlain. We also know that natural factors and stochastic events like severe storms and drought can also influence ecosystem condition and the outcome of management. We have not included a full set of indicators to track natural factors in all our proposed indicator suites, but these probably should be included in the future. The influence of these factors can also be considered using scenarios and modeling.

Finally, we believe it would be both short-sighted and naive to move forward with an indicators program without also investing in additional process-oriented research, to provide a context for understanding and interpreting the monitoring data. Additional research is needed in all the issue areas and priorities for research funds should be established at the same time that an indicators program is endorsed and implemented.

We recommend the following specific steps to follow up on this work and continue the development of an ecological indicators program for the Lake Champlain Basin Program:

1. Convene both technical and policy-level workshops to consider the information in this report and select an initial set of indicators for implementation.
2. Revise the current monitoring programs among the partner institutions in the LCBP in order to collect the data necessary to implement the indicators program. We suggest particular attention be paid to improving current data collection to maximize its utility, and adding indicators in the phosphorus, sport fish, and pelagic food web issue areas.
3. Establish acceptable levels for the state indicators in the Lake Champlain ecosystem indicators program as soon as practical. Use these levels as a basis for defining acceptable levels of the pressure and response indicators over time.
4. Continue to explore the linkages between issue areas in *Opportunities for Action* and explicitly consider a set of indicators that can capture those linkages.
5. Add socio-economic indicators in core issue areas.
6. As part of a regular “state of the lake” report, publish a first scorecard for a core set of indicators as soon as practical, and commit to biannual updates of the scorecard and state of the lake report.

As the LCBP moves forward with an ecological indicators program, we believe it is imperative that people be considered an explicit part of the Lake Champlain ecosystem; it is not possible to manage ecosystems as if we are external to them, or continued environmental degradation will be the most likely, if unintended, result. A thoughtfully implemented ecological indicators program can support more holistic management by providing the data and frameworks necessary to inform the critical choices that must be made.

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## TABLE OF ABBREVIATIONS

| Abbreviation | Definition |
|---|---|
| ANS | Aquatic nuisance species |
| BMP | Best Management Practice |
| LCBP | Lake Champlain Basin Program |
| LCFWMC | Lake Champlain Fish and Wildlife Management Cooperative |
| NOAA | National Oceanographic and Atmospheric Administration |
| NY DEC | New York Department of Environmental Conservation |
| NY DOH | New York Department of Health |
| NYS SWCC | New York State Soil & Water Conservation Committee |
| QC ME | Quebec Ministry of the Environment |
| SLU | St. Lawrence University |
| SUNY | State University of New York |
| TFM | Chemical lampricide (3-trifluoromethyl 4-nitrophenol) |
| TMDL | Total Maximum Daily Load |
| USGS | United State Geological Survey |
| UVM | University of Vermont |
| VCGI | Vermont Center for Geographic Information |
| VT AFM | Vermont Agency of Agriculture, Food and Markets |
| VT DEC | Vermont Department of Environmental Conservation |
| VT DOH | Vermont Department of Health |
| VT FW | Vermont Department of Fish and Wildlife |

# I. INTRODUCTION

The concept of an indicator is familiar to most people even if they do not recognize the term. That familiarity comes from an annual checkup at the doctor's office or even a read of the business section of the newspaper. To assess state of health, a physician might measure blood pressure, take a pulse and temperature, and draw a blood sample to see how much iron or cholesterol is in it. These measurements are all indicators that paint a picture of overall condition and tell the doctor whether any treatments are necessary. The business pages of the newspaper report the Dow Jones Industrial Average, the inflation rate, the unemployment rate, and at the end of the year, perhaps the gross domestic product (GDP). To an economist or business leader, these measurements indicate the state of the national economy and help guide investments and other financial management decisions. In both of these examples, a few measurements are used to represent a myriad of other things that have not been examined.

In a similar way, ecosystem indicators are also increasingly being used to help inform environmental management and decision-making. An ecosystem indicator can be any measure that provides information about the condition of the ecosystem or the effectiveness of management. In the same way that it is impossible to examine all aspects of human health and the economy, it is also impossible to measure and evaluate all the critical processes and components in an ecosystem. Therefore, ecological indicators are selected and used to determine the condition of the ecosystem with a reduced set of measures that represent or "indicate" its overall state.

The Lake Champlain ecosystem is a large and complex array of places including not just the lake itself, but also all the land that drains into the lake, the wetlands and streams that hold and carry water to the lake, and the urban/built environment where people live and work. It is also a managed ecosystem, a single natural entity that is shared between two countries and many communities. Before 1990, a variety of independent federal, state, and provincial agencies and institutions with diverse interests and expertise were implementing a variety of management programs on the lake and in its watershed with varying degrees of coordination. In November of 1990, the United States Congress passed the Lake Champlain Special Designation Act, which led to the creation of the Lake Champlain Basin Program (LCBP) in the following year. This group and its "Management Conference" were charged with coordinating and facilitating the management of Lake Champlain. They were also directed to develop a first ever comprehensive management plan. In 1996, that plan, *Opportunities for Action*, was completed and released to the public.

As the actions in this comprehensive management plan were implemented, the LCBP needed a way to track progress and make better use of the data collected in its various monitoring programs. The LCBP funded this project, the Lake Champlain Ecosystem Indicators Project, to develop a set of ecosystem indicators for the Lake Champlain Basin Program. Indicators were needed both to evaluate the condition of the Lake Champlain ecosystem and to evaluate the effectiveness of the management actions being implemented as part of the comprehensive plan. In Phase 1 of the project, we reviewed the literature and proposed a conceptual approach and a list of candidate indicators for the LCBP. In Phase 2 of the project, we applied our approach and gathered as much information as possible to evaluate potential indicators. In this report, we present a final set of recommendations for a Lake Champlain Ecological Indicators Program.

Evaluating the condition of the Lake Champlain ecosystem is a difficult task given the complexity of the ecosystem, the multitude of pressures derived from human activities in the basin, and the many ways that management actions influence ecosystem response. Clearly, no single environmental measure can accurately reflect the condition of an entire ecosystem (Bertram and Stadler-Salt 2000, Jackson et al. 2000, The Heinz Center 2002). Therefore, a suite of indicators must be developed that collectively represents the condition of the ecosystem. Comprehensive indicator suites should be able to document the extent to which ecosystems are affected by human pressures (Hughes et al. 1992) and evaluate how ecosystem structure, processes and organization may be changing as a result of those pressures (Müller et al. 2000). In this project, we present a suite of indicators that illustrate, to the extent possible given the data that are currently available, the condition of the Lake Champlain ecosystem and its response to management. We present our indicators within a conceptual framework that incorporates at least a portion of the complexity of the system.

Ecosystem indicators are valuable management tools from many perspectives. The information they provide can increase understanding of the environment, enhance program planning and guide resource distribution (Sylvester 2001). By systematically separating acceptable measured values of indicators from unacceptable values, managers can identify those ecosystem components that require management attention (Kelly and Harwell 1990). Acceptable levels are explicit statements of the desirable ranges of measured values for ecosystem indicators. They should reflect management goals, scientific understanding and social values.

The establishment of acceptable levels ensures a predictable and systematic evaluation of ecosystem condition and management effectiveness. If the measured value of each indicator is within its range of acceptable levels, the ecosystem is assumed to be in a healthy or desired state. When the measured value of an indicator is outside its range of acceptable levels, management intervention is required. Without information about acceptable levels, an indicator is not interpretable, that is, it cannot be used to make management decisions. Unfortunately, the failure to define acceptable levels is a common shortcoming of many indicator initiatives.

Most management actions are implemented in an effort to maintain or enhance ecosystem condition. Indicators that are linked directly to specific management objectives and actions or to ecosystem components of direct societal concern (Kelly and Harwell 1990, Landres 1992, Rapport 1992, Water Quality Guidelines Task Group 1996, Jackson et al. 2000, Zorn et al. 2001, Frid 2003, Vaughan et al. 2003) can be used to help evaluate the effectiveness of these management actions and progress towards achieving management goals. Although many indicator programs have been developed to evaluate ecological conditions, many of these programs do not take the extra step of linking to management actions so that interventions can be evaluated. However, when these linkages are made, indicators can provide systematic information about the ecosystem that reduces uncertainty (Hammond et al. 1995) and facilitates better decision-making (Bernstein 1990), particularly when implemented within an adaptive management framework (Holling 1978, Lee 1993).

Adaptive management incorporates experimentation into the design and implementation of ecosystem management policies (Holling 1978, Lee 1993). By adopting an adaptive management approach, policy-makers acknowledge that their understanding of the ecosystem and the potential outcomes of management actions is incomplete (Noss et al. 1997). Consequently, management actions are designed to test hypotheses and predictions about how managers and scientists think the ecosystem is working. Using this approach, changes in measured indicator levels provide information about not just the condition of the ecosystem and the effectiveness of the management actions in achieving the desired results, but also the validity of the hypotheses upon which the management actions were based. By organizing indicators in an adaptive management approach, managers have the opportunity to increase their understanding of the ecosystem regardless of the outcomes of management, which increases the potential success of future management strategies.

Furthermore, resource managers can simplify the process of communication among scientists, policy makers and the public by selecting and using indicators that can be understood and appreciated by a broad audience (OECD 1993). Indicators that relate to social values (Hess et al. 2000), ecosystem components of direct societal concern (Kelly and Harwell 1990) and ecosystem services (Rapport 1992) can be effective educational tools for building public support for the management actions designed to improve environmental conditions. Therefore, all indicator programs need a mechanism for reporting results to stakeholders and decision-makers (Water Quality Guidelines Task Group 1996, Harwell et al. 1999).

**Defining Ecosystem Indicators and Frameworks**

The literature on environmental, ecological and ecosystem indicators is large and growing. This literature appears in published technical journals, books, government documents, and increasingly, on websites created by programs implementing indicator projects. Hunsaker and Carpenter (1990), in a review and analysis conducted for the US Environmental Protection Agency (US EPA), define an indicator as a "characteristic of the environment that when measured, quantifies the magnitude of stress, habitat characteristic, degree of exposure or degree of ecological response to exposure." Many organizations have adopted this definition (e.g., Council of Great Lakes Research Managers 1991, Water Quality Guidelines Task Group 1996). Müller et al. (2000) define indicators as "variables or indices that represent, integrate and characterize information embodied in comprehensive data sets which are often not measurable directly." For the Great Lakes Basin Ecosystem Health project (Bertram and Stadler-Salt 2000) an indicator is a "parameter or value that reflects the condition of an environmental (or human health) component, usually with a significance that extends beyond the measurement of the value itself." While the term "indicator" is frequently redefined on a case-by-case basis, ecological indicators are essentially a shorthand method for describing and evaluating the structure and function of an ecosystem (Hughes et al. 1992, Landres 1992) and the way the ecosystem responds to management actions.

Ecosystem indicators range in complexity from measurements of simple environmental characteristics like water quality to multimetric indices of condition, such as Karr's Index of Biotic Integrity (IBI) (Karr et al. 1986). As ecologists have become more aware of the complex, long-term and cumulative impacts of human activities on the environment, most environmental management and monitoring initiatives that use indicators have evolved from simple stress-specific measurements to more holistic efforts to evaluate the broader condition of entire ecosystems (Nip and Uno de Haes 1995, Karr 1996, Fisher 1998, Wickham et al. 1999, Hess et al. 2000). How well the ecosystem is described by the selected suite of ecological indicators depends on the quantity and diversity of the indicators and the quality of the data used to develop those indicators. Given limitations in institutional resources, indicators must be selected to maximize unique and relevant information with minimal redundancy (Council of Great Lakes Research Managers 1991). This can be accomplished through the use of a framework to organize the indicator selection process (Council of Great Lakes Research Managers 1991) and provide a context for evaluating the data that are collected (Hughes et al. 1992).

One of the most widely used and adapted indicator frameworks (e.g., Hammond et al. 1995, Crabtree and Bayfield 1998, SDI Group 1998, Bertram and Stadler-Salt 2000 and Müller et al. 2000) is the Pressure-State-Response (PSR) framework developed by the Organisation for Economic Co-operation and Development (OECD) (OECD 1993). This framework (Figure 1) is based upon the notion that human activities exert pressures on the environment that result in changes in the state of the environment and its natural resources. These changes elicit a societal response by changing environmental policies or implementing management actions (OECD 1993). The societal response creates a feedback loop to the environmental pressures by promoting changes in the human activities that caused the environmental degradation in the first place (OECD 1993). Responses can also be focused directly on changing the state of the environment. Because environmental management is a young field and response indicators have not been used for as long as pressure and state indicators, the number of response indicators can be limited by data availability and by human understanding of management outcomes (OECD 1993).

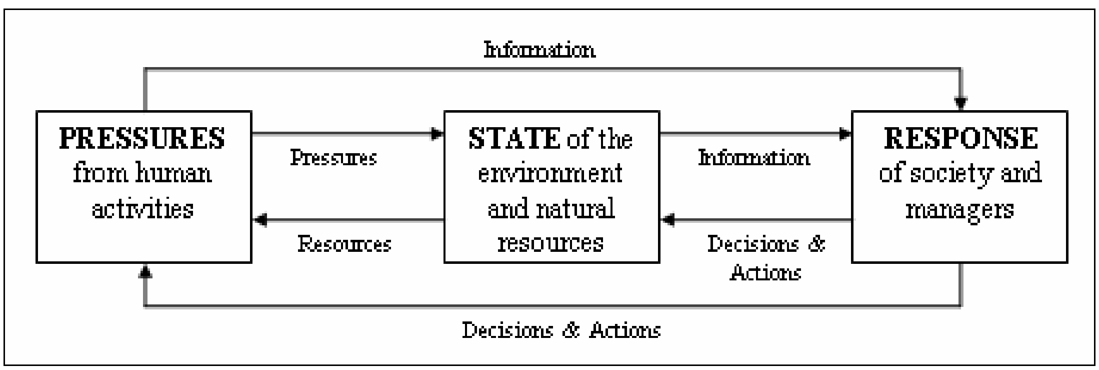



**Figure 1: Pressure-State-Response Framework from OECD (1993).**

## Types of Indicators

There are many classification schemes for indicators (Table 1), including a variety of types and categories of ecological response measures developed for diverse management purposes. There are numerous recommendations in the literature for including certain indicator groups when developing a suite of indicators. Several authors emphasize the importance of early warning indicators because expedient management intervention before serious degradation occurs can eliminate the need for expensive rehabilitation or restoration (Kelly and Harwell 1990, Rapport 1992, Nip and Uno de Haes 1995). These indicators might be measures that respond rapidly to a spectrum of pressures, or measures that are sensitive to specific, expected pressures. Based on field studies that show that structural indicators respond faster than process indicators because of the redundancies in ecological systems (e.g., Schindler 1987), several authors counsel using structural indicators in any program focused on monitoring for ecological change (Hughes et al. 1992, Angermeier and Karr 1994). Nip and Uno de Haes (1995) also stress the importance of structural abiotic indicators because they play an intermediary role between pressures and biological responses, and therefore, should be able to predict changes in biotic components of the system prior to their occurrence. In addition to being more responsive, abiotic elements are generally better understood than the underlying ecological processes and they are less expensive to monitor (Angermeier and Karr 1994). Using a strictly ecological perspective, Angermeier and Karr (1994) recommend selecting indicators that can capture: (1) physiochemical conditions, (2) trophic base, (3) habitat structure, (4) temporal variation, and (5) biotic interaction.

**Table 1: Types of indicators described in the literature.**

| **Type** | **Definition** |
|---|---|
| Structural | Measures the biotic and abiotic components of the system. |
| Process/Functional | Measures changes in rates and pathways of ecosystem processes. |
| Disturbance | Measures disturbance regime responsible for maintaining the ecosystem structure. |
| Compliance/ Management | Evaluates progress towards the attainment of management goals and objectives. |
| Diagnostic/ Sensitive | Provides insight into the specific cause of ecosystem change or noncompliance. |
| Early Warning | Quickly identifies changes in system condition prior to substantial degradation to allow for management intervention. |
| Screening | Provides general assessment of system condition and detects signs of ecosystem stress. |
| Intrinsic Importance | Measures aspects (species or process) of the system that are of direct value, economic or otherwise. |
| Pressure | Characterizes direct and indirect impacts from human activities. |
| State/Condition | Measures environmental condition and the quantity and quality of natural resources. |
| Response | Measures societal or management response to the ecosystem state. |
| Sensitivity | Measures ecosystem response to stress or the ability to recover from disturbance. |
| Long-term | Detects ecosystem change or degradation over larger spatial and temporal scales. |
| Index | Combines several variables into an overall measure of ecological condition. |

Adapted from Kelly and Harwell 1990, Council of Great Lakes Research Managers 1991, Landres 1992, Karr 1992, Rapport 1992, OECD 1993, Nip and Uno de Haes 1995, Water Quality Guidelines Task Group 1996, Harwell, et al. 1999 and Hess, et al. 2000.

Several authors have created multimetric indices to assess ecosystem integrity. Perhaps the most widely used of these indices, Karr's Index of Biotic Integrity (IBI), uses 12 measures associated with fish communities to arrive at a score that indicates how much a given stream deviates from an undisturbed or reference condition (Karr et al. 1986). The IBI has since been adapted to different geographic regions and modified to create new indices like the Benthic Index of Biotic Integrity (B-IBI) based on benthic invertebrates instead of fish (Kerans and Karr 1994). Other examples of commonly used indices include the U.S. Environmental Protection Agency's Rapid Bioassessment Protocols (RBP) and Ohio's Invertebrate Community Index (ICI), which were developed to assess water quality in streams and rivers quickly over large geographic areas at a relatively low cost (Watzin and McIntosh 1999). Both of these indices primarily use metrics related to organism tolerances but the RBP also includes measures of habitat quality and invertebrate functional groups. The Hilsenhoff Biotic Index (HBI) (Hilsenhoff 1988) is another commonly used index that is based solely on organism tolerances and abundance.

Although all of these indices are used widely, Schaeffer et al. (1988) and Suter (1993) warn that assessment of ecosystem health should not be based upon a single reductionist index or even an inventory of species. Results from surveys to measure indices are widely variable and it is generally unclear which component metrics are driving the resulting index score (Watzin and McIntosh 1999). Xu, et al. (1999) argue that indices are too narrow in scope to adequately reflect ecosystem complexity. Ecosystem condition must be contextually examined, including the type of ecosystem and the types of pressures affecting the system (Kelly and Harwell 1990).

Because the different types of indicators have varied purposes, the suite of indicators should be as comprehensive as possible and include many different types of indicators (Kelly and Harwell 1990, Hughes et al. 1992, Water Quality Guidelines Task Group 1996). Many environmental variables may be classifiable as several indicator types depending on their properties (Kelly and Harwell 1990). For example, the measurement of an ecosystem process, such as the rate of primary productivity, may be both a functional indicator and an early warning indicator if the measurement responds quickly to anthropogenic changes. To ensure detection of change as soon as possible, several authors suggest that indicator suites include measurements that span multiple levels of biological organization (Spellerberg 1991, Karr 1992, Water Quality Guidelines Task Group 1996). Likewise, Kelly and Harwell (1990) recommend including indicators that are responsive over a range of spatial and temporal scales. Because management concerns, social values and scientific understanding change overtime, it is important that any indicator suite be dynamic, flexible and adaptable to changing local conditions (Haskell et al. 1992, Rapport 1992, Water Quality Guidelines Task Group 1996, Fisher 1998, Watzin and McIntosh 1999, Bertram and Stadler-Salt 2000).

### Desirable Indicator Characteristics

To ensure that all indicators are of high quality, selection should be based upon a set of predetermined criteria (Water Quality Guidelines Task Group 1996) that take into account the ecological, social, economic and institutional considerations of the particular application (Hunsaker et al. 1990). There are many characteristics that have been recommended for consideration in indicator selection (Table 2). Likewise a variety of approaches to using these characteristics have been suggested (Kelly and Harwell 1990, Water Quality Guidelines Task Group 1996, Fisher 1998, Manley et al. 2000). These approaches emphasize the need to balance technical, operational, and administrative considerations. They also stress weighing the benefits of available historical data and the ability of the indicator to discriminate environmental differences in the face of considerable natural variation. Ideally, indicators should be the best possible environmental measures, not just characteristics for which data are already available (Water Quality Guidelines Task Group 1996).

**Table 2: Characteristics that can be used to select indicators.**

| **Characteristic** | **Explanation** |
|---|---|
| Ecologically Relevant | Relates to essential components of the ecosystem and responds to relevant pressures. |
| Politically/Socially Relevant | Linked to the concerns of the public and the government. |
| Measurable | Can be quantified using a standard procedure with documented performance and low measurement error. The collection of the data should not harm or alter the ecosystem. |
| Interpretable | Able to distinguish acceptable from unacceptable conditions; natural variability understood. |
| High Sensitivity | Has a high signal-to-noise ratio with known variability. |
| Statistically Sound | Able to document trends and show significant differences. |
| Related to Management Action | Allows for determination of current policy effectiveness and adaptive management. |
| Understandable | Public able to understand the value of the indicator. |
| Specific | Response consistently corresponds to a specific stressor. |
| Historical Data Available | Enhances potential for trend analysis and understanding natural variation. |
| Cost-effective | Measurement expense corresponds to quality of information gained and the institutional needs. |
| Responsive to Multiple Pressures | Broadly identifies impact/degradation of the ecosystem. |
| Appropriate Scale | Spatial and temporal scales of measurement are scientifically justifiable and correspond to management concerns. |
| Timely | Responds quickly to allow for management action. |
| Highlight Emerging Problems | Able to identify when anticipated problems arise in the ecosystem. |
| Unique | Provides unique information to indicator suite. |
| Provides New Information | Provides new information regarding ecosystem structure and function that increases scientific understanding of the system. |
| Operationally Feasible | Data collection methods are appropriate and feasible considering available personnel and equipment. |

Adapted from: Hunsaker, et al. 1990, Kelly and Harwell 1990, Spellerberg 1991, Hughes, et al. 1992, Karr 1992, Rapport 1992, OECD 1993, Angermeier and Karr 1994, Hammond, et al. 1995, Water Quality Guidelines Task Group 1996, Fisher 1998, Whitford 1998, Harwell, et al. 1999, Jackson, et al. 2000, Manley, et al. 2000, Müller, et al. 2000.

While many indicator initiatives simply present a set of data trends without a context for interpretation, some efforts have acknowledged the need to compare indicator values to an established benchmark. The OECD (1993) recommends determining threshold or reference values that can be used to gauge the significance of all indicator values, while Schaeffer et al. (1988) suggest defining an acceptable range of values that can be used to compare indicator measures between areas or over time. The acceptable levels for each indicator must be closely tied to the management goals (Water Quality Guidelines Task Group 1996). Rogers and Biggs (1999) describe a hierarchical, adaptive approach for managing rivers in Kruger National Park, South Africa. In their framework, managers and scientists translate a broad vision statement into increasingly specific management objectives and goals. Indicators, bound by thresholds of probable concern (TPCs), are measured variables used to evaluate the attainment of scientifically defined management endpoints (Rogers and Biggs 1999). TPCs are hypotheses of the limits of acceptable ecosystem change derived from the collective understanding of scientists and managers (Rogers and Biggs 1999).

In the realm of U.S. park and wilderness management, indicators of quality are used to evaluate crowding and other elements of the park visitor experience (Manning et al. 1996, Manning 1999, Manning and Lawson 2002). Indicators of quality are specific variables that are evaluated relative to standards of quality. These standards represent the threshold of visitor use that degrades park resources and the visitor experience beyond an acceptable level (Manning 1999). Visitor surveys are one approach used to generate appropriate standards of quality for the indicators that are selected (Manning and Lawson 2002).

The ecological (Rogers and Biggs 1999) and social (Manning et al. 1996) perspectives used to define acceptable levels in the two examples described above are very different. Rogers and Biggs (1999) advocate an ecological approach based on empirical data analysis and scientific understanding. Manning and his colleagues have emphasized the social perspective, focusing on techniques for defining acceptable levels using user surveys and the values they express. Because acceptable levels are an expression of the desired state of the ecosystem, the process of establishing acceptable levels should incorporate both ecological understanding and social values.

Most indicator suites focus on the state of the environment rather than the relationship between society and ecosystems (Azar et al. 1996). Indicators of environmental state illustrate changes in the ecosystem after they occur, therefore, they show when *past* societal activities were unsustainable *after* the ecosystem is degraded (Azar, et al. 1996). In a more proactive management approach, Azar et al. (1996) recommend developing indicators that focus on the ways society currently affect the environment, indicators that might provide managers with an early warning when ecosystem change is expected as a result of specific human activities.

To ensure an ecologically viable condition, the desired ecosystem state must be within the limits of what is scientifically defensible for a stable, healthy ecosystem (Nip and Uno de Haes 1995, Rogers and Biggs 1999). Because ecosystems can exist in multiple stable states (Holling 1973, Haskell et al. 1992, Rapport 1992), this does not pre-determine a particular ecological state.

Ecosystem managers can and do target socially desirable ecosystem conditions in order to maximize public support for their management efforts (Karr 1992, Rapport 1998, Harwell et al. 1999).

## II. PRESSURE-STATE-RESPONSE FRAMEWORK AND INDICATOR SUITES FOR LAKE CHAMPLAIN

Based on our review of the literature, we chose to adapt the Pressure-State-Response (PSR) framework as a basic approach to organizing and selecting indicators for Lake Champlain (Figure 2). As noted by Hammond et al. (1995), this framework is readily understandable by the public. Indicators that are organized as pressures, states, and responses clearly show what is happening to the state of the environment, why it is happening and what is being done about it. By categorizing the indicators into pressure-state-response groups, the relationships between human activities that act as pressures on the ecosystem, lake condition, and management responses that seek to reduce or mitigate these ecosystem pressures can be shown.

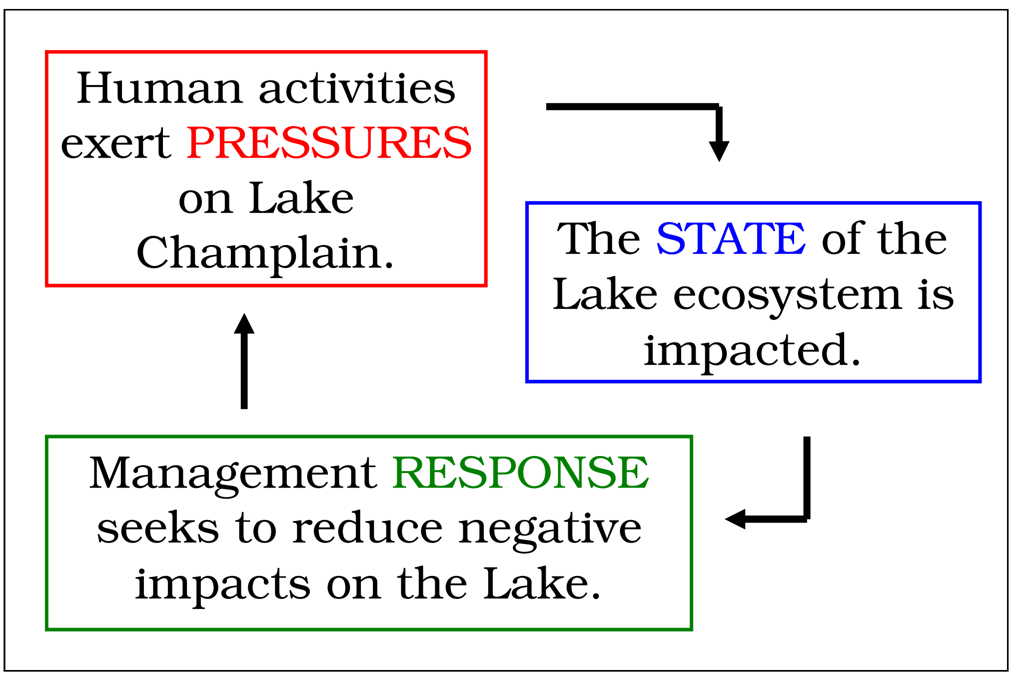



**Figure 2: PSR framework for the Lake Champlain Ecosystem Indicators Project.**

The PSR framework also serves as a conceptual model that can organize our understanding of the important causal linkages in the Lake Champlain Basin ecosystem. A conceptual model of an ecosystem is a working hypothesis of ecosystem form and function. It depicts relationships and documents assumptions about which processes and relationships are important (Manley et al. 2000). It can be used as a tool to direct the selection of indicators that relate to important processes and components (Kelly and Harwell 1990, Hughes et al. 1992, Nip and Uno de Haes 1995, Griffith 1998, Jackson et al. 2000, Manley et al. 2000). Such a model should be based on ecological theory and include known pressures from human activity and the ecosystem services provided to society (Hughes et al. 1992, Manley et al. 2000). At the same time, we recognize that there are limitations to the simple, linear cause-effect relationships in the PSR diagrams. Although they help us to conceptualize the Lake Champlain ecosystem, these simple models cannot fully explain the response of any ecosystem to natural and anthropogenic pressures.

Although the need for a framework is discussed in the literature, one of the many widespread shortcomings in other indicator efforts is the lack of such a conceptual framework. In these efforts, indicators are simply presented in lists, with an explanation of what each individual indicator is measuring. The indicators are not directly related to management actions or human activities, nor are the potential relationships among indicators acknowledged. We believe that it is essential that each indicator be presented within a conceptual model that explicitly relates the measure to our understanding of how the ecosystem is functioning. The model must also clearly present the cause and effect relationships that we believe are most important. Because our databases and scientific understanding are incomplete, these relationships will frequently be presented as hypotheses, rather than documented functional relationships. As data are collected as part of the indicators program, they can be used in an adaptive management context to test these hypotheses and to make appropriate adjustments as our knowledge base improves.

The importance of the conceptual model also becomes clear as we think about how to best present the indicators to decision makers and the public. Our models will provide a vehicle for explaining the indicator measures and their significance. They will also allow both our indicators team and LCBP managers to communicate clearly with appropriate outside expert groups, stakeholders in the basin, and others about the design of the monitoring program and ways that the data being collected can be used to inform management progress.

Because the levels of phosphorus (P) in the waters of Lake Champlain are known to impact many aspects of what is valued about the Lake Champlain ecosystem, the Lake Champlain Basin Program (LCBP) has identified the management of P as one of its highest priorities. Consequently, we also placed extra emphasis on phosphorus in our ecosystem indicators project. Phosphorus pollution affects many aspects of the lake ecosystem, so it is important to identify indicators that clearly relate to P dynamics in the lake. However, because of the complexity of the Lake Champlain ecosystem there is no single relationship (or indicator) that can capture overall changes in the state of Lake Champlain with regard to P. Indicators must be identified that track the amount of P in the lake, the rates at which P flows into and out of the lake, and the rates at which P is stored and used by the biota.

Detailed computer models can be helpful in identifying useful environmental indicators because these models provide a quantitative infrastructure for analyzing complex systems and a structured way of thinking, in this case, about how P behaves and influences the condition of Lake Champlain. We chose to model P because of its high priority in the LCBP, the relatively large amount of monitoring data that is available, and the existence of a relatively good, if still incomplete, understanding of how P behaves in lake ecosystems. This modeling approach could be applied to a number of other pollutants as well. The model we developed, the Phosphorus Environmental Indicator (PEI) model, is a dynamic mass balance procedure that accounts for all the major storage, inputs and outputs of P for a given lake segment. The model tracks, over time, changes in the magnitude of P storage (mass and concentrations) and inputs and outputs of P (loads). Inputs of P are influenced by human population levels, population growth rates, land conversion strategies, the overall character of land use in the watershed and the hydraulic connectivity of one lake segment with another.

**Guiding Criteria for Lake Champlain Indicator Selection**

Although many characteristics of good ecological indicators are presented in the literature, it is rare to see an articulation of these characteristics before an indicator program is established. For our Ecosystem Indictors Project, we focused on selecting indicators with the following characteristics:

- Ecologically relevant
- Politically/Socially relevant
- Measurable
- Statistically sound
- Interpretable

All of the Lake Champlain ecosystem indicators that we propose relate clearly to our PSR framework and are either ecologically relevant, meaning they relate to essential components of the ecosystem and respond to relevant stressors, or socially relevant, meaning they are linked to the concerns of the public and the government. They are also measurable, statistically sound (to the extent that we could evaluate this), and interpretable. Indicators that are both ecologically relevant and socially relevant have the greatest value; however, some indicators may fall into only one of these two categories and still be very valuable.

We also chose to focus on selecting indicators that could build, as much as possible, on data that are currently being collected by the partners in the LCBP. Most of the existing indicator projects that we have reviewed largely limit themselves to existing and often long running data sets when selecting potential indicators. Although there is enormous merit in using existing data because these data provide a perspective over time that helps to define natural variability and suggest acceptable levels, many important measures of ecological condition have not historically been collected.

In fact, there are very few long-term datasets available for Lake Champlain. Nonetheless, we started by gathering as many datasets germane to Lake Champlain in the focus areas as possible in order to evaluate current monitoring approaches and the state of the scientific understanding of Lake Champlain. In this report, we present indicator measures developed from available data and begin the process of suggesting more informative measures that would require alternative or additional data collection. Clearly, new monitoring efforts will have to be considered judiciously, but if we want a comprehensive program based on adaptive management and a full framework of understanding from an ecosystem perspective, the LCBP will need to consider new data collection.

## Acceptable Levels

In order to interpret trends and measured values, each indicator needs an explicitly defined range of acceptable levels that incorporates both ecological understanding and social values. The ecological definition of acceptable levels is based on a scientific assessment of the range of values for every indicator that collectively represent a functioning, viable ecosystem. In practice, however, specifying a range of values for ecosystem variables from a scientific perspective is very difficult. Ecosystems are complex and naturally variable. The definition of a range of acceptable levels must incorporate natural variability and uncertainty. Ecological thresholds, levels of ecosystem components that are irreversible or that trigger major changes in other ecosystem components, must also be considered. This can be difficult because monitoring data are often limited and scientific understanding about these complex interactions is generally incomplete.

For example, Lake Champlain is a naturally mesotrophic lake that is threatened by anthropogenic nutrient enrichment. Increasing the phosphorus concentration favors eutrophic phytoplankton species over mesotrophic species. These shifts in the phytoplankton community composition could have detrimental effects on the rest of the Lake Champlain food web, including recreationally important sport fish. On the other hand, if phosphorus concentrations are too low, primary productivity is limited, thereby providing less support for higher levels of the food web. The definition of acceptable levels for phosphorus concentration in Lake Champlain from an ecological perspective should be based upon maintaining a mesotrophic condition with sufficient primary productivity to sustain the food web. With about twelve years of consistent Lake Champlain phosphorus data and a limited understanding of the intermediate trophic levels, it is difficult to pinpoint an appropriate level of phosphorus from an ecological perspective at this time.

The social component of defining acceptable levels involves an understanding of the social values pertaining to the ecosystem indicators. It may be possible to define a narrower range of acceptable levels from a social perspective. This is the case with the phosphorus example given above. While it is unclear whether current phosphorus levels in many sections of Lake Champlain are an ecological hazard, most segments are above the levels that interfere with human use and enjoyment (Smeltzer and Quinn 1996). These levels were the basis for the in-lake phosphorus criteria. Therefore, our current acceptable levels are based on social preferences, not ecological considerations.

It is important to balance ecological and social considerations when establishing acceptable levels, selecting levels that ensure an ecologically sustainable system that also meets social expectations. Sources of information that can be used to help formulate acceptable levels include legal and administrative mandates (e.g., law), agency policy, historic precedent, interest group politics, personnel and financial resources, and public opinion (especially that gathered in a systematic way from members of the public who are most directly interested in and affected by resulting decisions) (Manning and Lawson 2002).

**Series of Issue Areas Selected for the PSR Diagrams**

Because the complexity of the Lake Champlain ecosystem, like any ecosystem, is too great to be captured in a single PSR diagram, we developed a series of PSR diagrams that draw upon the goals in the LCBP's management plan *Opportunities for Action* and collectively represent the larger lake system. Each PSR diagram addresses an issue area central to the health and management of Lake Champlain. We selected the issue areas based on the primary focus areas in *Opportunities for Action*, logical groupings of components of the ecosystem, and data availability. The issue areas we selected are phosphorus, bacteria, mercury toxicity, the sport fish community, the plankton in the pelagic food web, water chestnut and recreation. A PSR diagram was constructed for each issue area that explicitly identifies indicators of pressure, state and response. For the purpose of this study, pressure indicators represent the human activities that lead to ecosystem degradation (even though natural factors can also be pressures that drive ecosystem change), state indicators characterize the current condition of the ecosystem, and response indicators represent management actions taken to reduce the pressures and improve the condition of the ecosystem.

The PSR diagrams are conceptual models. As such, they are qualitative. They document our hypothesized understanding of the important ecological pathways for each issue area. By applying data to the conceptual model, we can look for corresponding trends amongst the indicators. For instance, if the data show a decrease in a pressure indicator, such as pollution loading, do we also see an improvement in a state indicator, such as a water quality measure? By collectively examining data trends within the PSR framework, we can develop an overall picture of how the ecosystem is functioning, and provide a more complete context for environmental decision-making. As more data become available, we can begin to examine these relationships in a more quantitative manner. For example, our PSR framework, or conceptual model for phosphorus shows that we expect an increase in phosphorus load with an increase in population. However, we do not know the magnitude of such a change. As data are collected over time we may be able to determine how great an increase in load we might expect per unit of population increase. These relationships are critical ones for managers to understand. They will allow specificity in determining level of effort and designing management approaches that have the greatest likelihood of achieving results.

Although the Lake Champlain Basin Program has been operational for over a decade, data are still very limited in many issue areas. The program's greatest monitoring effort has been devoted to phosphorus; therefore, we developed a quantitative model for this issues area. This model allowed us to explore our understanding of phosphorus dynamics in the lake. We also used this model as a case study of how a conceptual model can organize data collection that can then be used in a quantitative model to examine relationships between potential indicators and not just disconnected trends in individual indicators.

The PSR models, and the indicators, measures and available data that go with them are presented and discussed in the following sections. In each model, pressure indicators are shown in red, state indicators in blue and response indicators in green. Boxes represent indicators for which there are exiting data; circles represent indicators for which data are not currently available. One-way and two-way arrows are used to depict the linkages among the various indicators. Two way arrows represent known or hypothesized linkages with feedbacks; some single direction arrows may in fact have feedbacks too but the net flow is in the direction of the arrow. Because the indicators listed in the PSR models are very general, more specific measures were selected for each indicator. The selection of these specific measures was based first upon the data that are currently available. In some cases, the available data are less than ideal and in others there are no data available at all. In these instances, a recommended measure is provided. Some indicators appear in more than one PSR diagram, and many indicators are clearly relevant to more than one issue area. These overlapping indicators show the synergy between the issue areas. Because the various lake segments have different environmental challenges, often the PSR models must be considered separately for each lake segment. These considerations are highlighted where applicable.

## A. Issue Area: Phosphorus

Phosphorus is the nutrient that is presumed to most commonly limit the growth of phytoplankton in Lake Champlain. It has an important role in both the ecological dynamics and the human use and enjoyment of the lake; therefore, it forms the basis for the first PSR model (Figure 3). The LCBP has identified the management of phosphorus as one of its highest priorities. High levels of phosphorus in lake waters can alter the composition and density of both the plankton and macrophytes in the lake. This increase in primary production has myriad potential implications throughout the food web, right up through the fish community. Higher primary production can also limit light penetration, impair oxygen levels in deeper waters and lake sediments when this biomass decomposes, and greatly affect the desirability of the lake for swimming and boating.

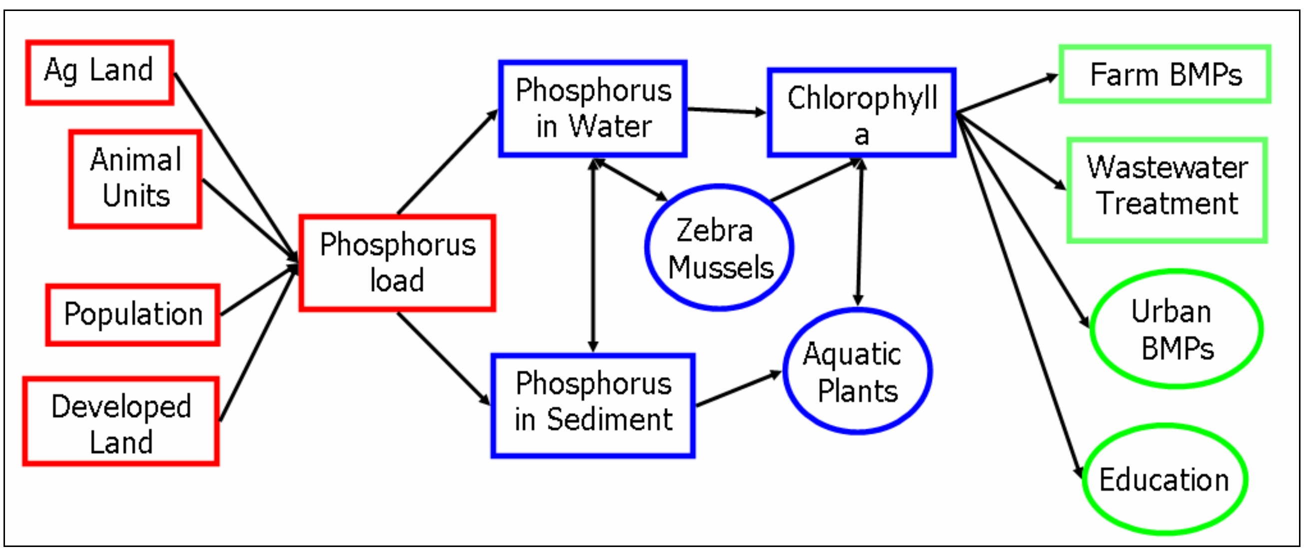



**Figure 3: PSR diagram for the phosphorus issue area.**

The PSR model for phosphorus specifies the major sources of phosphorus, the pressures, the important state variables with respect to the movement of phosphorus through the ecosystem, and the primary management activities implemented to reduce phosphorus generation and loading (Figure 3 and Table 3). The phosphorus concentrations in Lake Champlain are the result of the historical and current phosphorus load. The natural phosphorus load has been augmented by anthropogenic loading from human and agricultural activity in the basin. This excess phosphorus load is accelerating the natural process of lake eutrophication.

**Table 3: Indicators for the phosphorus issue area.**

| Indicator | PSR | Available Measure | Year | Source | Ref | Recommended Measure | Frequency (minimum) |
|---|---|---|---|---|---|---|---|
| Population | P | Human population by state/province | US: 1950 - 2000; Canada: 2001 | US census data at Holmes and Associates; Statistics Canada | Figure 4 | Human population by lake segment subwatershed | Update every 10 years |
| Developed land | P | Percent developed land by subwatershed | 1993 | LandSat imagery; data at VCGI | Figure 5 | Percent developed land by subwatershed | Update every 10 years |
| Agricultural land | P | Percent agricultural land by subwatershed | 1993 | LandSat imagery; data at VCGI | Figure 6 | Percent agricultural land by subwatershed | Update every 10 years |
| Animal units | P | Stocking density (animal units/ha) by subwatershed | VT:2001-2002; NY:1993-2002; QC:1998-2003 | VT AFM; NYS SWCC; QC ME | Figure 7 | Stocking density (animal units/ha) by subwatershed | Update every 2 years |
| Phosphorus load | P | Annual mean tributary P load by lake segment | 1991-2002 | LCBP long-term bio-monitoring | Table 4 | Annual mean tributary P load by lake segment | Annually |
| P in water column | S | Annual mean P concentration by lake segment | 1991-2002 | LCBP long-term bio-monitoring | Table 5 | Annual mean P concentration by lake segment | Annually |
| Chlorophyll *a* | S | Lake segment annual average chl-*a* | 1991-2002 | LCBP long-term bio-monitoring | Figure 8 | Lake segment annual average chl-*a* | Annually |
| P in sediment | S | Concentration of P in top 10 cm of lake sediment | 1994 | Hydroqual | Figure 9 | Concentration of P in top 10 cm of lake sediment | Update every 5 years |
| Aquatic plants | S | Electronic data unavailable | 1998-2003 | VT DEC | Page 24 | Biomass/$m^2$ for areas less than 10 m deep | Update every 5 years |
| Zebra mussels | S | Data on adults unavailable | | | Page 25 | Biomass/$m^2$ for areas less than 30 m deep | Update every 5 years |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| Farm BMPs | R | Percent of farms and animal units treated | VT:1996-2003; NY:2002 | VT AFM; NYS SWCC | Figure 10 | Phosphorus load reduction from implementation of agricultural BMPs by subwatershed | Update every 2 years |
| Wastewater treatment | R | Lake segment P load from WWTF | 1991, 1995-2002 | VT DEC; NY DEC | Figure 11 | Difference between lake segment P load from WWTF an the TMDL allocated load | Update every 2 years |
| Urban BMPs | R | Percent of stormwater permits that have expired | 2003 | VT DEC | Page 26 | Phosphorus load reduction from implementation of urban BMPs by subwatershed | Update every 2 years |
| Education | R | Phosphorus specific data unavailable | | LCBP | Page 27 | Dollars spent on phosphorus-oriented outreach and education | Update every 2 years |

## PRESSURE INDICATORS

There are two primary sources of phosphorus in the watershed – people and animals. Every individual person residing or visiting in the basin generates approximately 1.6 grams of phosphorus a day through normal physiology (Sawyer 1954). Census data shows that the resident population, and consequently the amount of phosphorus generated in the Champlain Basin is increasing, particularly in the Vermont portion of the basin (Figure 4). To date, population data for the Champlain Basin has been characterized by ecologic-economic zones (Holmes, et al. 1993) rather than by lake-segment subwatersheds. From an ecological perspective, population data accumulated by subwatershed would be more informative. This could help identify areas where water quality is most likely to be impacted by a growing human population.

The resident population is not the only factor to consider when assessing pressures from people. Every tourist that visits the basin also generates phosphorus. Tourism is a vital part of the Lake Champlain Basin economy, but mangers must also consider that promoting the growth of tourism could also result in increased phosphorus loading to the lake. Tourism brought an estimated $3.8 billion to the basin in 2000 (LCBP 2003); the overall contribution of these tourists to phosphorus loading to the lake has not been quantified.

Knowing how many people are in the watershed allows us to estimate how much phosphorus is *generated.* This does not necessarily translate directly into phosphorus *load*, but it does indicate the need for effective management to minimize load. For example, the amount and the pathway through which phosphorus is transported from the people to the lake depend upon whether resident's homes are sewered or have septic systems. Both sewage treatment and septic systems also vary in their effectiveness of phosphorus removal.

Additional phosphorus is also generated by a growing human population as a result of the land area conversions necessary for living. In the Lake Champlain Basin, both agricultural and natural or forested lands are being converted to developed land. In 1993, when land use in the Champlain Basin was last determined, developed land was 5.6% of the basin. Although still a relatively small percentage of the overall watershed, the amount of developed land has been increasing over the last few decades.

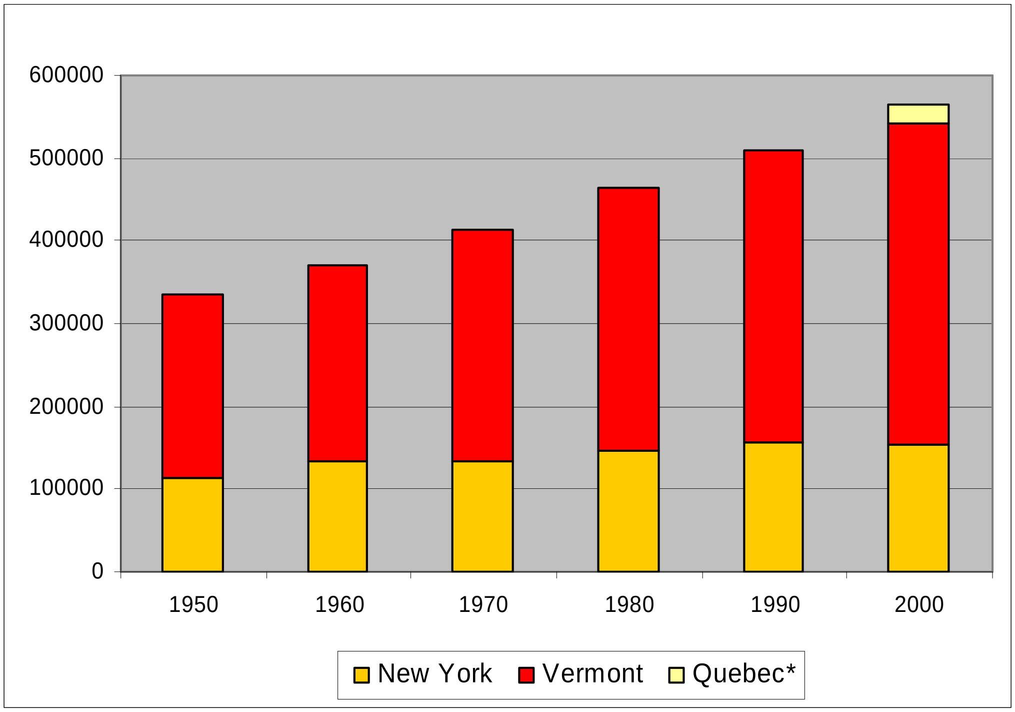



**Figure 4: Lake Champlain Basin resident population (data from 2001).**

Developed land contributes more phosphorus on a per area basis than agriculture and other land uses (Hegman et al. 1999). Vermont has a larger percentage of the resident population in the Lake Champlain Basin than New York, and contains the subwatersheds with the highest percentage of developed land. Burlington Bay, Shelburne Bay and St. Albans Bay are the lake segments that have the highest percentage of developed land (Figure 5). In just these three small lake segments, there are a total of 7,000 ha of developed land. These lake segments have five times more developed land than the Main Lake subwatershed, the largest of the thirteen lake segments.

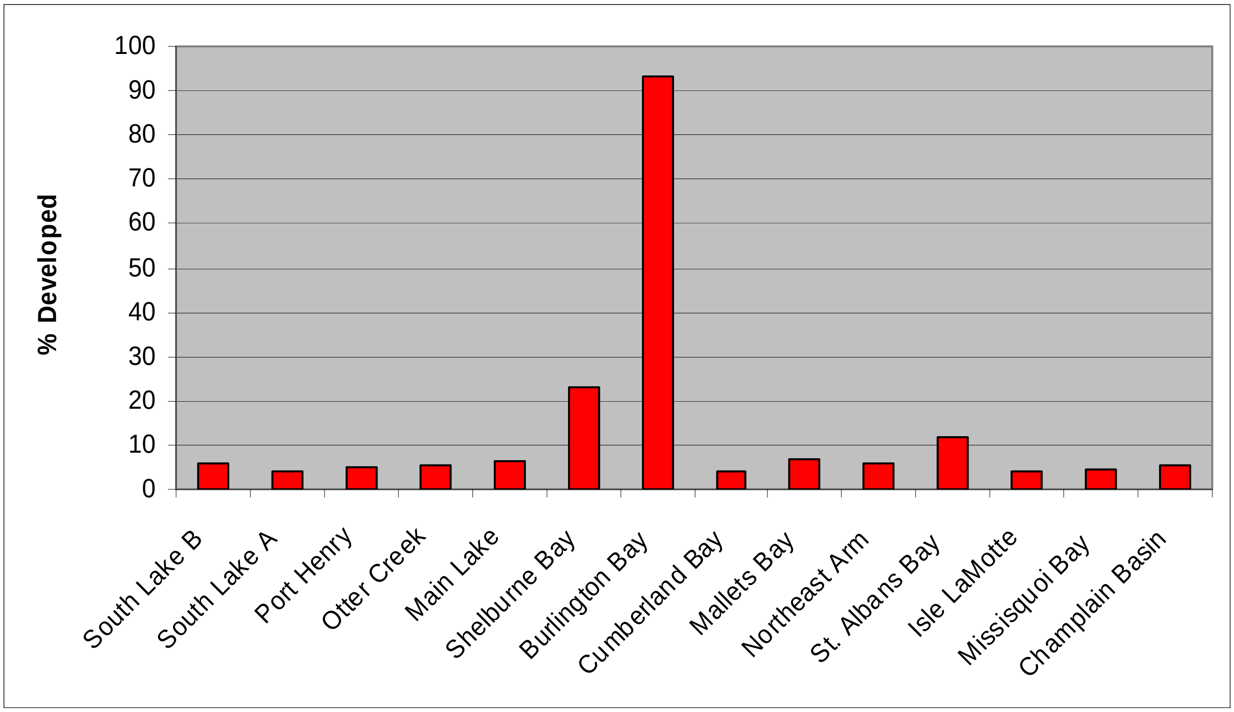



**Figure 5: Percent of lake segment subwatershed area that is developed land (based on 1993 Landsat data).**

While more phosphorus is generated on developed land on a per acre basis, more of the total nonpoint phosphorus load is derived from land used for agricultural activities (Hegman et al. 1999). A simple indicator of the pressure on the lake segments of Lake Champlain from agriculture is the percent of the subwatershed in agricultural land use (Figure 6). However, there is a broad range of specific land uses, with varying intensities and environmental impacts, included in agricultural land use as the category is used here. Therefore, additional indicators are necessary to more fully characterize the pressure on Lake Champlain resulting from agricultural activity.

The major sources of phosphorus in agricultural runoff are fertilizers and animals. The most significant of these animals are dairy cows. Pressures on the Lake Champlain ecosystem resulting from agricultural activity can also be indicated by the number of animals (measured as "animal units," each 1000 pounds) and the number of farms and in each subwatershed. Studies of the agriculture in the region suggest that while the number of dairy farms is likely to continue to decrease through the next decade, the number of animal units on each farm is likely to increase (LaDue et al. 2003).

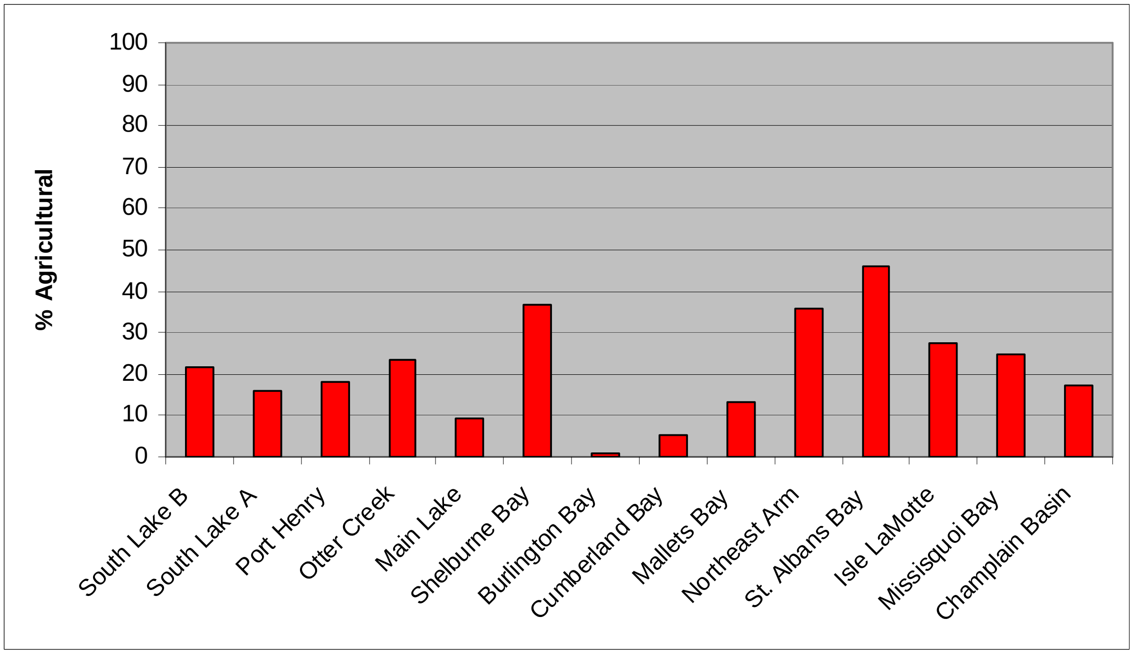



**Figure 6: Percent of lake segment subwatershed in agricultural land use (based on 1993 Landsat data).**

Increasing animal density, or an increase in agricultural intensity, increases the potential for nutrient losses from agricultural land (Watzin, et al. 2003c). Although difficult with the currently available data, examining agricultural intensity in the basin provides a more complete understanding of the pressure resulting from agricultural activity than either animal units or agricultural land use alone. Land use was last measured basin-wide in 1993. In Vermont, farm and animal unit data are collected continuously in conjunction with Best Management Practice (BMP) tracking programs. In New York, farm data were collected in 1993-1995 for the rotating river basin study program. This database has been updated since with an emphasis on larger farms that are implementing BMPs. Quebec is also developing a database on farm practices through a farm by farm inspection program that will also identify problem areas. Although the current agricultural data are better than they ever have been in the past, they still may not fully account for farms (and animal units) not participating in cost share programs or for changes in farm densities or operations since BMPs were contracted. Nevertheless, an estimate of stocking density was calculated using the available data. Although both stocking density and number of farms could both be used to indicate agricultural intensity (Figure 7), because stocking density relates more directly to the amount of phosphorus generated in the watershed and likely spread on the land (see review in Watzin et al. 2003c), it is the better specific measure.

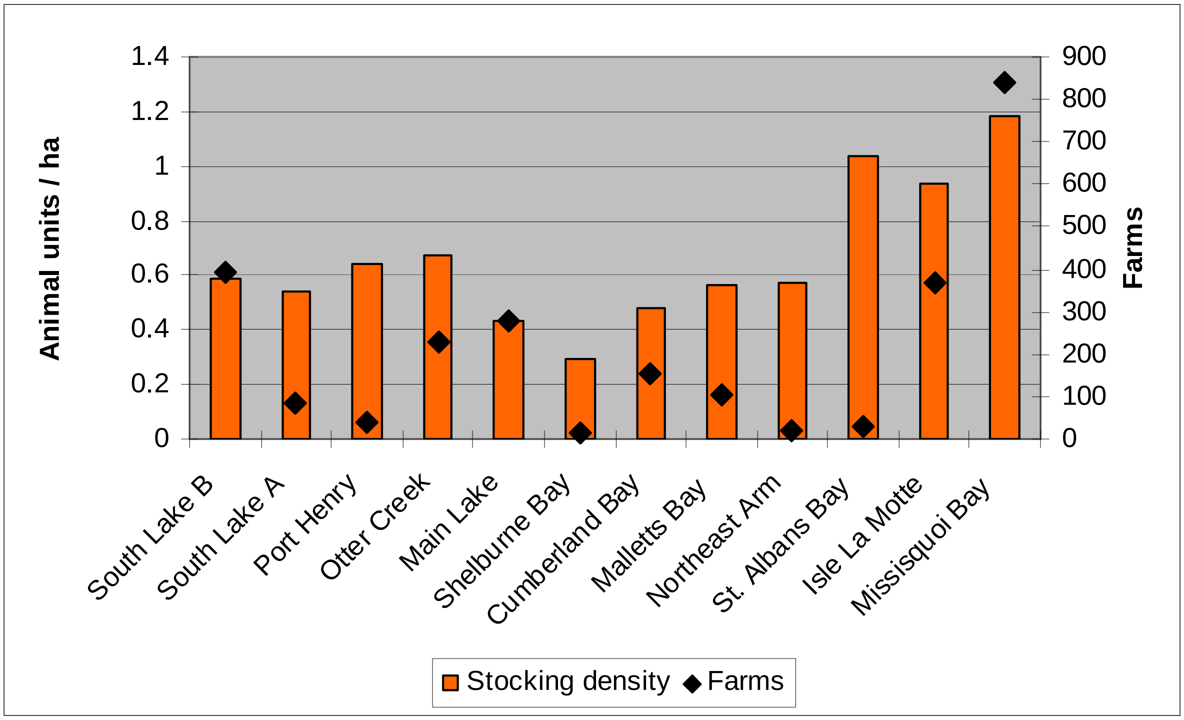



**Figure 7: Agricultural intensity by subwatershed, measured as stocking density (1000 pound animal units per hectare of agricultural land) [shown with bars] and number of farms [shown with diamonds].**

In the 1993 Water Quality Agreement between New York, Vermont and Quebec, all jurisdictions agreed to a common set of in-lake phosphorus concentration criteria that are the basis for the Lake Champlain Phosphorus TMDL and the phosphorus loadings targets established for each tributary watershed in *Opportunities for Action* (VT DEC and NYS DEC 2002). In a first look at trends in phosphorus loading to the lake from 1990 - 2000, Medalie and Smeltzer (2004) found that phosphorus loads have remained above the TMDL-derived acceptable levels in all tributaries except the LaPlatte River, which flows into Shelburne Bay (Table 4). Although the phosphorus loads from some other tributaries show a statistically significant declined over the first 10 years of monitoring, no tributary watershed has yet met the loadings target. In five tributaries, the pressure from tributary phosphorus loading is increasing.

**Table 4: Phosphorus loading from major tributaries (from Medalie and Smeltzer 2004).**

| Lake Segment | Tributary | 1999-2000 Estimated Load at Mouth (mt/yr) | Derived Target Load (mt/yr) | Trend |
|---|---|---|---|---|
| South Lake B | Poultney | 35.6 | 13.2 | None |
| | Mettawee/Barge Canal | 56.8 | 29.9 | None |
| South Lake A | Putnam | 1.9 | 1.2 | Increasing |
| Otter Creek | Otter | 110.7 | 46.9 | Decreasing |
| | Lewis | 11.2 | 3.9 | None |
| | Little Otter | 9.1 | 4.0 | None |
| Main Lake | Winooski | 154.2 | 73.4 | None |
| | Bouquet | 23.4 | 12.7 | Increasing |
| | Ausable | 45.5 | 13.7 | None |
| | Little Ausable | 7.8 | 4.2 | None |
| Shelburne Bay | LaPlatte | 5.1 | 7.1 | Decreasing |
| Cumberland Bay | Saranac | 24.8 | 13.3 | Increasing |
| Mallets Bay | Lamoille | 54.5 | 25.9 | None |
| Isle LaMotte | Great Chazy | 34.0 | 15.8 | Increasing |
| | Little Chazy | 8.4 | 3.0 | Increasing |
| Missisquoi Bay | Missisquoi | 128.0 | 49.2 | None |
| | Pike | 41.4 | 27.3 | None |

## STATE INDICATORS

The premise of phosphorus management is that anthropogenic inputs of phosphorus promote increased algal growth and premature eutrophication, which impairs human use and enjoyment of Lake Champlain (Vollenweider 1968, Rechow and Chapra 1983, NRC 1992, Smith 1998). In keeping with this, both phosphorus and chlorophyll *a* are appropriate and important state indicators. Chlorophyll *a* can be used as a general measure of the density of phytoplankton in the water.

The in-lake phosphorus criteria are annual average total phosphorus concentration goals for each lake segment. These in-lake criteria are, by regulation, the acceptable levels of phosphorus for Lake Champlain. The criteria are based upon relationships established between human use and enjoyment of the lake and average phosphorus concentrations in the water (Smeltzer and Quinn 1996).

Total phosphorus and chlorophyll *a* have been measured regularly at thirteen lake monitoring stations throughout Lake Champlain since the early 1990s. These long-term biomonitoring data are used to characterize the state of the ecosystem with regards to phosphorus and to measure progress towards the in-lake criteria. Medalie and Smeltzer (2004) examined long-term trends in phosphorus concentrations in the lake and found that concentrations were consistently above the in-lake criteria during 1990-2000 in the Main Lake, South Lake A, St. Albans Bay, the Northeast Arm, and Missisquoi Bay lake segments (Table 4). Chlorophyll *a* concentrations in the lake have also varied over time. In the Main Lake, no consistent trend in phosphorus concentration is evident through 2002 (Figure 8), but considerable variability in these measures exists. This variability will make trends detection difficult in all lake segments.

**Table 5: Annual mean lake segment TP concentrations in 1990, 2000, and 2003. Trends (from Medalie and Smeltzer 2004).**

| Lake Segment | In-lake Criteria (µg/L) | 1990 Mean TP Conc. | 2000 Mean TP Conc. | Trends (1990-2000 only) | 2003 Mean TP Conc. |
|---|---|---|---|---|---|
| South Lake B | 54 | 57 | 52 | Decreasing | 52 |
| South Lake A | 25 | 37 | 36 | None | 44 |
| Port Henry | 14 | 13 | 16 | None | 14 |
| Otter Creek | 14 | 14 | -- | | 15 |
| Main Lake | 10 | 11 | 11 | None | 9 |
| Shelburne Bay | 14 | 15 | -- | | 11 |
| Burlington Bay | 14 | 12 | 13 | None | 10 |
| Cumberland Bay | 14 | 14 | 13 | None | 12 |
| Mallets Bay | 10 | 9 | 11 | Increasing | 11 |
| Northeast Arm | 14 | 12 | 19 | Increasing | 19 |
| St. Albans Bay | 17 | 25 | 31 | Increasing | 31 |
| Isle LaMotte | 14 | 11 | 14 | None | 13 |
| Missisquoi Bay | 25 | 36 | 38 | None | 44 |

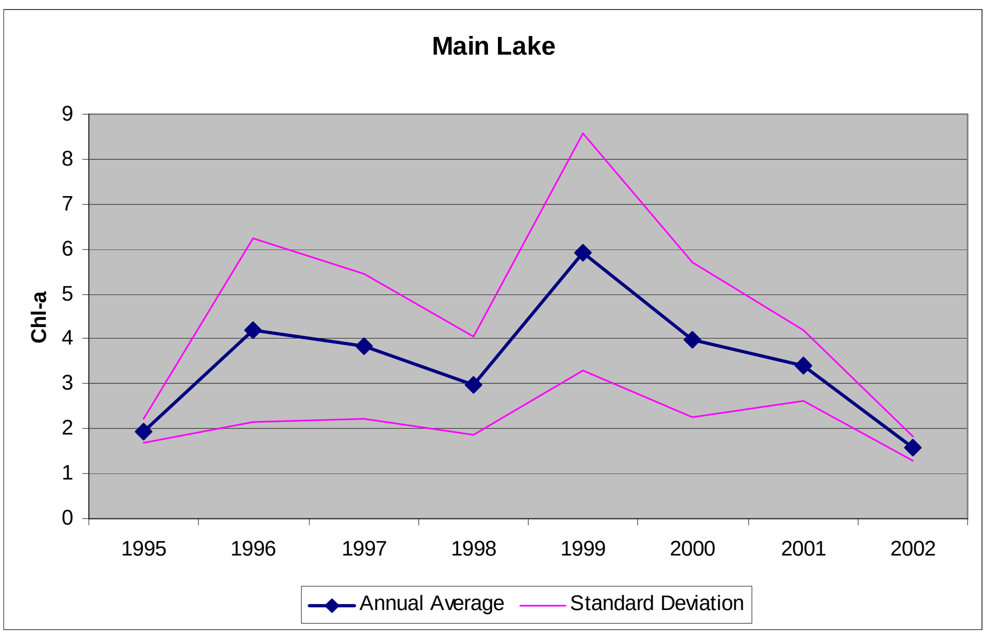



**Figure 8: Annual average chlorophyll-*a* concentration measures at Station 19 in the Main Lake, with standard deviation.**

The concentrations of phosphorus and chlorophyll *a* in the lake water are influenced by many components of the ecosystem.  Some of the phosphorus delivered to the lake is in the particulate form, settles to the bottom of the lake and is stored in the sediments.  Depending upon water depth and relative difference in the concentration of phosphorus in the sediment and in the overlying water, and the oxygen concentration and geochemistry of the sediment, phosphorus delivered to the sediment can be resuspended and recycled back into the water column where it may become available to phytoplankton.

The concentration of phosphorus in the sediment is not monitored regularly in Lake Champlain nor is its role in lake phosphorus dynamics well characterized.  One study of benthic phosphorus cycling was conducted in the mid-1990s (HydroQual, Inc. 1999).  As part of this study, TP was measured from sediment cores collected at various lake monitoring stations (Figure 9). HydroQual, Inc. (1999) found that phosphorus resuspension was greatest in eutrophic lake segments.  For some lake segments, internal phosphorus loading from the sediments may be a key process in maintaining high phosphorus concentrations in the water column.  In the following section we describe our modeling efforts, which clearly shows that internal phosphorus loading is an important process in at least Missisquoi Bay, and probably some other lake segments as well (for example, the South Lake and St. Albans Bay).  However, further studies are necessary to characterize this aspect of the lake ecosystem.

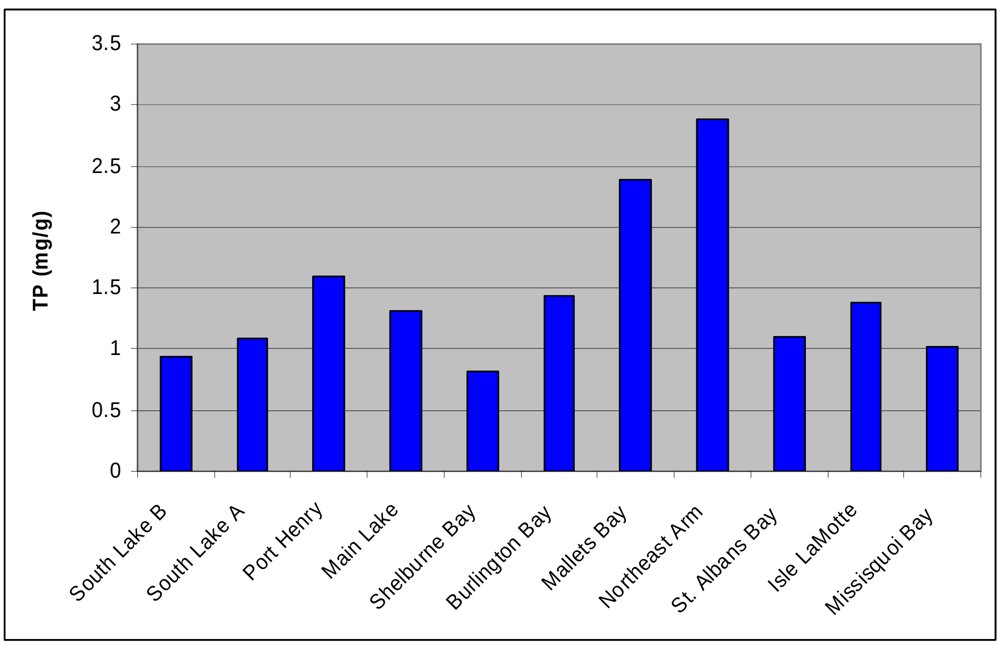



**Figure 9: Average total phosphorus in top 10 cm of lake sediments from stations at or near the long-term biomonitoring sites (calculated from data presented in HydroQual, Inc. 1999).  Although in some cases multiple sites were sampled in the HydroQual study, only the site nearest the long-term monitoring site is included in this graph.**

The presence of aquatic plants, or macrophytes, can also influence the utilization of phosphorus by phytoplankton. These rooted plants remove phosphorus from the sediment and also reduce light penetration in the overlying water column. In Vermont, VT DEC has been conducting regular plant surveys to track the spread of water chestnut in the South Lake since 1998 and scattered earlier data exist. These surveys include qualitative assessments of native and nonnative plant densities but the data have not yet been compiled into an accessible format. No data were found in New York. Because the growth of aquatic plants can be a key source of phosphorus storage and regeneration, we recommend that information on the distribution and abundance of aquatic plants, particularly in shallow lake segments, is necessary for a more complete understanding of phosphorus dynamics.

Finally, zebra mussels may also be influencing water clarity, at least in the segments of Lake Champlain where zebra mussel density is high. The long-term monitoring program only tracks veliger and new recruit densities, not adult densities, however, there is an increase in water clarity in the South Lake, where zebra mussels invaded first and presumably have reached their highest adult densities (VT DEC 2004). Zebra mussels are filter feeders; they feed on plankton that they remove from the water column. Large reductions in phytoplankton biomass have been observed in the Hudson River (Pace et al. 1998; Baker et al. 1998), Lake Oneida (Idrisi et al. 2001), and Saginaw Bay, Lake Huron (Fahnenstiel et al. 1995) following zebra mussel proliferation in those systems. In Lake Champlain, zebra mussels are probably also reducing phytoplankton biomass, which might appear not only as increased water clarity, but also decreases in chlorophyll *a*.

Zebra mussels can also change both the concentration and the form of phosphorus in the water column. Increases in soluble reactive phosphorus and total soluble phosphorus have been observed in several instances of zebra mussel introduction (Heath et al. 1995; Holland et al. 1995; Arnott and Vanni 1996; James et al. 2001). In laboratory experiments using a natural plankton community from Lake Champlain, Brines (2004) demonstrated an increase in soluble reactive phosphorus and total nitrogen, and a decrease in TN:TP ratios, chlorophyll *a* concentration, and total phytoplankton biovolume in 200-liter aquaria with zebra mussels compared to those without. However, because there currently is no lakewide data on adult zebra mussel densities, the implications of these changes for phosphorus management are unclear.

## RESPONSE INDICATORS

Because the in-lake phosphorus concentration criteria are exceeded in many lake segments, the Lake Champlain Basin Program has focused on management activities to reduce the pressures that result in phosphorus loading to the lake. Pressures resulting from agriculture are reduced through the implementation of BMPs. These BMPs have been applied throughout the basin and can be tracked by farm or by animal unit (Figure 10). Missisquoi Bay, Otter Creek, South Lake B, Main Lake and Isle La Motte are the lake segments with the greatest pressure from agriculture.

Ideally, BMPs will be implemented on all farms in the Lake Champlain Basin needing them. In 2000, an analysis looking at the load reductions that might be achieved through implementation of BMPs on all these farms suggested that the current credits for practices associated with manure management may overestimate the real reductions in phosphorus load; for some watersheds, a negative load was generated when this approach was taken (Donlon and Watzin 2000). In the future, the phosphorus load reductions used as measures for the agricultural BMP indicator should be adjusted to correct this problem.

Because the phosphorus problems are most extreme in the South Lake, St. Albans Bay, and Missisquoi Bay, the greatest improvements in water quality in the lake might come from targeting farms for management in these areas. In 2004, priority for cost-share dollars has been given to the Missisquoi and St. Albans watersheds in Vermont. The Province of Quebec has also made Missisquoi Bay a target area and has invested in both technical assistance through agroenvironmental advisory clubs, and a variety of inspection and regulatory control programs. Ideally, response indicators should track the P reductions associated with all these efforts.

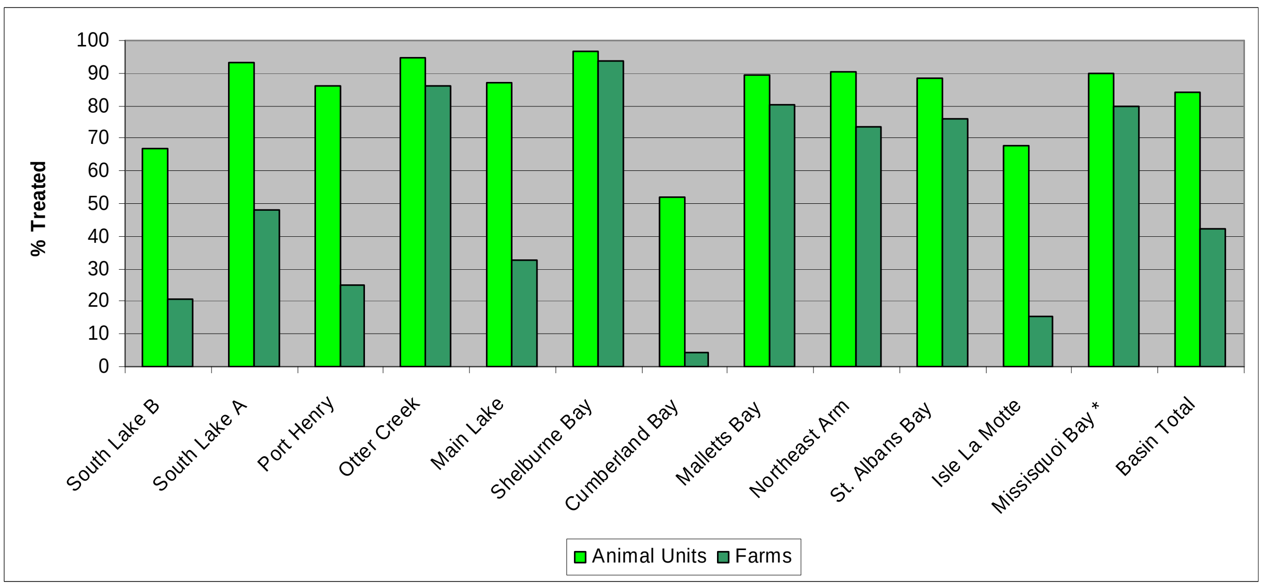



**Figure 10: Percent of farms and animal units where BMPs have been implemented. * Missisquoi Bay shows only the Vermont portion of the basin because BMP data for Quebec were not available.**

The human population in the basin generates both point and nonpoint source phosphorus; point source phosphorus through household sewerage and wastewater, and nonpoint source phosphorus through runoff from the developed land that supports the human population. Wastewater treatment facility upgrades have been the primary management action to reduce point source phosphorus inputs into Lake Champlain. As with tributary nonpoint phosphorus loads, acceptable wastewater treatment facility loads for each lake segment have been determined as part of the Lake Champlain phosphorus. Total Maximum Daily Load (TMDL) allocation (VT DEC and NYS DEC 2002). The phosphorus load from wastewater treatment facilities in Vermont has declined substantially since 1991 (see Figure 11). As of 2002, Mallets

Bay was the only lake segment in Vermont exceeding its TMDL wastewater treatment facility phosphorus load allocation.

Although all wastewater treatment plants have an allocated load in the TMDL, many treatment plants acre currently discharging significantly below this load. This was a way for the states to allow development to continue to occur in growth areas, however, it means that actual phosphorus load to Lake Champlain from point sources could increase in the future. This increase will have to be offset by reductions from the nonpoint source load if the in-lake concentrations in Lake Champlain are ultimately to be achieved.

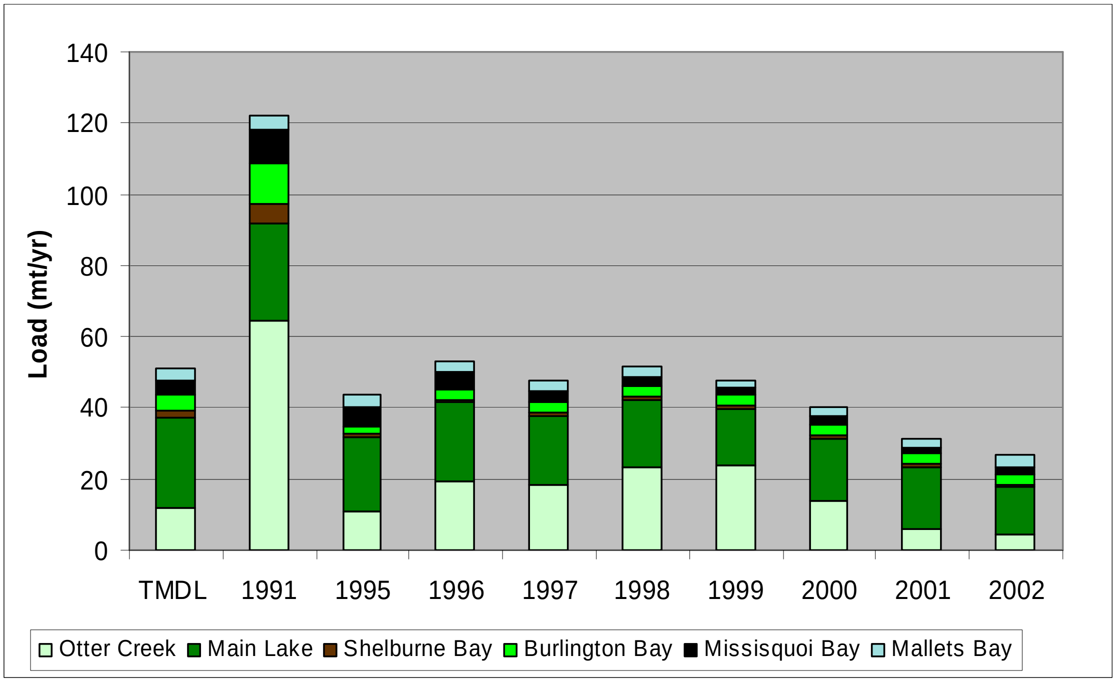



**Figure 11: Wastewater treatment facility phosphorus loads in Vermont for selected lake segments from 1991 and 1995-2002 compared to load allocation in phosphorus TMDL.**

Developed land is a major source of nonpoint source phosphorus associated with the human population. At the national level, the amount of developed land almost quadrupled between 1954 and 1997, from 18.6 million acres to about 74 million acres in the contiguous 48 states. Between 1982 and 1992, the amount of developed land in the state of Vermont grew by more than 25%, about twice the rate of population growth in the state (Vermont Forum on Sprawl 1999). Chittenden County in Vermont had highest rate of growth of developed land, at 40%, and the second highest rate of population growth, at about 14%.

Because the impervious surfaces that go along with this development prevent rainwater infiltration and adsorption of nutrients in the soil, developed land generates a high phosphorus load. Stormwater management is one approach to reduce the phosphorus and other pollutants in runoff. Although there are very few data available on stormwater management in the basin, many developments are required to have a permit for stormwater discharges into surface waters.

The purpose of the permits is to ensure that stormwater is treated in order to reduce the impact to freshwater ecosystems. In 2003, eighty-eight percent of the 1872 stormwater permits issued in the state of Vermont had expired. Although this does not necessarily mean that all these stormwater treatment systems have failed, it does suggest that some might not be operating efficiently or effectively. In Vermont, the new stormwater law passed by the legislature in March 2004 will require that all expired permits be updated. No data on the status of stormwater permits in New York was available.

Another potential source of nonpoint source phosphorus from developed land is failed septic systems. Although on a watershed-wide basis, Budd and Meals (1994) estimated that the inputs of phosphorus from failed septic systems is likely to be minor, in some specific areas, the local impacts could be significant. The shoreline around Inner Mallets Bay, for example, has a high density of development on poor soils for septic systems. Many systems are old and this situation may be contributing to localized problems in this area (Budd and Meals 1994). For the Mallets Bay lake segment, an indicator related to the number of failed septic systems might be considered, but such as indicator is probably not needed lakewide.

A variety of education efforts have been undertaken by the LCBP to address nonpoint source pollution, including most recently, a lawn care brochure and a series of posters about residential sources of nonpoint source phosphorus. Although it is extremely difficult to calculate the loadings reductions that come from such activities, awareness is the key to changes in personal behavior, and reducing phosphorus in the human point and nonpoint source waste stream.

## SEMI-QUANTITATIVE PHOSPHORUS MODEL FOR LAKE CHAMPLAIN

Although our Pressure – State – Response framework for phosphorus (Figure 3) includes the major biotic, abiotic and managerial components necessary for a basic understanding phosphorus dynamics in Lake Champlain, it does not provide much insight into the relative importance of the numerous ecosystem components and pathways of phosphorus transport and transformation. Computer models can provide a quantitative infrastructure for analyzing such complex ecosystems and help guide the process of thinking about natural dynamics, indicator selection and management options. We developed the Phosphorus Ecological Indicator (PEI) model for just these purposes.

The PEI is a simple dynamic mass balance model that was created using STELLA® simulation software (ISEE Systems). The model is a dynamic mass balance accounting procedure that follows, over time, all important storage and major inputs and outputs of phosphorus for a given lake segment. The PEI model is deterministic because no variability is associated with the model inputs or with any of the internal computations. Model inputs that define the characteristics of both the watershed and the lake segment are annual averages or totals from which monthly average values of phosphorus storage, fluxes and phosphorus and chlorophyll *a* concentrations are calculated. The model tracks, over time, change in the magnitude of phosphorus storage, fluxes and phosphorus and chlorophyll *a* concentrations in a lake segment. All simulation runs were for a period of 20 years. The model is initially calibrated so that calculated water column phosphorus levels correspond to average phosphorus concentrations from the long-term monitoring data (VTDEC 2004).

Even though the PEI model is extremely simple, as environmental models go, it is very flexible. All model inputs can be changed at any time in the simulation run to emulate changing conditions or to simulate implementation of some new management condition in the watershed. However, the model does not simulate seasonal cycles or individual events. The model also does not consider the effects of natural factors that can stress lake ecosystems with regard to the impacts of phosphorus pollution.

All computer models are merely a set of mathematical algorithms that are organized to describe a prescribed problem. The models are driven by data that define modeling conditions and by calculations made by the computer based on the model algorithms. The best available state-of-science algorithms for the Lake Champlain Basin were used in the PEI model. The data entered into the PEI model are derived, in as much as possible, from local monitoring programs and research studies and, when necessary, from the literature. These data are estimators of various aspects of the biotic and abiotic character of the lake segment, pressure indicators and response indicators. In some cases local data, estimators of various aspects of the biotic and abiotic character of the lake segment, were minimal and sometimes highly variable. A discussion of many data required by the PEI models is given in Appendix B.

There are many factors that tend to stress a lake ecosystem with respect to phosphorus; some are natural with much inherent variation and cannot be controlled by management efforts while others result from anthropogenic activity and can be manipulated by management. The natural factors include sunlight, air and water temperatures, wind and precipitation (Table 6). All such factors can fluctuate considerably over time and ARE NOT incorporated into the deterministic PEI models. The human factors, or pressure indicators, and internal loading factors ARE incorporated into the PEI models. The major pressure indicators in the PEI model are listed in Table 7. Appendix A provides more detailed information for many parameters input into the PEI models.

**Table 6: Natural stressors that can influence lake ecosystems and the outcomes of phosphorus pollution. These stressors are not incorporated into the PEI model.**

| **Natural Factors** | **Remarks** |
|---|---|
| 1. Sunlight Intensity and Amount | Influences plant/algal growth in presence of needed nutrients. Not incorporated into the PEI model. |
| 2. Air and water temperatures | Influences plant/algal growth in presence of needed nutrients and water exchange patterns. Not incorporated into the PEI model. |
| 3. Wind direction and intensity | Influences water exchange patterns. Not incorporated into the PEI model. |
| 4. Precipitation amounts & patterns | Influences water exchange patterns and timing of inflows to lake segment. Not incorporated into the PEI model. |

**Table 7: Factors that can stress lake ecosystems and influence the outcomes of phosphorus pollution and which are incorporated into the PEI models. Human factors are pressure indicators while internal loading factors are those associated with the cycling of phosphorus between the water column and the bottom sediments.**

| **Human Factors (Pressure Indicators)** | **Remarks** |
|---|---|
| 1. Human Population<br>a. No. of equivalent full time residents<br>b. Rate of population change | Influences P discharges in liquid & solid wastes and the amount of land in basin converted from forestry and agriculture to urban land. Influences amount of commercial & industrial development and urban non-point source runoff. Typically causes a slow but relentless increase in urban runoff and point source discharge since population growth is typically not controlled. Total population and rate of population growth are indicators incorporated into the PEI model. |
| 2. Land Resources<br>a. Land use type (Ag, Forest, Urban)<br>b. Area in each land use<br>c. P export coefficients | Influences the amount of P exported from land areas in different use. P export coefficients specific to the Lake Champlain Basin are incorporated into the PEI model, however, the model does not account for change in soil P content due to BMP implementation on agricultural lands. The PEI model simulates different patterns of urban growth so that change in amount of land area in each use category is tracked over time as a function of population growth. PEI model does not explicitly track change in the number of domestic animals on farms over time. |
| **Internal Loading Factors** | |
| 1. Sediment TP Assay<br>2. Sediment Bulk density<br>3. Sediment depth<br>4. TP settling rate<br>5. TP diffusion/resuspension rate constant | Determine the mass of TP stored in the bottom sediments and hence the rates at which phosphorus moves from the sediment to the water column. Determines the proportion of the TP inputs that settles onto the lake segment bottom. |

The PEI model tracks two state indicators with respect to phosphorus; the monthly average total phosphorus (TP) concentration in the water column and the monthly average concentration of chlorophyll a. Whenever the value of a pressure indicator that drives the PEI model is changed, the concentrations of TP and chlorophyll *a* in the lake segment may also change. Specifically, all or one of the following indicators of pressure can be changed: rate of population growth, sewered vs. unsewered populations, wastewater treatment plant efficiencies, urban growth patterns (sprawl vs. high-density), agricultural erosion and nutrient management and urban erosion and storm water management. In reality all these pressures change over time usually more or less simultaneously so that existing monitoring programs have difficulty separating the impacts of any change due to an individual response. The PEI model can emulate these real world conditions but, unlike most on-ground monitoring programs, can also evaluate change in the state of a lake segment due to an individual management response that might be taken.

## Phosphorus Mass Balance for Lake Segments

Figure 12 is a schematic compartment-flux diagram that illustrates for a lake segment the concept of mass balance upon which the PEI model is based. Phosphorus enters the lake segment water column through direct surface runoff from urban, forested, and agricultural areas that drains into the lake segment (red arrows on Figure 12) and therefore can be controlled through implementing land-based management programs. Some of the inputs to the water column come directly from the watershed (non point source and point source loads). Those inputs from adjacent lake segments (black arrows on Figure 12) in advective and exchange flows are less amenable to control through management. Phosphorus also enters the water column from lake bottom sediments. Phosphorus is output from the lake segment water column in advective and exchange flows with adjoining segments and by the settling of phosphorus-laden particles onto the lake floor. Phosphorus can be removed from the bottom sediments by harvesting macrophytes, dredging or containing the phosphorus so that it cannot move into the water column. Management of the internal cycling of phosphorus, although technically possible, may not be environmentally acceptable.

Mass balance accounting of phosphorus by the PEI mode for a lake segment involves, for both the water column and bottom sediments, simultaneously summing up all phosphorus inputs, all phosphorus outputs and then determining, on a monthly basis, whether the summed inputs are greater or less than the summed outputs: just like the end-of-the-month checkbook balancing routine we all go through. If, for example, inputs exceed outputs for both the water column and bottom sediments then the amount of stored phosphorus increases for both. For the water column, this translates into increased phosphorus concentration while increased amounts of phosphorus in the sediment are likely to increase internal loadings of phosphorus. The concept of mass balance is explained in some detail in Appendix B.

Lake Champlain is a collection of 13 interconnected physically unique smaller segments. Some are highly interconnected with adjacent segments so that large amounts of water are exchanged while others exchange little water with adjacent segments. Some have relatively long retention periods while others have short retention periods. Thus, even though each lake segment processes phosphorus generally according to the schema outlined in Figure 12, the magnitude of phosphorus storage and movement into, within and out of the lake segment varies according to the unique physical character of the segment and its contributing watershed. Consequently, each segment responds differently to the pressures of phosphorus pollution.

Some of the consequences of these differences can be examined with the PEI model. However, there were insufficient resources available through this project to model each of the 13 Lake Champlain segments. Therefore, consistent with available resources, we chose to develop a PEI model for three lake segments. We chose the Missisquoi Bay, Shelburne Bay, and Main Lake segments to serve as case studies because they are very different in their physical characteristics and watershed development.

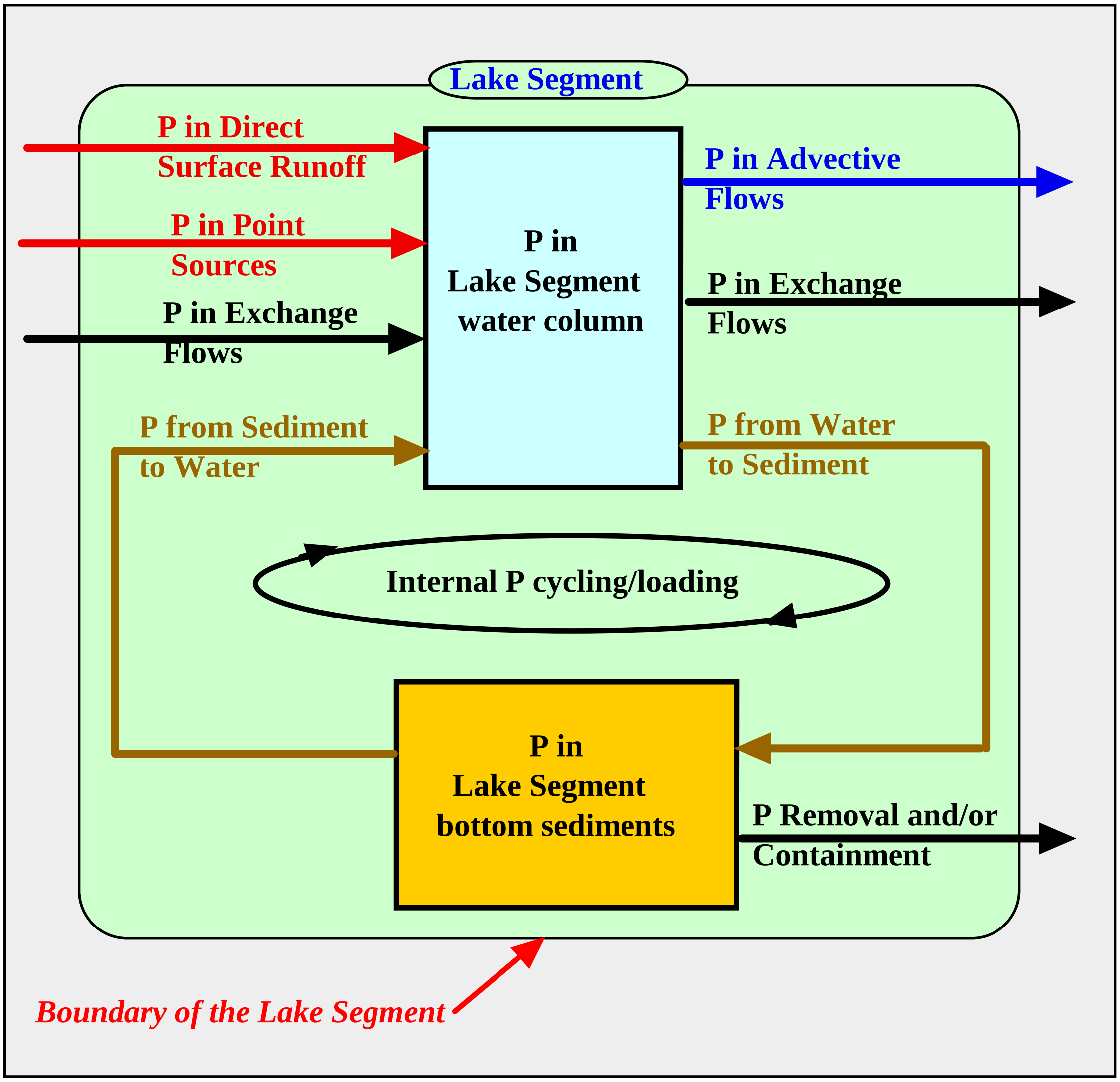



**Figure 12: Schematic compartment-flux (CF) diagram showing the major pathways for the input and output of phosphorus for a lake segment. The boxes represent phosphorus storage in the water column and bottom sediments of the lake segment. Arrows crossing the segment boundary and pointing to a box are input fluxes (loads) of phosphorus while arrows crossing the segment boundary and pointing from a box are output loadings of phosphorus. Phosphorus input due to atmospheric deposition is not considered in the PEI model. All fluxes (arrows) have units of mass /time; phosphorus storage (boxes) has units of mass or mass/volume (concentration).**

Missisquoi Bay is very shallow, drains an agricultural watershed, and is essentially disconnected from the rest of Lake Champlain. Shelburne Bay is deep, drains a watershed that is increasingly urban and suburban and is closely connected with the Main Lake and Burlington Bay lake segments. The Main Lake segment is very deep, directly drains only a relatively small watershed area and is very closely connected to numerous other lake segments. The physical characteristics of these segments are summarized in Table 8.

**Table 8: Summary of important physical characteristics of the Missisquoi Bay, Shelburne Bay, and Main Lake segments in that drive dynamic phosphorus mass balance computations in the PEI models. (*computed in PEI model based on lake segment surface area and volume from Smeltzer 1999)(**from Smeltzer 1999)**

| Parameter | (units) | Missisquoi Bay | Shelburne Bay | Main Lake |
|---|---|---|---|---|
| Average Depth* | (meters) | 2.8 | 14.6 | 40.5 |
| Advective Outflow** | (hm3/year) | 2039 | 79 | 9402 |
| Exchange Flow** | (hm3/year) | 297 | 4816 | 75034 |
| ExchQ/Adv Q Ratio* | (dimensionless) | 0.15 | 62.5 | 8.0 |
| Retention Period based on advective outflow* | (years) | 0.100 | 1.77 | 1.79 |
| Retention Period based on total outflow* | (years) | 0.088 | 0.028 | 0.224 |

The deterministic PEI models are not quantitative forecasting models nor do they produce output that is amenable to statistical evaluation and thus PEI model outputs ought not be viewed in that manner. The models are mass balance accounting routines that generate trends, over a 20 year period, for parameter outputs (i.e., water column phosphorus and chlorophyll *a* concentrations). Values of these outputs for different management alternatives can be compared at, say the end of a 20-year simulation run, by some percentage difference but statistical comparisons are not appropriate. Similarly, the trend lines for these outputs can be compared, for example, are they increasing or decreasing, or by average trend line slopes, but again, statistical comparisons are not appropriate.

The strength of these deterministic models is their ability to ask questions about different scenarios and to evaluate the relative contribution of different factors to future conditions. The PEI models can be used to examine questions like “how might we expect the TP concentration in the water column of a given lake segment change over the next 20 years if a particular management alternative is implemented?” Do we expect TP concentration to increase or decrease, a lot or a little? Should other management alternatives be considered? Is a particular management alternative worthy of receiving additional study? Are there additional options to move towards some management goal?

## The Main Lake Segment

The Main Lake segment is, by far, the largest segment of Lake Champlain, yet only a relatively small land area (about 504,250 ha) drains directly into it. About 205,000 people live in the watersheds that drain directly into the Main Lake segment. It has an average depth of about 40 meters. The Main Lake segment is directly connected to six other lake segments, namely the South Lake, Shelburne Bay, Burlington Bay, Malletts Bay, Cumberland Bay and Isle LaMotte segments. Each of these connecting segments have advective and exchange flows with the Main Lake, therefore, there is considerable mixing in this lake segment. Based on the advective outflow, the Main Lake segment has a retention period of about 1.8 years. However, the ratio of the total exchange flow (sum of the advective and exchange flows) to the advective outflow to the Isle LaMotte segment is about 8:1, thus, the actual retention period in the Main Lake is only about 0.25 years. The flow from the South Lake segment is the largest component of exchange flow entering the Main Lake. Consequently, the water quality of the Main Lake segment, a “collective average” of the water qualities of the various advective inflows and exchange flows with the connecting lake segments, should rather quickly reflect major changes in the water qualities of these adjoining segments. The internal cycling of phosphorus from the sediments to the water column is expected to be about 2.2 %/year of TP stored in top 150 cm of bottom sediment.

Values of some input parameters for the PEI model of the Main Lake segment are summarized in Table 9. These parameter values reflect the current conditions for the watershed area that drains directly into the Main Lake segment. At a sustained population growth rate of 1.2%/year, the Main Lake PEI model calculates an increased population of about 56,000 people by the year 2020 (from 204,000 to 260,000) for the Main Lake watershed area. This 27% population growth increases the amount of urban land area by about 25% while the areas devoted to forests and agriculture decrease.

PEI model Simulation Run ML1 suggests that, if current conditions persist for the next 20 years, phosphorus loadings from urban runoff and point sources increase substantially (about 27-28% each). These increases in phosphorus loadings are the direct consequence of a 25% population growth assuming that society does nothing more than it now does to treat its wastewater and manage its land resources.

The input of phosphorus to the Main Lake segment in the exchange flows from the six adjacent lake segments is estimated by the PEI model to total about 1138 mt TP/year, a little more than 10 times the amount of phosphorus that enters from the watershed itself. Consequently, the water quality in the Main Lake segment is predominately controlled by the collective water qualities in the exchange flows from the adjoining lake segments.

The Main Lake PEI model assumes that the TP concentrations in the advective and exchange flows from adjoining segments are the average values based on monitoring data and remain constant over the 20 years of each simulation run. In reality, monitoring data appear to indicate that water column TP concentrations are decreasing in some segments while increasing in others (Medalie and Smeltzer 2004).

**Table 9: Some input parameter values and phosphorus loadings for the Main Lake segment. The items marked with an asterisk are inputs to the PEI model that define current conditions (ca. 2000). All other values were computed with the Main Lake PEI model assuming all conditions in the watershed that drains directly into the Main Lake remain unchanged over 20 years except that population grew 1.2%/year. This is Simulation Run ML1.**

| Parameter | Units | Current | 20 years in future |
|---|---|---|---|
| Population | No. people | 204,860* | 260,400 |
| Rate of Growth | %/year | 1.2* | 1.2 |
| Urban Land Area | ha | 35,390* | 44,180 |
| P Export Coeff | kg TP/ha/yr | 1.50* | 1.50 |
| Agr Land Area | ha | 49,180* | 45,390 |
| P Export Coeff | kg TP/ha/yr | 0.42* | 0.42 |
| Forest Land Area | ha | 419,680* | 414,680 |
| P Export Coeff | kg TP/ha/yr | 0.04* | 0.04 |
| | | | |
| **Nonpoint Source TP Loads as calculated by the PEI model** | | | |
| Agricultural | mt/year | 20.65 | 18.56 |
| Forestry | mt/year | 16.79 | 16.59 |
| Urban | mt/year | 53.09 | 68.08 |
| Point Source TP loads | mt/year | 11.68 | 14.85 |
| Total TP load to Main Lake directly from watershed | mt/year | 102.21 | 118.08 |
| Total TP load to Main Lake from advective and exchange flows | mt/year | 1,138.40 | 1,138.40 |

Figure 13 shows the sensitivity of Main Lake TP concentration over 20 years to the implementation of phosphorus management programs that reduce phosphorus input to the Main Lake segment. If current conditions persist, accounting for population growth, Main Lake TP concentration increases from 11.79 μg/L currently to 13.04 μg/L in 20 years (Simulation Run ML1). If phosphorus management programs are implemented in year 5 to reduce by 70% the current agricultural and urban phosphorus loadings for the watershed that drains directly into the Main Lake, a very aggressive management scenario, the Main Lake TP concentration shows a rather quick 0.5 μg/L decrease then continues to increase to 12.33 μg/L by year 20 (Simulation Run ML2). Thus, it appears that management programs implemented only in the watershed that drains directly to the Main Lake may make little long-term difference in the TP concentration in the waters of the Main Lake.

If management programs reduce agricultural and urban TP loads by 70% in the-watershed that drains directly into the Main Lake AND, as well, reduce the TP content of South Lake waters by 0.5 μg/L /year between years 5 and 10 (Simulation Run ML3), TP concentrations in Main Lake waters decrease from 12.19 μg/L in year 5, to 10.27 μg/L in year 20. However, between years 10 and 20, the TP concentration increases because of continuing population growth. Based on a comparison of these scenarios, it seems that the water quality in the Main Lake is far more sensitive to change in South Lake water quality than to watershed management responses designed to limit TP discharge from just the Main Lake watershed itself.

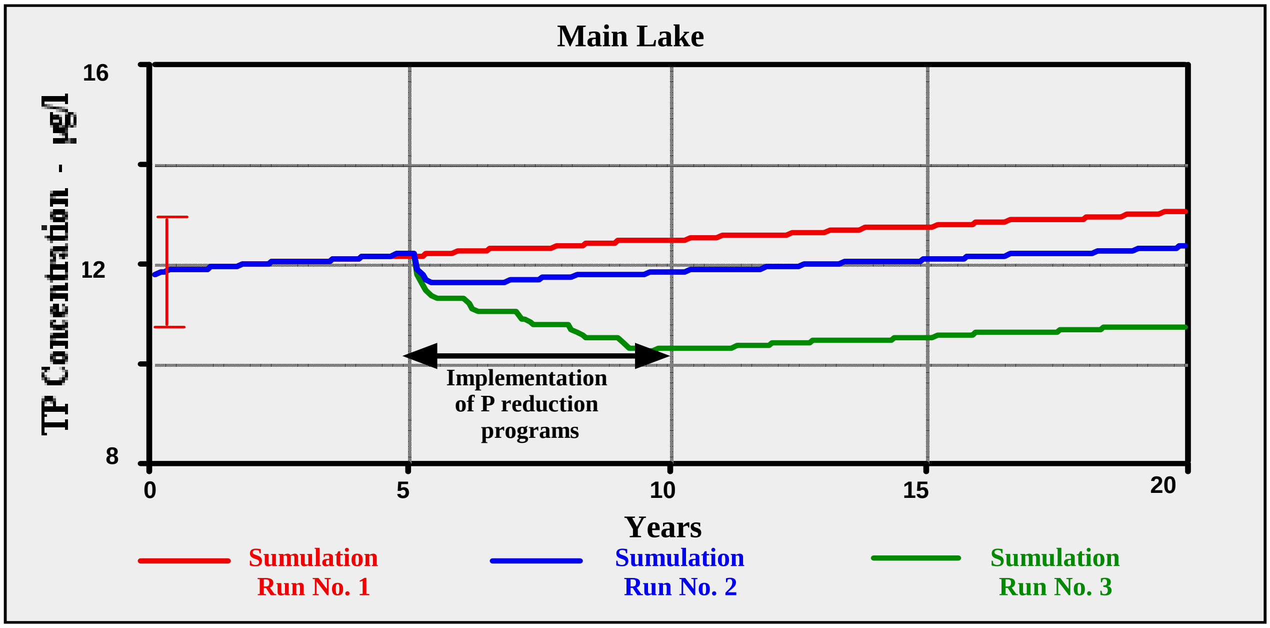



**Figure 13: Sensitivity of average monthly TP concentration in the water column of Main Lake segment to alternative phosphorus management scenarios as computed with the PEI model. Simulation Run ML1 assumes a 1.2%/year population growth rate, that urban development patterns continue as at present, and land and wastewater management remain unchanged over the 20 years. Simulation Run ML2 is the same as ML1, except that the TP in agricultural and urban runoff is reduced by 70% in year 5. Simulation Run ML3 is the same as ML2 except that the TP concentration in the South Lake segment is also reduced 0.5 µg/L each year between years 5 and 10. The PEI model computes nonpoint source TP loads based on current agricultural, urban and forestry TP export coefficients of 0.42, 1.50, and 0.04 kg TP/ha/year, respectively. The error bar is the average standard error of monthly mean TP concentrations determined from monitoring data.**

Monitoring data collected on the Main Lake between 1992 and 2000 show a distinct seasonal pattern in monthly mean TP concentrations and considerable variation among average annual and average monthly mean values. The standard error of the monthly mean TP concentrations varied over the seasonal cycle, averaging about 1.2 µg/L. When this variation is viewed as an error bar within the context of Figure 13 it appears that it may be difficult to distinguish differences in the average monthly TP concentrations between simulation runs ML1 and ML2. Thus, these simulation runs suggest that the use of TP concentration as the primary indicator for the measuring change in the state of lake pollution may not provide sufficient sensitivity to detect subtle concentration trends or differences in the Main Lake resulting from the implementation of management responses. In all three simulations, the long-term trend in TP concentrations is upwards. Additional study is needed to assure that the Main Lake monitoring data are collected and analyzed in ways most likely to detect small changes in TP concentrations over time.

### The Shelburne Bay Lake Segment

Shelburne Bay, with an average depth of 14.6 meters (Table 8), is not as deep as the Main Lake segment but deeper than the Missisquoi Bay segment. Water column TP concentrations average about 15 µg/L, somewhat higher than the Main Lake segment. The internal loading of phosphorus in Shelburne Bay is probably similar to the Main Lake. The waters of Shelburne Bay are believed to be substantially mixed by exchange flows with both the Main Lake and the Burlington Bay segments (Manley, personal communication). These exchange flows are some 60X greater than the advective flows leaving Shelburne Bay (Table 6). The average retention period for Shelburne Bay, based only on the advective outflow, is around 1.75 years. A more realistic retention time based on the total outflow (exchange + advective flows) is about 0.028 years (Table 6).

The Shelburne Bay PEI model is functionally identical to the Main Lake PEI model but is run under conditions that define the unique physical character of Shelburne Bay and current levels of human activity in the Shelburne Bay watershed. The current values of these parameters are listed in Table 10 along with levels estimated by the Shelburne Bay PEI model for 20 years in the future assuming a 2.4%/year population growth in the watershed.

**Table 10: Some input parameter values and phosphorus loadings for the Shelburne Bay segment. The items marked with an asterisk are inputs to the PEI model that define current conditions (ca. 2000). All other values were computed with the Shelburne Bay PEI model assuming all conditions in the watershed that drains directly into Shelburne Bay remain unchanged over 20 years except that population grows 2.4%/year. This is Simulation Run SB1.**

| Parameter | Units | Current | 20 years in future |
|---|---|---|---|
| Population | No. people | 20,000* | 32,320 |
| Rate of Growth | %/year | 2.4* | 2.4 |
| Urban Land Area | ha | 4,130* | 6,340 |
| P Export Coeff | kg TP/ha/yr | 1.5* | 1.5 |
| Agr Land Area | ha | 6,600* | 5,490 |
| P Export Coeff | kg TP/ha/yr | 0.42* | 0.42 |
| Forest Land Area | ha | 5,580* | 4,470 |
| P Export Coeff | kg TP/ha/yr | 0.04* | 0.04 |
| **Nonpoint Source TP Loads as computed by the Shelburne Bay PEI model** | | | |
| Agricultural | mt/year | 2.8 | 2.3 |
| Forestry | mt/year | 0.2 | 0.2 |
| Urban | mt/year | 6.2 | 9.1 |
| Point Source TP loads | mt/year | 1.1 | 1.8 |
| Total TP load to Shelburne Bay | mt/year | 10.3 | 13.4 |

The Shelburne Bay watershed is urbanizing rapidly and both agricultural and forest lands are quickly being converted to urban lands. If the current population growth rate of 2.4%/year continues the PEI model indicates that population will increase from 20,000 currently to 32,320 in 20 years, a 62% increase. Over the same time period, if existing urban development patterns remain unchanged, the urban land area in the Shelburne Bay watershed in 20 years is estimated to be about 153% of what it is today and agricultural lands would shrink by about at least 20%.

Phosphorus loading to Shelburne Bay from non point and point sources as estimated by the PEI model are given Figure 14 for Simulation Run SB1. In Simulation Run SB1, current management conditions for urban and agricultural lands remained constant over the 20-year simulation period. Phosphorus loadings from forest lands are small compared to loads from agricultural and urban lands and point source discharges. Urban non point source loads of phosphorus substantially increased over time while agricultural non point source loadings decreased somewhat as agricultural land area shrunk. The current total phosphorus load to Shelburne Bay is estimated by the PEI model to be 10.3 mt TP/year and 13.4mt TP/year 20 years from now.

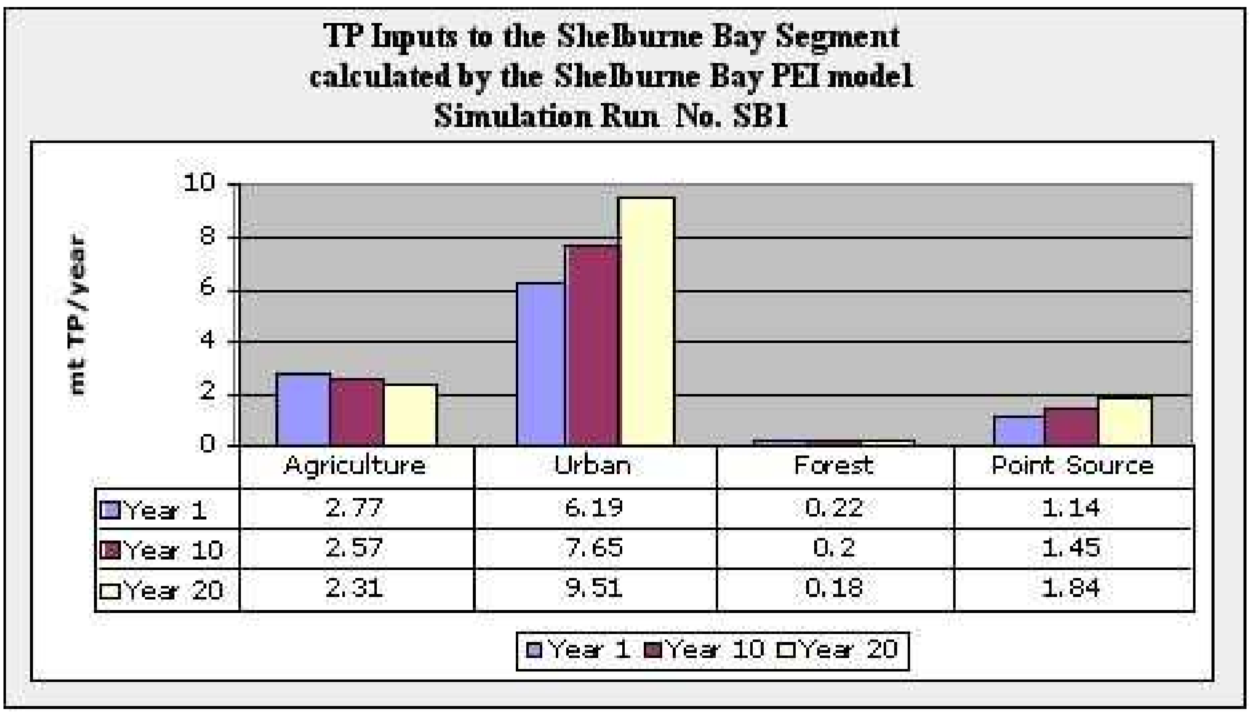



**Figure 14: Phosphorus inputs to the Shelburne Bay segment as calculated by the Shelburne Bay PEI model. The data shown are for Simulation Run SB1 in which, except for a 2.4%/year increase in watershed population, current conditions of wastewater treatment and land management were assumed to remain constant over the 20 year simulation period.**

Four simulation runs within the PEI model were run to assess the sensitivity of phosphorus concentrations in the water column to alternative P management strategies implemented within the Shelburne Bay watershed. This sensitivity analysis is summarized in Figure 15. Simulation Run SB1 assumed a population growth of 2.4%/year and no change from current management of urban and agriculture over the 20-year simulation period. The population growth rate remained at 2.4%/year in Simulation Runs SB2 and SB3, but phosphorus management programs were implemented during year 5 to reduce the non point source urban phosphorus load by 70% in Run SB2 and both urban and agricultural non point source loads by 70% in Run SB3. Simulation Run SB4 is identical Run SB3 except that the population growth rate was reduced to 1.2%/year during year 5.

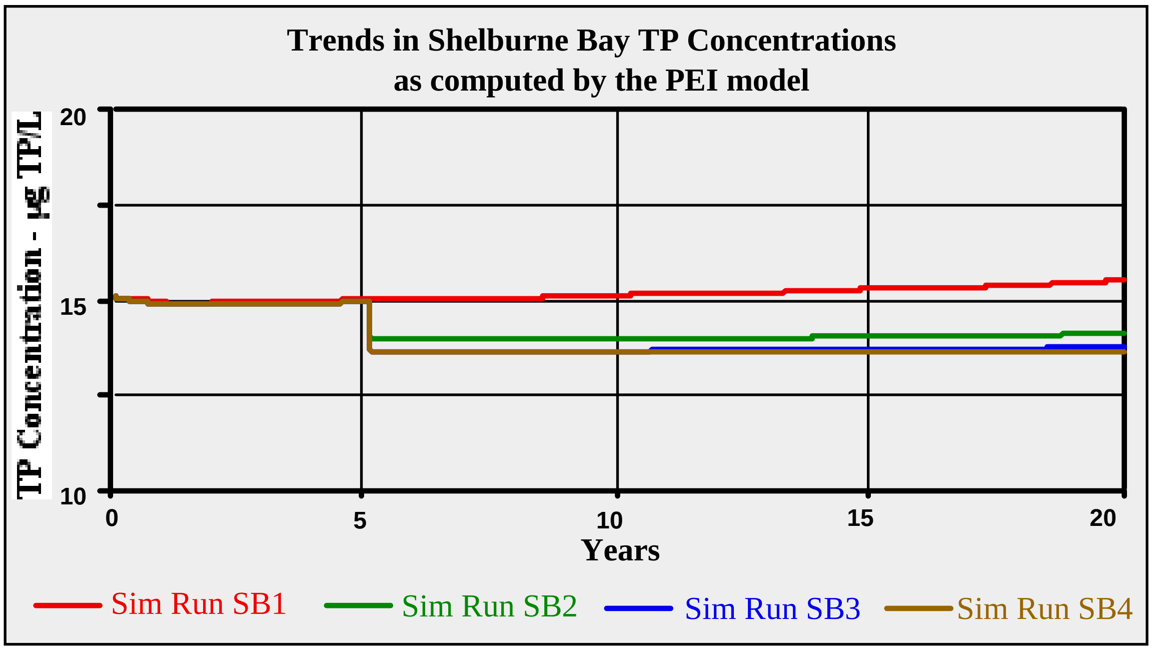



**Figure 15: Sensitivity of TP concentration in the water column of the Shelburne Bay segment to alternative phosphorus management scenarios as computed with the PEI model. Simulation Run SB1 assumes a 2.4%/year population growth rate, and that urban development patterns continue as at present and land and wastewater management remain unchanged over the 20 years. Simulation Run SB2 is the same as SB1, except the TP in urban runoff is reduced by 70% in year 5. Simulation Run SB3 is the same as SB2 except that additionally the TP in urban runoff is also reduced by 70% in year 5. Simulation Run SB4 is the same as SB3 except that additionally the population growth is reduced to 1.2%/year in year 5. The PEI model computes non-point source TP loads based on current agricultural, urban and forestry TP export coefficients of 0.42, 1.50 and 0.04 kg TP/ha/year, respectively.**

For Run SB 1, in which all watershed management remained constant at current levels for the entire simulation period, the TP concentration in Shelburne Bay increased by about 2.5% over 20 years to 15.47 μg/L. In Runs SB2 and SB3 urban and urban and agricultural non point source phosphorus loads were reduced from current levels by 70%, respectively. These reductions in phosphorus loading during year 5 decreased the Shelburne Bay TP concentrations about 6.4% and 8.7% by year 20, respectively. Therefore, the TP concentration of Shelburne Bay water column is somewhat sensitive to urban and agricultural non point source loadings. The decrease in population growth rate in Simulation Run SB4 had little additional effect on the TP concentrations in Shelburne Bay. The trend lines for all simulation runs continued a slight increasing trend during years 5 - 20 suggesting that phosphorus inputs continued to exceed outputs.

The PEI model estimates that about 57.5 mt TP/year enters Shelburne Bay if all the exchange flow comes from the Main Lake segment or 64.2 mt TP/year if all the exchange flow came from the Burlington Bay segment. Thus, the input of phosphorus to Shelburne Bay in exchange flow is about 6 times the phosphorus load from the watershed itself. Figure 16 illustrates the sensitivity of TP concentration in Shelburne Bay to hypothetical phosphorus concentrations in the exchange flow that range from 10 to 14 μg/L

In this analysis it does not matter whether the exchange flow is from the Main Lake or Burlington Harbor or, as is most likely, a mix of the two. The green dot represents the current phosphorus concentrations for Simulation Runs 1SB - 4SB. The concentration of phosphorus in Shelburne Bay is directly related and quite sensitive to the phosphorus concentration of the exchange flow. In this case for every 1 μg/L change in exchange water concentration there is a 1.25 μg/L change in Shelburne Bay phosphorus concentrations.

This level of sensitivity exists because of the high exchange flow mixing between the Shelburne Bay segment and the Main Lake and/or Burlington Harbor segments. On the other hand, for each 10% decrease in nonpoint source phosphorus loads from the watershed there was, for Simulation Run 3SB, a 1.25% decrease in the TP concentration in Shelburne Bay. Is the quality of water in Shelburne Bay more sensitive to the phosphorus levels in the exchange flows from the Main Lake and/or Burlington Harbor than to phosphorus loadings from the Shelburne Bay watershed? Further examination of this question may lead to alternative management strategies not now under consideration. Additional study appears to be warranted.

**The Missisquoi Bay Lake Segment**

At an average depth of about 3 meters the Missisquoi Bay lake segment is the shallowest in Lake Champlain. It drains a watershed of about 284,000 ha (56% USA and 44% Canada) within which some 34,000 people live (70% Canada and 30% USA). While about 68% of the land area is in forest some 27% is agricultural. Intensive agriculture is the predominant activity in the watershed.

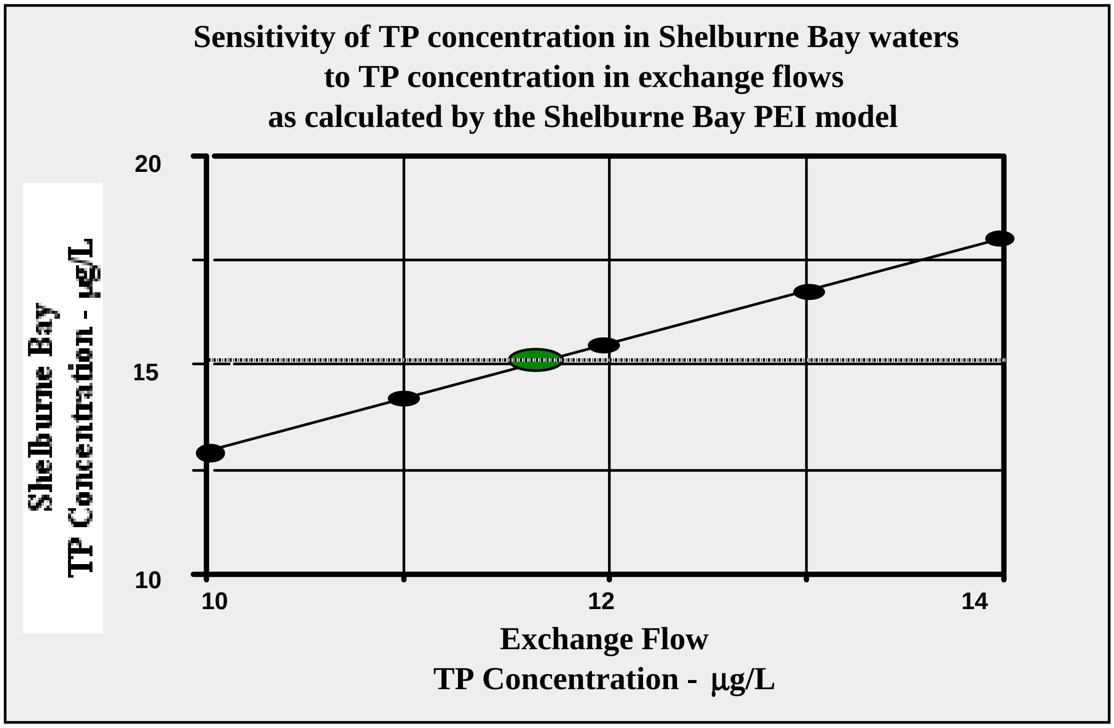



**Figure 16: Sensitivity of TP concentration in the water column of Shelburne Bay to the TP concentration of the inflow exchange waters from adjacent lake segments as computed with the PEI model. This relationship is valid for Simulation Run No. SB1 which assumes a 2.4%/year population growth rate, that current urban development patterns continue and land and wastewater management remain unchanged in the Shelburne Bay watershed over the 20 year simulation period. At inflow exchange flow TP concentrations higher than about the current concentration of 11.7 µg /L (green dot) the TP concentration of Shelburne Bay water at the end of 20 years (solid line) will be greater than the initial concentration (hashed line). The reverse is true whenever inflow exchange flow TP concentrations are less than about 11.7 µg /L.**

For the period 1992-2000 the phosphorus concentration in Missisquoi Bay averaged about 44.9 µg TP/L, nearly 80% above the 25 µg TP/L standard. Consequently, aquatic macrophyte and algae growth have been prolific during summer periods and this growth has adversely impacted recreational use. Monitoring data suggest that the non point and point source phosphorus loads entering the Bay averaged about 153 mt TP/year and 5 mt TP/year, respectively, over the 1995-2000 period (Medalie and Smeltzer 2004). Non point source phosphorus inputs overwhelmingly derive from agricultural activity in the Missisquoi Bay watershed.

Values for some input parameters for the Missisquoi Bay PEI model are summarized in Table 11. These parameter values reflect the current conditions for the watershed area that drains directly into the Missisquoi Bay segment. At a sustained population growth rate of 1.2%/year, the PEI model calculates a population increase of about 9,200 people in 20 years (from 34,000 to

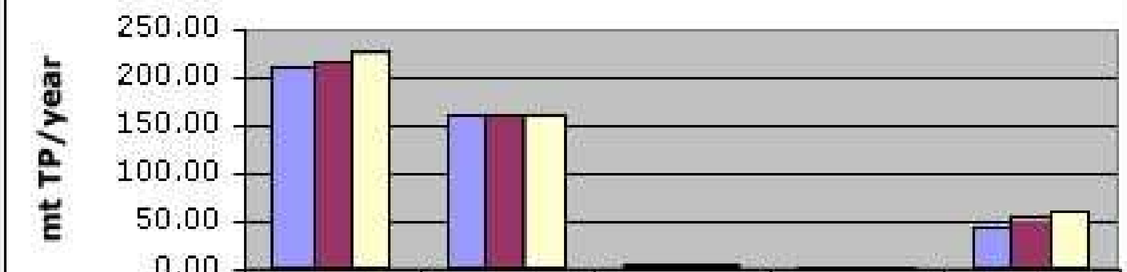
43,200) for the Missisquoi Bay watershed area (Table 11). This 27% population growth increases the amount of urban land area by about 6% while the areas devoted to forests and agriculture decrease slightly.

**Table 11: Some input parameter values and phosphorus loadings for the Missisquoi Bay segment. The items marked with an asterisk are inputs to the PEI model that define current conditions (ca. 2000). All other values were computed with the Missisquoi Bay PEI model assuming all conditions in the watershed that drains directly into the Missisquoi Bay remain unchanged over 20 years except that population grew 1.2%/year. This is Simulation Run MB1.**

| Variable | Units | Current | 20 years in future |
|---|---|---|---|
| Population - USA | No. people | 10,000* | 12,710 |
| Rate of Growth | %/year | 1.2* | 1.2 |
| Population - Canada | No. people | 24,000* | 30,510 |
| Rate of Growth | %/year | 1.2* | 1.2 |
| Urban Land Area - USA | ha | 8,490* | 8,980 |
| P Export Coeff | kg TP/ha/yr | 1.5* | 1.5 |
| Urban Land Area - Canada | ha | 6,150* | 7,320 |
| P Export Coeff | kg TP/ha/yr | 1.5* | 1.5 |
| Agr Land Area - USA | ha | 37,540* | 37,300 |
| P Export Coeff | kg TP/ha/yr | 1.73* | 1.73 |
| Agr Land Area - Canada | ha | 39,230* | 38,060 |
| P Export Coeff | kg TP/ha/yr | 1.63* | 1.63 |
| Forest Land Area - USA | ha | 113,530* | 113,280 |
| P Export Coeff | kg TP/ha/yr | 0.04* | 0.04 |
| Forest Land Area - Canada | ha | 79,350* | 78,760 |
| P Export Coeff | kg TP/ha/yr | 0.04* | 0.04 |
| **Nonpoint Source TP Loads - as calculated by the PEI model** | | | |
| Agricultural – USA | mt/year | 65.0 | 64.5 |
| Agricultural – Canada | mt/year | 63.9 | 63.0 |
| Forest – USA | mt/year | 4.5 | 4.5 |
| Forest – Canada | mt/year | 3.2 | 3.2 |
| Urban - USA | mt/year | 12.7 | 13.5 |
| Urban - Canada | mt/year | 9.2 | 11.0 |
| | | | |
| Point Source TP loads - USA | mt/year | 0.6 | 0.7 |
| Point Source TP loads - Canada | mt/year | 1.4 | 1.7 |
| | | | |
| Total TP load to Missisquoi Bay | mt/year | 160.5 | 162.1 |

The PEI model can be extremely useful to assist managers understand how the Missisquoi Bay processes phosphorus, how management programs might perform over the long term and even suggest previously unconsidered new management options. With this understanding improved management of the lake and watershed resource may be possible. For example, the PEI model suggests that, with the exception of inevitable population growth (1.2%/year), if current conditions were to remain constant over the next 20 years (Simulation Run No. 1), the total input of phosphorus to the Bay from its watershed would increase from the present 160.5 mt TP/year to 162.1 mt TP/year in 20 years (Table 11). This increase isn't much but what does this really mean?

First, phosphorus enters Missisquoi Bay not only in nonpoint and point sources from its watershed but also in exchange flow with the Northeast Arm lake segment and internal loading from bottom sediments. For Simulation Run MB1, the PEI model estimates that the total input of phosphorus to Missisquoi Bay from all sources increases from its current level of 207.8 mt TP/year to 224.5 mt TP/year (Figure 17), mostly from increasing amounts of phosphorus from the bottom sediments (internal loading) although nonpoint sources are the largest external inputs. Simulation Run MB1 suggests that internal loading is an important issue in understanding how Missisquoi Bay processes phosphorus.




| | TOTAL INPUT | Non Point Inputs | Exchange Flow Inputs | Point Inputs | Inputs from Sediment |
|---|---|---|---|---|---|
| YEAR 1 | 207.80 | 158.60 | 4.20 | 2.00 | 43.00 |
| YEAR 10 | 216.60 | 159.10 | 4.20 | 2.30 | 51.00 |
| YEAR 20 | 224.50 | 160.00 | 4.20 | 2.50 | 58.10 |

**Figure 17: Phosphorus inputs to the Missisquoi Bay segment as calculated by the Missisquoi Bay PEI model. The data shown are for Simulation Run MB1 in which, except for a 1.2%/year increase in watershed population, current conditions of wastewater treatment and land management were assumed to remain constant over the 20 year simulation period.**

Because of the shallow water depth and potential resuspension of bottom sediments, the internal cycling of phosphorus between the water column and sediments is expected to be substantial. The average phosphorus sedimentation rate in Missisqoui Bay is about 1.09 g TP/m$^2$/year (HydroQual, Inc. 1999) meaning about about 98 mt TP/year settles to the bottom sediment, about 62% of the 158.6 mt TP/year input in non point source runoff. HydroQual, Inc. (1999) also reported that phosphorus moves from bottom sediments to the water column at a rate of 1.03 g TP/m$^2$/year (50% SD). Based on this, the PEI model estimates 43 mt TP/year currently moves from the sediment into the Misssiquoi Bay water column. Therefore, only 42% of the phosphorus that settles to the bottom sediments moves into the water column while 58% accumulates in the sediments. As phosphorus accumulates in the sediment the rate at which phosphorus moves from bottom sediments to the water column can be expected to increase.

An additional, and one of the more powerful uses to which the PEI models can be put is to examine the sensitivity of an environmental indicator (e.g., phosphorus and/or chlorophyll *a* concentrations in the water column) to various management possibilities. For example, will the implementation of a particular program that is designed to reduce the agricultural non point source TP loading tend to reduce levels of phosphorus concentration in the Bay waters? Simulation Run MB1, in which present-day management is assumed to continue unchanged over the next 20 years, suggests that the TP concentration in the Bay will increase about 15% from 44.9 μg TP/L currently to 51.7 μg TP/L by year 20 even though there was little increase in the phosphorus loading from the watershed. According to the PEI model only the phosphorus input moving from the bottom sediments to the water column substantially increased during this simulation run suggesting that the TP concentration is sensitive to levels of internal phosphorus loading. A 20% increase in internal loading (Figure 17) appears to lead to a 15% increase in water column phosphorus concentration (Figure 18) for Simulation Run MB1.

Figure 18 displays the sensitivity of phosphorus concentration of the Missisquoi Bay water column to three alternative management programs designed to reduce the input loading of non point source phosphorus from agricultural lands. The first Simulation Run, MB1, assumes, as stated above, that present-day management does not change from current conditions for the next 20 years. This simulation run emulates status quo. Simulation Runs MB2 and MB3 emulate new management strategies designed to reduce TP inputs to the Bay in agricultural nonpoint source discharges by 25% and 75%, respectively.  Both strategies are implemented during year 5.

For both simulations, because of Missisquoi Bay's very short retention time, phosphorus concentration precipitously decreases during year 5 then, for simulations MB1 and MB2, begins to gradually increase over the duration of the 20-year period (Figure 18). This increase is due largely to internal phosphorus loadings from the sediments. For Simulation Run MB3, the lake phosphorus concentration continues to decrease slightly over the period after implementation of management that reduces 75% of the nonpoint source phosphorus loading. For management strategies that reduce nonpoint source phosphorus loadings by 0%, 20% and 75 % during year 5, phosphorus levels in Missisquoi Bay are estimated to change by year 20 by about +15%, 0% and -33%, respectively, when compared to the current level of 44.9 μg /L. The PEI model suggests that the concentration of phosphorus in Missisquoi Bay is sensitive to watershed management strategies for reducing nonpoint source phosphorus loadings, however, the model also suggests that large decreases in nonpoint phosphorus loading are needed before the Bay waters will likely see sustained long-term trends of decreasing phosphorus concentration.

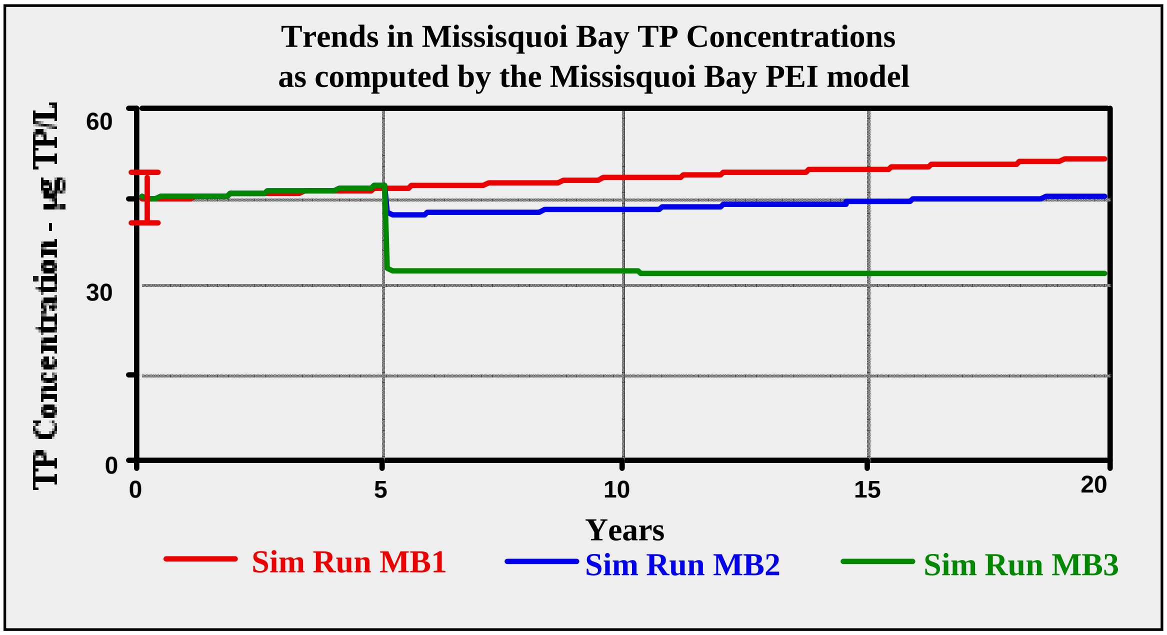



**Figure 18: Sensitivity of long-term changes in TP concentrations in the water column of Missisquoi Bay to levels of reduction in non point source TP loading as computed by the Missisquoi Bay PEI model. From years 0 to 5 current watershed conditions are assumed to remain unchanged. During year 5 the model emulates reductions of 0%, 25%and 75% of current non point source TP loadings for Simulation Runs MB1, MB2 and MB3, respectively. The population is assumed to grow at 1.2%/year over the 20 year period. The error bar is +/- one standard error based on 1992 - 2002 monitoring data.**

The bar on Figure 18 shows an interval of +/- one standard error (average from the 1993-2002 TP monitoring data). It appears that it may be difficult to discern long-term differences in phosphorus concentrations among simulation runs MB1 and MB2 but the impact of simulation run MB3 may become apparent. As was the case with the Main Lake, additional study is needed to determine how TP monitoring data may be collected and analyzed to maximize our ability to detect small changes in TP concentration over time.

PEI model Simulation Runs MB1, MB2 and MB3 (Figure 18) suggest that programs which reduce nonpoint source phosphorus loads have are not likely to produce decreasing trends in phosphorus concentration over the next twenty years and indicate that internal loading is a large input of phosphorus to the water column. Therefore, we decided to use the PEI model to explore one other potential management approach, namely, an approach that reduces the internal phosphorus loading. Simulation Run MB4 explored this management option. This run is identical to Simulation Run MB2, except that, each year between years 10 and 20 about 3% of the phosphorus in bottom sediment is removed. In the real world, this could be accomplished by dredging, macrophyte removal or by some means of physical or chemical containment of the phosphorus within the sediment. The environmental impacts to biota of such a management approach might be significant and would need considerable additional study.

Figure 19 compares the long-term trends in water column TP concentrations for runs MB2 and MB4. In Simulation Run MB4, beginning with year 10, the amount of phosphorus in the bottom sediments shows a decrease because of the annual removal operations simulated in the PEI model. Since internal loadings of phosphorus is proportional to the amount of phosphorus stored in the bottom sediments, the TP concentration of the Missisquoi Bay waters also show a decreasing trend. Thus, the PEI model suggests that management options that reduce the amount of phosphorus in the bottom sediments and/or that make the stored phosphorus unavailable for internal loading might lead to long-term reductions in the TP concentration of Missisquoi Bay waters. It must be noted, however, that such improvement in water quality may be difficult to discern because of the high level of variability in the monitoring data.

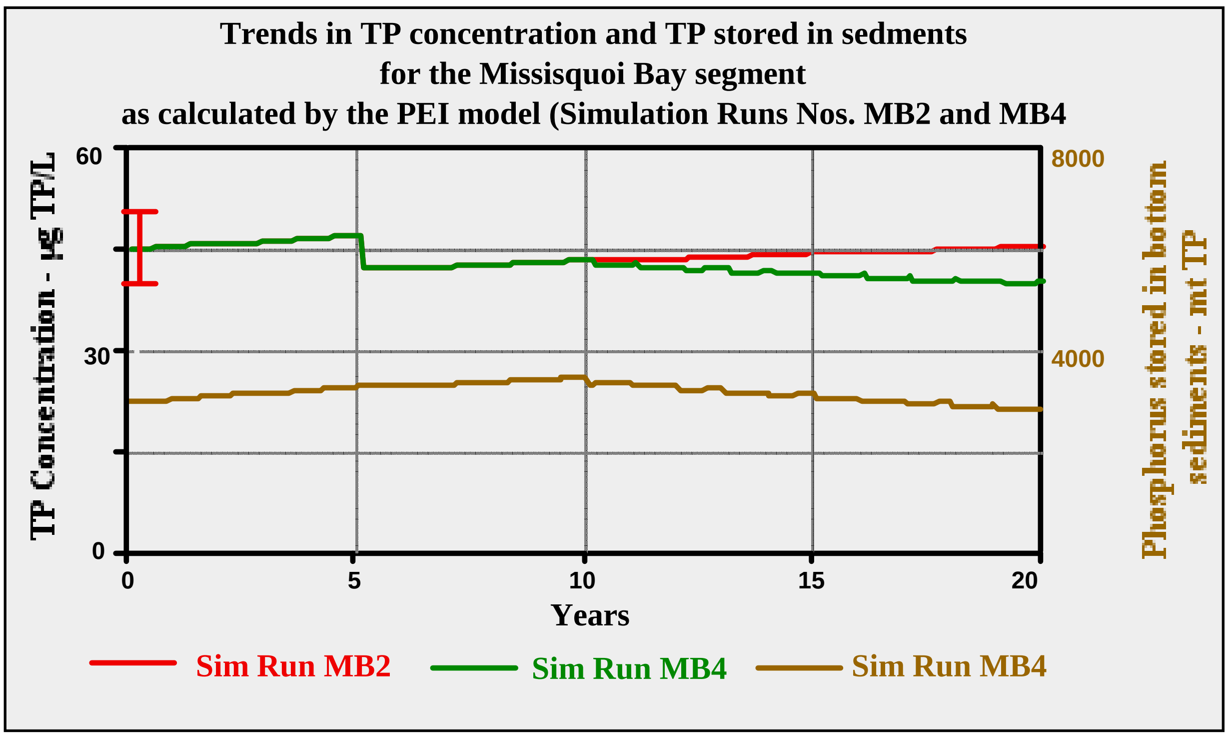



**Figure 19: Comparison of sensitivity of long-term changes in TP concentrations in the water column of Missisquoi Bay between Simulation Runs MB2 and MB4 as computed by the Missisquoi Bay PEI model. Simulation Run MB2 assumes only a 25% reduction in non point source TP loading in year 5. Simulation Run MB4 is the same as Simulation Run MB2 except that, additionally, the PEI model simulates making portions of the phosphorus stored in the bottom sediments unavailable for internal loading. In Simulation Run MB4 about 3% of the phosphorus stored in the sediments is removed each year between years 10 and 20. The population is assumed to grow at 1.2%/year over the 20 year period. The error bar is +/- one standard error based on 1992 - 2002 monitoring data.**

The PEI model also tracks chlorophyll *a* concentration over time based on the following linear regression relationship in which Chl *a* and TP concentrations are in mg/L.

$$\ln(\text{Conc Chl } a) = 5.1512 + 0.7872 * \ln(\text{TP Conc})$$

$(R^2 = 0.5734, p < 0.0001)$

This relationship was derived from the annual means of the TP and chlorophyll *a* concentration from samples collected at 52 lake stations in 1991 and 92 (VTDEC and NYSDEC 1994). Figure 20 shows the trends in average annual chlorophyll *a* concentrations for Simulation Runs MB1, MB2 and MB3. The patterns of change in chlorophyll *a* mirror the trends in TP concentrations in Figure 18.

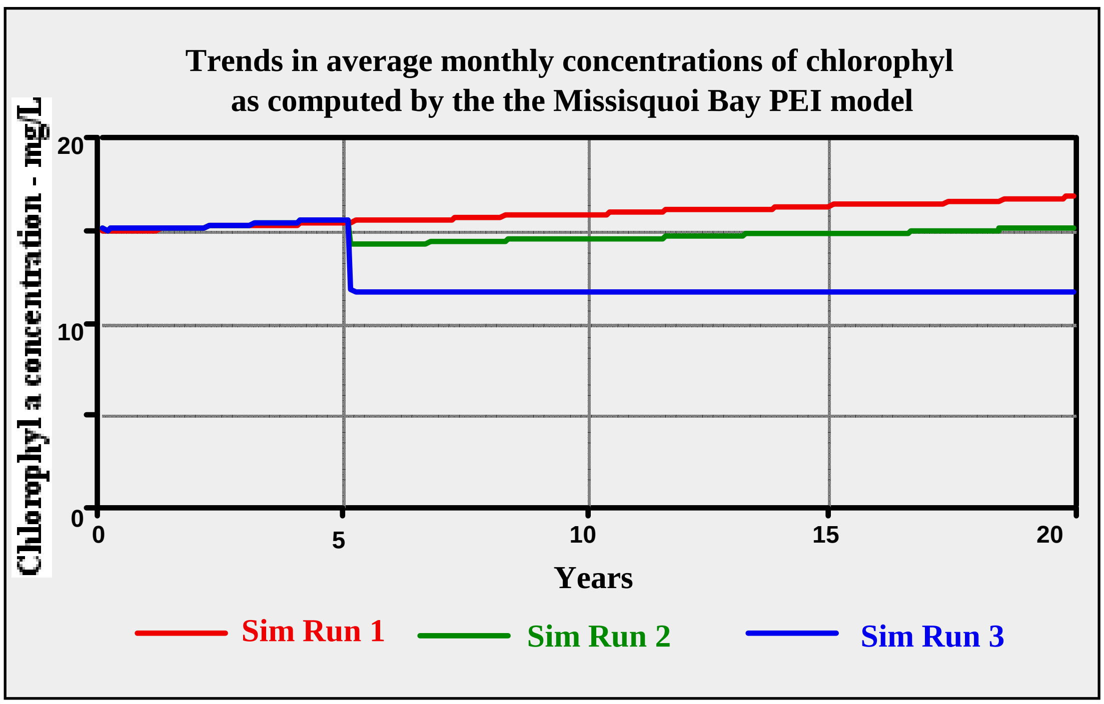



**Figure 20: Sensitivity of average monthly chlorophyll *a* concentration in Missisquoi Bay as computed with the PEI model to different soil erosion/nutrient management strategies for agricultural fields. Simulation Runs MB1, MB2 and MB3 are trends for phosphorus management options implemented during Year 5 that yield, respectively, 0%, 25% and 75% reductions in non point source TP loading to the Bay. The population is assumed to grow at 1.2%/year over the 20 year period.**

**Discussion - Mass Balance and PEI Modeling**

We believe that dynamic mass balance PEI models are useful to help guide the process of thinking about natural dynamics, indicator selection and management options. They are also very useful in exploring management scenario and potential future conditions under these scenarios.

In Figure 12 we presented a diagram that illustrated the mass balance concepts upon which our PEI models are based. In this discussion it is useful to consider the concept of mass balance at two different scales, the lake-scale in which Lake Champlain is a single lumped entity and the segment-scale in which considers just one of Lake Champlain's segments. Compartment-flux diagrams for each are shown in Figure 21. They, in many ways, are similar. But a careful consideration of each leads us down somewhat different pathways with regard to choosing phosphorus pressure - state - response indicators.

Regardless of scale, whenever the sum of the inputs (loadings) exceed all the outputs, the amount of any element or pollutant stored inside the lake or lake segment increases. For phosphorus, as the stored amount increases, we typically see increased levels of eutrophication and society responds by attempting to reduce phosphorus point and nonpoint source loading.

At the lake-scale (Figure 21A) the phosphorus loadings to the lake include both point and nonpoint sources. For thousands of years before Europeans arrived in the Lake Champlain basin, soil particles and associated nutrients were carried into the lake in surface runoff (now called nonpoint source loading). These soil particles accumulated to form the lake bottom sediments, which have always contained some phosphorus. As the basin was developed over the last 300 years, surface runoff continued to carry sediments into the lake along with increasing amounts of phosphorus. As industrial activity and the sewering of municipalities in the basin increased, the number of point sources discharging into the waters of the basin multiplied. Consequently, phosphorus has accumulated in the bottom sediments of Lake Champlain from thousands of years of natural geologic erosion processes and about 300 years of progressively more intensive human activity. Lake Champlain has a past and the eutrophication we now see is, in part, a consequence of that past.

What, at the lake-scale, might be societal responses to reduce this eutrophication? Notions of mass balance say that as long as inputs exceed outputs accumulation will continue. The compartment-flux diagram for this scale (Figure 21A) indicates only two management options are available: namely, to lower inputs is by reducing point source loadings or by reducing nonpoint source loadings. Response indicators that track, over time, the accomplishment of programs to reduce these loadings might be:

> a. For Point sources - track average annual monthly phosphorus concentration and average annual monthly flows for all industrial, commercial, municipal and private wastewater treatment plants. The product of these indicators would be average annual mass output due to point sources.

b. For nonpoint sources – because nonpoint source loadings are typically very difficult to measure directly, indicators might track level of implementation of urban and agricultural BMPs, ratio of animal numbers/animal units per ha cropland, the phosphorus soil index, or other surrogate parameters.

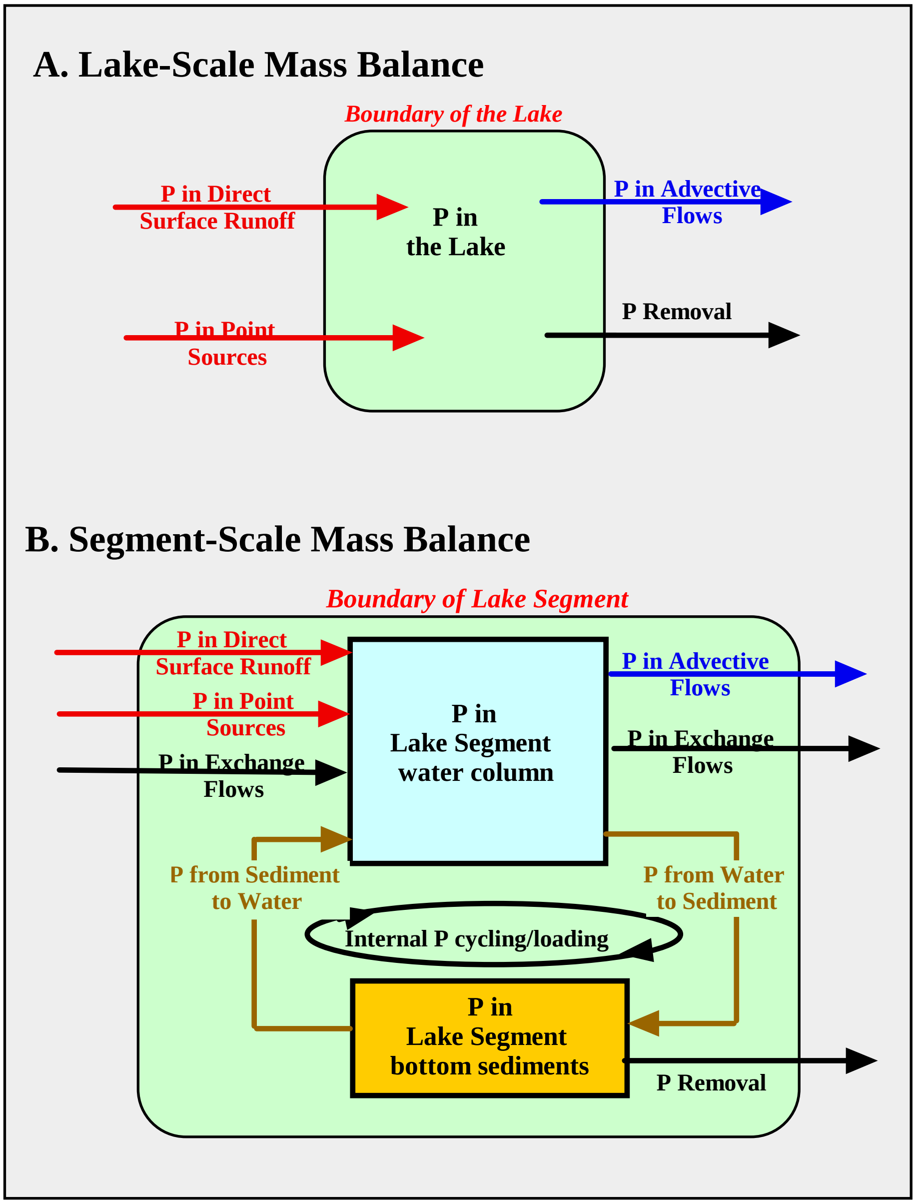



**Figure 21: Compartment flux diagrams illustrating mass balance at the lake-scale and the segment-scale. These diagrams help guide the selection of phosphorus indicators.**

Even when change in such lake-scale indicators suggests that society is getting better at preventing phosphorus from entering the lake, the next question might be are eutrophication levels also being reduced? To determine this, it is necessary to measure indicators of the state of the lake. Such indicators might include measures of TP and/or chlorophyll *a* concentration in the lake water. The LCBP's monitoring program already tracks such indicators. However, the literature is full of reports in which programs designed to reduce nonpoint source phosphorus loadings from watersheds could not, over the period of study, definitively demonstrate change in phosphorus levels in the receiving body of water.

Additionally, there are confounding issues in lake ecosystems. It is known, even in the absence of phosphorus inputs, that the phosphorus in lake sediments can move into the water column (internal loading), but the rates are frequently so slow that it may take decades or even centuries to substantially deplete the sediment phosphorus store. Therefore, decreases in eutrophication levels would occur slowly even if inputs abruptly decreased. Thus, it is not clear, for a complex lake system such as Lake Champlain that changes in state indicator levels can be directly related to change in response indicators. Consequently, indicators within this lake-scale context need to be designed to look at the long-term and not address the shorter-term year-to-year variation seen in monitoring data. These long-term indicators need to measure changes in public policy, public expectations and social norms within the urban and agricultural sectors to measure progress towards the long-term objective.

The segment-scale (Figure 21B) is the same compartment-flux diagram used in our Main Lake, Shelburne Bay and Missisquoi Bay PEI models. In addition to point and nonpoint source input loadings shown in the lake-scale diagram, phosphorus enters the lake segment in advective and exchange flow from adjacent segments. Our PEI models suggest that phosphorus inputs in advective and exchange flows currently are about 11X and 6X the nonpoint source loading for the Main Lake and Shelburne Bay segments, respectively. Consequently, the level of eutrophication (as indicated by the TP concentration of water column) is not very sensitive to reductions of point and nonpoint source inputs in the watershed.

In our 20-year simulation runs for each segment in which nonpoint source loading was reduced by 70% in year 5, the TP concentration in the water column in year 20 was estimated to change by + 6.3% and - 6.4% from current TP levels in the Main Lake and Shelburne Bay segments, respectively. For both segments the PEI model estimate of TP concentration in 20 years remained above the current water quality standard. It is difficult to imagine reductions in current nonpoint source loading of 70%, let alone any more than this.

The phosphorus loading in the exchange and advective flows, in fact, could be reduced by decreasing the TP concentrations in the connected segments. Importantly, these segment-scale mass balance models incorporate the fact than many of Lake Champlain's segments are closely linked by these exchange and advective flows. Such linkages mean that changes made in the watershed of one segment may have a substantial effect on eutrophication levels in adjacent linked segments. For such lake segments it appears unlikely that eutrophication levels can be related definitively to reductions in point and nonpoint sources solely in their own watersheds. For example, the Main Lake PEI model suggests that the TP concentration in the South Lake is a primary driver of the TP concentration in the Main Lake. Similarly, the Shelburne Bay PEI model indicated that the TP concentration in the Main Lake is a primary driver of the TP concentration in the Shelburne Bay. Clearly, management efforts to reduce phosphorus inputs and TP concentrations in the South Lake indirectly influence the water quality of the Main Lake and Shelburne Bay segments. Although this fact is acknowledged in the original model developed as part of the Diagnostic-Feasibility Study (Smeltzer and Quinn 1996) and the Lake Champlain phosphorus TMDL (VTDEC and NYSDEC 2002), this management approach deserves additional consideration.

The Missisquoi Bay lake segment is very different from the Main Lake and Shelburne Bay segments. The PEI model estimates the current nonpoint source phosphorus input to Missisquoi Bay to be about 37X of exchange flow input. We expected that water column TP concentration in Missisquoi Bay would be very sensitive to reductions in non point source loadings. To check this expectation we ran a series of 20-year simulation runs in which current agricultural nonpoint source loading was reduced by 0%, 25% and 75% in year 5. The estimated TP concentration at year 20 was found to be sensitive to nonpoint source reductions. Some 15 years after the reduction occurred, TP concentrations were +15%, 0% and -33% of the current TP concentration, respectively. However, even with a massive 75% reduction in nonpoint source loading, the PEI model estimated that the current TP concentration standard could not be achieved by year 20.

The Missisquoi Bay PEI model also allows assessment of internal phosphorus loadings. The current internal loading of phosphorus is estimated at about 27% of the nonpoint source inputs and will increase to about 36% by year 20. The model indicates that phosphorus accumulation continues over the 20 year period. This large increase in internal loading will likely mask much of the hoped-for effects of reductions in nonpoint source inputs. However, when nonpoint source inputs are reduced by 25% in year 5 AND 3% of the accumulated sediment phosphorus is removed from the Bay each year, TP concentration is reduced by 10% of the current TP concentration by year 20 AND, importantly, there is a continuing downward trend in TP concentration.

The segment-scale PEI models allow examination of the uniqueness of each segment. Is the segment shallow or deep, is the watershed land-use agricultural, urban and/or forested, are exchange/advective flow inputs large or small, is internal phosphorus cycling important or minimal, and other questions. With such differences, each segment we examined with the PEI model processed phosphorus in a unique fashion. So at the segment-scale, the indicators that might be chosen to track changes should also probably be different. At this scale the options include lowering inputs by reducing point and nonpoint source loadings as well as reducing the inputs in exchange and advective flows and in the internal sediment loading. Response indicators for an individual lake segment might include:

a. For point sources - track average annual monthly phosphorus concentration and average annual monthly flows for all significant point source discharges. The product of these indicators would be average annual mass loading resulting point sources.

b. For nonpoint sources - track the level of implementation of urban and agricultural BMPs, ratio of animal numbers/animal units per ha cropland, the phosphorus soil index, or other surrogate parameters.

For lake segments that are highly connected, addition, these response indicators would also need to be tracked in the watersheds of those linked segments. Specific response indicators would likely differ from one segment to the other depending on segment characteristics and pressures.

State indicators at the segment-scale include measures of TP and chlorophyll *a* concentration of the water column. Such indicators can give an assessment of eutrophication level and possibly indicate trends within the constraints imposed by monitoring data variability. However, it is not at all clear, even at the segment-scale, that change in state indicators can be definitively related to change in response indicators. At the segment-level the PEI models can provide an opportunity to tailor the choice of indicator to the uniqueness of the segment itself and its linkages to adjacent segments.

Our dynamic mass balance PEI models are but models, and thus not necessarily reality. Outputs from the models are only as good as the data that goes into them, the algorithms used and the organization of the algorithms within the model. In the development of these models a number of issues arose for further consideration:

> (a) Monitoring programs provide TP and chlorophyll *a* concentration data for about 6 months each year and assess phosphorus loading output from selected watersheds. Even though these data incorporate considerable natural variation, they were useful for initializing and calibrating the model, however, additional data would add greatly to the precision of the model estimates.
>
> (b) Estimates of the average annual advective and exchange flows among lake segments are available, however, little is known of seasonal differences or of natural variation in these estimates.
>
> (c) HydroQual, Inc. (1999) has provided much basic information on internal phosphorus cycling but a better understanding of these processes in Lake Champlain is urgently needed. Spatially explicit estimates of the rate constants that drive internal loading are needed for those segments where this is a dominating process.
>
> (d) Lastly, phosphorus export coefficients that define phosphorus loads from specific land uses are assumed to remain constant over time unless some management program changes it. Research suggests that most practices currently employed on croplands continue to increase the levels of phosphorus stored in the soil over time leading to proportionate increases in the phosphorus export coefficient even with unchanging management practices. We expect that fertilized urban soils change in a similar fashion. Such change needs to be recognized in future models and considered when tracking nonpoint source phosphorus loads.

In summary, our PEI models specifically evaluate the impacts on TP concentration in the lake segments resulting from human activities in the watershed (Table 7) and the changes in the level of management response to control phosphorus inputs. Although the models do incorporate the impact of internal exchange flows (a natural phenomenon) on the lake water quality, they do not assess the random and highly variable effects of changes in temperature, wind direction and velocity, sunlight intensities and periods, and precipitation patterns and intensities. For all lake segments, a portion of the modeled change in TP concentration was associated with anthropogenic pressures and management responses and a portion was the result of natural factors, such as exchange flows and phosphorus movement from sediments to the overlying waters, which cannot be influenced by management actions.

Based on the three PEI models studied in this project, the following factors must be considered in selecting appropriate indicators related to phosphorus pollution.

- Each lake segment is unique and different specific indicators may be necessary to assess the impact of management responses on individual lake segments.
- The impacts of human pressures and management responses on the water quality of a given lake segment are frequently overwhelmed by natural environmental variability, by exchange flows with adjoining lake segments and internal loadings.
- Many lake segments are closely connected by exchange flow with adjacent segments so that water quality in any given segment not only depends on conditions in its own watershed but that in adjacent watersheds as well. Indicators must be selected with these linkages among lake segments in mind.
- TP concentration is not a sensitive indicator of the aggressiveness or effectiveness of management in many lake segments, especially over the short term, but indicators of land use management practices may be.
- More emphasis should be placed on monitoring indicators that track change in land use, land management practices, urban development patterns, and the like.

## ASSESSMENT OF ADEQUACY AND ADDITIONAL MEASURES

The current monitoring efforts for phosphorus are among the most extensive in the Lake Champlain Basin. Despite that, they incorporate considerable variability, do not fully characterize the conditions in the lake and cannot fully inform decision-making. Some of the measures used for our indicator suite are seriously out of date, and some are missing all together. Perhaps of most concern, our pressure indicators rely on land use measures that are more than 10 years out of date. Since these indicators are used to track the nonpoint source load throughout the watershed, they must be updated.

The state indicators attempt to capture the major compartments of phosphorus in the lake. Currently, we only track phosphorus in the water column, and not in the sediments. Our PEI modeling, however, demonstrates that information about phosphorus in the sediment is critical for predicting response times in key lake segments like Missisquoi Bay. Other studies have found similar linkages (Nurnberg 1984, Carvahlo et al. 1995). At least for the shallow lake segments and lake segments where anoxia is possible, sediment data are a critical data gap. We are also not tracking the phosphorus stored in aquatic plant and zebra mussel biomass, which can be important compartments in many lake segments. Again, a focused data collection effort in these areas is also needed.

Our response indicators focus on tracking the major management activities supported by the TMDL and *Opportunities for Action.* Although we are tracking the implementation of a core set of BMPs on farms, we are not tracking the implementation of BMPs in urban and suburban areas. This response indicator is essential for evaluating management success.

Until we fill some of the critical data gaps in this issue area, we will not have a credible way to evaluate our progress towards the phosphorus goals in the TMDL. Although we report on the reductions in phosphorus loads that we have achieved by upgrading our sewage treatment plants and by implementing manure management practices on farms, we do not track the increases in phosphorus load resulting from increases in population, from land use conversions, or from increases in animal densities on farms. Preliminary calculations with our current agricultural credits show that they are overly generous, generating negative loads when applied in some watersheds (Donlon and Watzin 2000). We currently have no credits for urban practices or for many of the more innovative practices on farms. If we are to move forward with integrity, we must present a full balance sheet that accounts to the best of our abilities for all credits and debits.

## B. Issue Area: Bacteria in Recreational Waters

To protect human health, many public beaches on Lake Champlain are monitored for bacteria and may be closed when bacteria levels exceed standards recommended by the state health departments or other agencies. The PSR diagram in this issue area is focused on bacteria levels, the sources of bacteria in the lake, and the management actions taken to protect the public from high bacteria level and to reduce these levels over the longer term (Figure 22). Table 12 presents the indicators for the bacteria issue area.

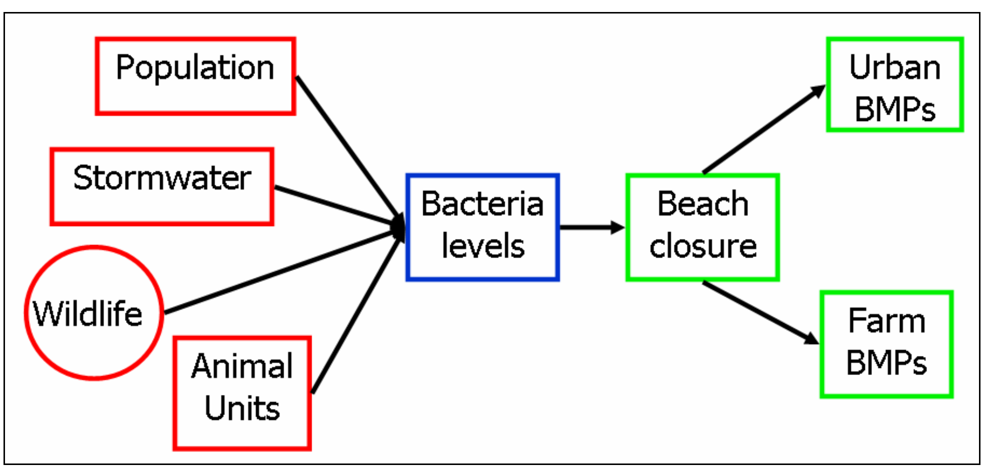



**Figure 22: PSR diagram for bacteria in recreational waters.**

**Table 12: Indicators for bacteria in recreational waters.**

| Indicator | P S R | Available Measure | Year | Source | Ref | Recommended Measure | Frequency (minimum) |
|---|---|---|---|---|---|---|---|
| Population | P | Human population by state/province | US: 1950 - 2000; Canada: 2001 | Holmes and Associates; Statistics Canada | Figure 4 | Human population by lake segment subwatershed | Update every 10 years |
| Stormwater | P | Percent samples that exceed state standard | 2002 | UVM (Burlington Bay only) | Figure 23 | Percent samples that exceed standard | Annually |
| Animal units | P | Stocking density (animal units/ha) by subwatershed | VT:2002;NY:2002; QC:1998-2003 | VT AFM; NYS SWCC | Figure 7 | Stocking density (animal units/ha) by subwatershed | Update every 2 years |
| Wildlife | P | Data unavailable | | | Page 56 | Measure when necessary in problem areas | As needed |
| Bacteria levels | S | Number of beach water samples that exceed state standards | 1997-2002 | | Figure 24 | Percent of beach water samples that exceed state standards at priority locations | Annually |
| Beach closure | R | Days of beach closure at Burlington beaches | 1990-2002 | | Figure 25 | Days of beach closure at all Champlain beaches | Annually |
| Farm BMPs | R | Percent of farms and animal units treated | VT:1996-2003; NY:2002 | VT AFM; NYS SWCC | Figure 10 | Bacteria load reduction from implementation of agricultural BMPs by subwatershed | Update every 2 years |
| Urban BMPs | R | Percent of stormwater permits that have expired | 2003 | VT DEC | Page 57 | Bacteria load reduction from implementation of urban BMPs by subwatershed | Update every 2 years |

## PRESSURE INDICATORS

People, domestic animals and wild animals are the sources of the bacteria found in the water at Lake Champlain beaches. As discussed in the phosphorus section, the human population in the basin is growing (Figure 4). We do not have data on trends in wild or domestic animals, although as the human population increases, the number of pets in the watershed is also likely to increase. Animals found in developed areas contribute to the bacteria that enter the lake because their waste is washed into the stormwater.

Basinwide, there are very limited data on stormwater flow and what is in it, however, monitoring in the Burlington Bay lake segment, which is among the most developed in the watershed, shows consistently high levels of coliform bacteria (Watzin et al. 2003a, 2004). In 2002, the *E. coli* level in 93% of samples collected from stormwater entering Burlington Bay exceeded the Vermont Water Quality Standard of 77 organisms/100 ml of water (Figure 23). Many of these exceedences were by two or three orders of magnitude.

Also similar to phosphorus, the extent of bacterial pollution at Champlain beaches depends not only on the number of people and animals, but also on the activities and behaviors that influence the transportation of bacteria to the beaches. Although very difficult to quantify, failed septic systems, particularly those directly along the lakeshore, are also likely sources of bacteria found in the lake.

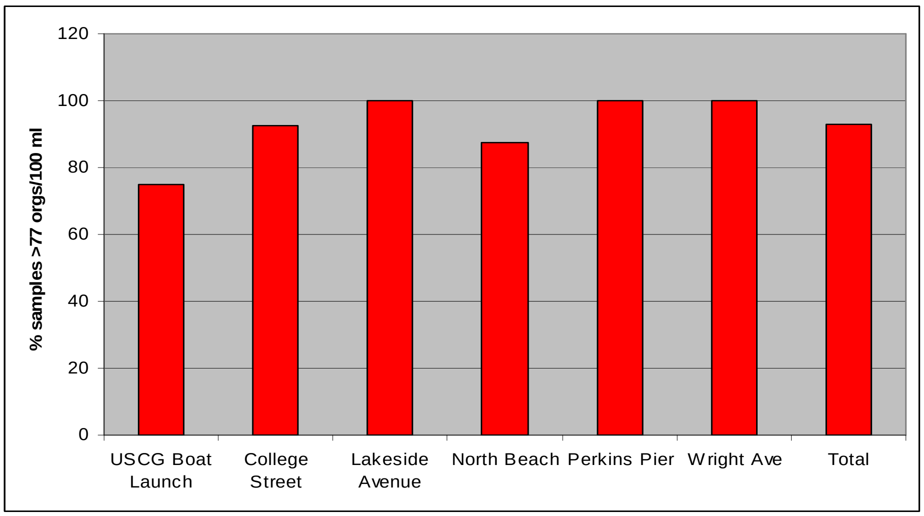



**Figure 23: Percent of stormwater samples with greater than 77 *E. coli* organisms per 100 ml, the VT standard, collected at various sampling sights in Burlington Bay in 2002 (Watzin et al. 2003a).**

Agricultural animals (Figure 7) are also a source of bacteria that can be transported to the lake through the tributaries. The bacterial load in tributaries from agricultural animals depends on factors including whether or not animals have direct access to streams, the characteristics of the riparian corridor, and a variety of other factors (Cassell and Meals 2002).

Wildlife, such as gulls, beaver and deer, may also be important sources of bacteria at Champlain beaches. The fecal matter in surface water from wildlife is generally considered part of the background or baseline level of bacteria in a stream. In 2001 and 2002, a study was conducted in the Mad River Valley, VT to examine baseline *E. coli* levels in streams in forested subwatersheds. Moir (2004) found baseline levels of *E. coli* to be high, particularly during storm events (95% C.I. = 20.9 – 66.4 organisms/100 mL) with 34% of the samples violating the Vermont Water Quality Standard of 77 organisms/100mL.

In an attempt to determine the contribution of wildlife to bacterial pollution at Colchester, VT beaches, a microbial source tracking study was conducted in Malletts Bay and the lower Winooski River in 2001. The DNA of the *E. coli* in the water samples was compared to the DNA of *E. coli* found in various animal feces. Only 28% of the 176 *E. coli* in the sample were positively matched to a host species (Jones 2002). Deer, humans, raccoons, cats, gulls and chickens were among the host species identified in this study. Given the limited data available at this time, the relative importance of wildlife as a source of fecal contamination at Champlain beaches cannot be specified.

## STATE INDICATORS

There are many different types of bacteria that can potentially be found at beaches in high enough quantities to pose a threat to public health. The presence of elevated levels of one type of bacteria suggests the presence of other pathogens at elevated levels. Consequently, a choice has to be made regarding what type of bacteria to monitor. Traditionally, fecal and total coliform were used to monitor fecal pollution. The U.S. EPA now recommends monitoring *E. coli* in freshwater and *Enterococcus* in saltwater or freshwater bodies. (U.S. EPA 1986; U.S. EPA 2002). The Vermont Water Quality Standards specify a single sample density of 77 *E. coli* per 100 ml (Dorfman 2002) as the level necessary to protect human health, and this criterion is used for Impaired Waters listing (303(d)) under the Clean Water Act. In New York, the Department of Health uses a 30-day, five sample geometric mean of 2,400 total coliforms per 100 ml (with a limit on the number of samples exceeding 5,000 total coliforms) or a fecal coliform standard of 1000 organisms per 100 ml in a single sample and geometric mean of 200 per 100 ml as its water quality standard (Dorfman 2002). In Quebec, a standard of 200 total coliforms per 100 ml is used. In both Vermont and New York, the standards are recommendations to public health authorities. Monitoring of public waters is not mandatory and beach closures are left to the discretion of local officials (Dorfman 2002).

Because the type of bacteria monitored and the standards against which bacteria levels are evaluated vary across the political regions of the basin, the data on bacteria levels at Champlain beaches are fragmented and disparate (Figure 24). *E. coli* is measured regularly at state and municipal beaches in Vermont. There is some sparse fecal coliform and total coliform data for municipal beaches in New York. At New York state park beaches, total and fecal coliform were measured regularly in 2001 and *Enterococcus* and fecal coliform in 2002. These differences in standards and measures and the variability in the sampling effort make it difficult to assess and address the problem of bacterial pollution on a lakewide basis. Because the water quality standard in Vermont is more restrictive than that used in New York, it is not surprising that the standard is exceeded more frequently in Vermont than in New York.

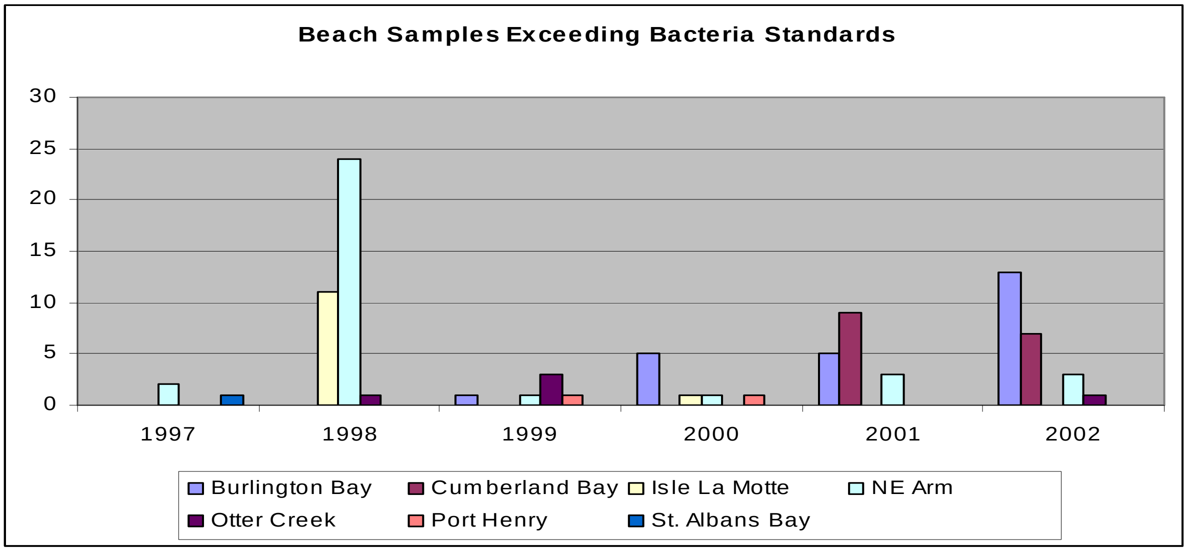



**Figure 24: Number of bacteria samples that exceed NY and VT standards for recreational water bodies.**

## RESPONSE INDICATORS

The primary response to bacteria levels in excess of state standards is the closure of public beaches in order to protect human health. However, as mentioned previously, a measured value in excess of the recommended standard does not necessarily result in a beach closure. There were no reported beach closures at NY State Parks in 2001 or 2002 despite samples that exceeded state standards. Although VT State Park beaches have been closed due to high bacterial levels, we were unable to obtain any record of beach closures. The record of beach closures in Burlington, which extends further back than the *E. coli* sampling data record, shows fewer days of beach closure than there are samples in excess of standards (Figure 25). Beach closure in Quebec is the result of toxic blue green algae blooms, not bacteria contamination.

Although intended primarily to reduce phosphorus loading to Lake Champlain, BMPs can also reduce the amount of bacteria reaching beaches from farms and stormwater. Indicators that relate to farm BMPs and to stormwater management appear in Figure 10 and Table 6.

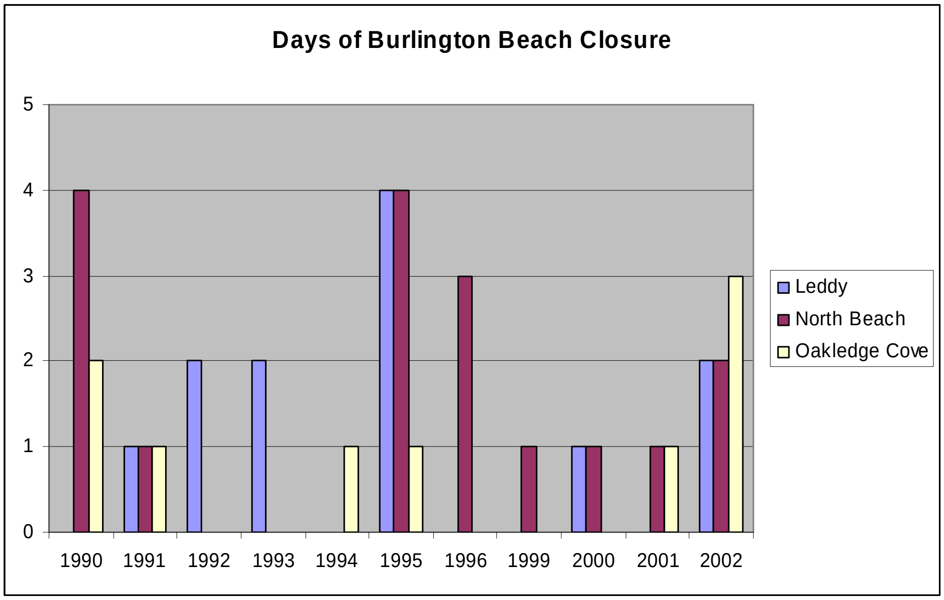



**Figure 25: Days of beach closure at Burlington beaches**

## ASSESSMENT OF ADEQUACY AND ADDITIONAL MEASURES

The pressure indicators include people, farm animals, wildlife and stormwater. Although data are being collected on the human and animal population, the only stormwater data we were able to locate were those being collected by UVM in Burlington Bay as part of a research project that ended in December 2004. An on-going data collection effort in urban areas throughout the basin is needed.

In this area, we have only proposed one state indicator, the level of bacteria in beach water samples. Although this indicator is being monitored at sites throughout the basin, an evaluation of the extent of coverage and the ability to draw general conclusions about bacteria in each lake segment should be undertaken.

The primary management response to unsafe levels of bacteria in beach water is to close the beach. Because of the differences in standards used to make decisions about closure in the three jurisdictions, it is not possible to compare the number of days of closure around the basin credibly. This situation should be addressed in the future. The other response indicators track the major management activities undertaken to reduce the level of contamination and therefore, the number of days of beach closure. Currently, the only data we have on urban BMPs in the number of permits that have expired. A better measure of the urban BMP indicator in the future would be the percent of the stormwater that is being treated. This is a critical data gap, as it is in the phosphorus issue area.

## C. Issue Area: Mercury Toxicity

Mercury and PCBs are the highest priority toxic pollutants in the Lake Champlain basin Program (LCSC 2003). They are both bioaccumulating toxins that are found in fish of Lake Champlain at concentrations that exceed U.S. Food and Drug Administration and U.S. EPA guidelines for the protection of human health. PCBs, polychlorinated biphenyls, are a family of industrial chemicals that have not been manufactured in the United States since the 1970s. PCBs are a concern because they persist in the sediments in some areas of the lake. Beginning in 1999, the largest single source of PCBs in Lake Champlain, the old Georgia Pacific sludge bed in Cumberland Bay, was remediated. PCBs have been significantly reduced in the sediments of Cumberland Bay (NY DEC 2002) and we expect that they will begin to decline lakewide.

Mercury, by contrast, is still in widespread use and continues to enter the lake via atmospheric deposition, tributaries and point source discharges (Shanley et al. 1999, Gao et al. in press). The load that is entering the lake through the tributaries is directly related to the mercury that is falling out on the land through atmospheric deposition. Mercury bioaccumulates through the food web and is a public health concern for people that consume contaminated fish. It can also impair the reproductive health of the fish themselves (Freidmann et al. 1996) and the health of piscivorous birds that feed on contaminated fish from the lake. Our PSR chain for mercury includes these sources and focuses on fish as the primary route of human exposure to the neurotoxin (Figure 26).

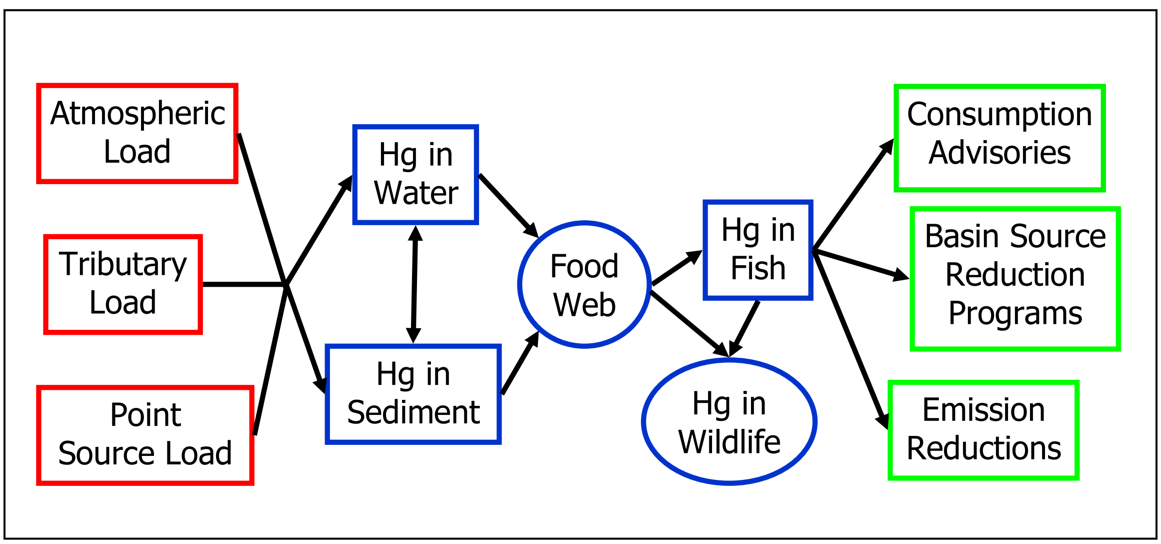



**Figure 26: Pressure-state-response diagram for mercury contamination in Lake Champlain.**

**Table 13: Indicators for mercury contamination in Lake Champlain.**

| Indicator | P S R | Available Measure | Year | Source | Ref | Recommended Measure | Frequency (minimum) |
|---|---|---|---|---|---|---|---|
| Atmospheric load | P | Mean Hg load from atmospheric deposition | 1994-1996 | NOAA | Page 61 | Annual mean Hg load from atmospheric deposition | Updated every 5 years |
| Tributary load | P | Discharge and Hg concentrations | 2000-2002 | USGS | Table 14 | Annual mean Hg load by lake segment | Updated every 5 years |
| Point Discharge | P | Estimated Hg load from point sources | NA | SLU | Figure 27 | Measured Hg load from point sources by lake segment | Updated every 5 years |
| Hg in Water | S | Total Hg concentration in water column by lake segment | 2001 | USGS | Figure 28 | Total and methyl Hg concentration in water column by lake segment | Updated every 5 years |
| Hg in Sediment | S | Total Hg concentration in sediment by lake segment | 1991 | UVM | Figure 29 | Total and methyl Hg concentration in sediment by lake segment | Updated every 10 years |
| | | | | | | Number of lake segments above threshold effects level | Updated every 10 years |
| Food web | S | Mean Hg concentration in plankton | 1997 | UVM | Page 64 | Mean Hg concentration in key lower trophic level species in selected lake segments | Updated every 5 years |
| Hg in Fish | S | Mean Hg concentration in walleye and yellow perch | 1988-2000 | VT DEC | Figure 30 | Body burden in key species by weight class | Updated every 5 years |
| Hg in Piscivorous Wildlife | S | Mean Hg concentration in selected wildlife species | | | | Body burden in selected wildlife species | |
| Consumption advisories | R | NY and VT fish consumption advisories | | VT DOH, NY DOH | Page 66 | Number of species for which advisories exist | Updated every 5 years |
| Basin source reduction programs | R | Dollars spent on source reduction | | LCBP | Page 68 | Hg load reduction achieved | Updated every 2 years |
| Emission reductions | R | | | | Page 68 | Hg load reduction achieved | Updated every 5 years |

## PRESSURE INDICATORS

Atmospheric loading is thought to be the principal source of mercury in the Lake Champlain Basin, with slightly more than 50% coming from outside the region and the remainder generated by local emissions (Fitzgerald et al. 1998, Shanley et al. 1999). Mercury that is released into the atmosphere by coal fired power plants and other industrial emissions is deposited onto the watershed and directly into the lake. Dry and wet mercury deposition data have been collected at the Proctor Maple Research Center (PMRC) on Mount Mansfield in Underhill, Vermont for over ten years. At PMRC, the average deposition rate over a two year period beginning in 1994 was 444 mg/ha/yr, with approximately 70% in the form of dry deposition and the remainder in precipitation (Shanley et al. 1999). PMRC is a forested area and the greater dry deposition rate is attributed to the additional surface area created by foliage. The rates and relative importance of wet and dry deposition on the 36% of the basin that is not forested is currently under investigation (Shanley, personal communication).

A portion of the mercury that is deposited on the watershed is washed into surface runoff and transported to the tributaries and into the lake. Based on studies conducted in the Nettle Brook catchment in Underhill, Vermont, approximately one third of the mercury flux into Lake Champlain from the tributaries is from a consistent but low concentration of dissolved mercury in the baseflow. The remaining two thirds is primarily particulate mercury exported during periodic high flow events (Shanley et al. 1999). Although mercury sampling was conducted in sixteen Lake Champlain tributaries in 2000, 2001 and 2002, the flow conditions sampled varied by year, making annual load comparisons difficult (Table 14). Studies conducted in the forested Nettle Brooke catchment show that the majority of the mercury load transported to the lake by the tributaries is associated with sediment and organic matter. Therefore, runoff from agricultural and developed land, which can contain high sediment loads, may be important sources of mercury loading that have not yet been fully examined (Shanley et al. 1999). The Nettle Brook studies have also demonstrated a high watershed mercury retention rate (92-95%) (Scherbatskoy et al. 1998). This suggests that low level mercury inputs to the lake can be expected to continue over the long term even if mercury inputs to the watershed decline.

**Table 14: Examples of mercury sampling data from Lake Champlain tributaries from 2000-2002.**

| | Year | Flow Condition | Discharge (cfs) | Total Hg (ng/L) |
|---|---|---|---|---|
| **Ausable River** | 2000 | Baseflow | 216 | 1.54 |
| | 2001 | low flow | 154 | 0.82 |
| | 2002 | snow melt | 1193-6210 | 2.57-4.60 |
| **Otter Creek** | 2000 | Baseflow | 272 | 2.53 |
| | 2001 | low flow | 238 | 0.83 |
| | 2002 | snow melt | 2271-2910 | 3.25-7.79 |
| | 2002 | storm events | 573-1587 | 1.08-2.03 |

Industrial and wastewater discharges are also sources of mercury. The concentrations of mercury in outflow of wastewater treatment facilities in the Champlain Basin are generally below the detection limit of the locally available methods and equipment (N. Kamman, VT DEC, personal communication). However, the mercury load from wastewater treatment facilities has been estimated based on discharges measured for the Lake Champlain Diagnostic Feasibility Study (VT DEC and NY DEC 1994) and typical mercury concentrations for wastewater treatment facilities elsewhere in the United States (N. Gao, St. Lawrence University, personal communication) (Figure 27). The estimated mercury load into South Lake A is substantially greater than the load into other lake segments. This is likely the result of the high discharges from the International Paper facility.

**STATE INDICATORS**

The mercury concentration in the water column in each lake segment was sampled in September of 2001 by the USGS (Shanley, personal communication). With the exception of the South Lake segments, mercury is relatively uniformly distributed throughout Lake Champlain (Figure 28). The higher concentrations in South Lake A and B may reflect inputs from the International Paper Company point source discharge (Shanley et al. 1999).

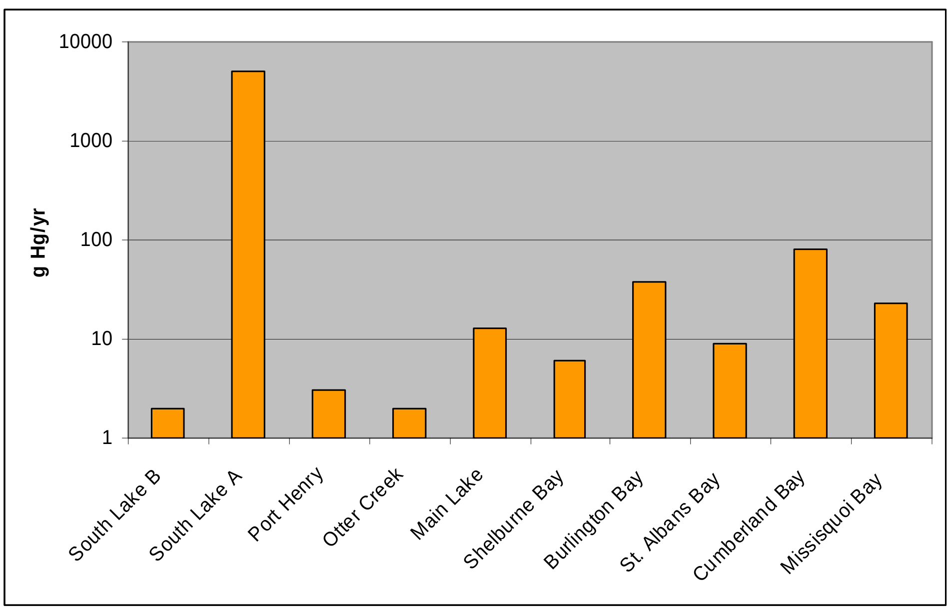



**Figure 27: Estimated annual total mercury loads from wastewater inputs by lake segment.**

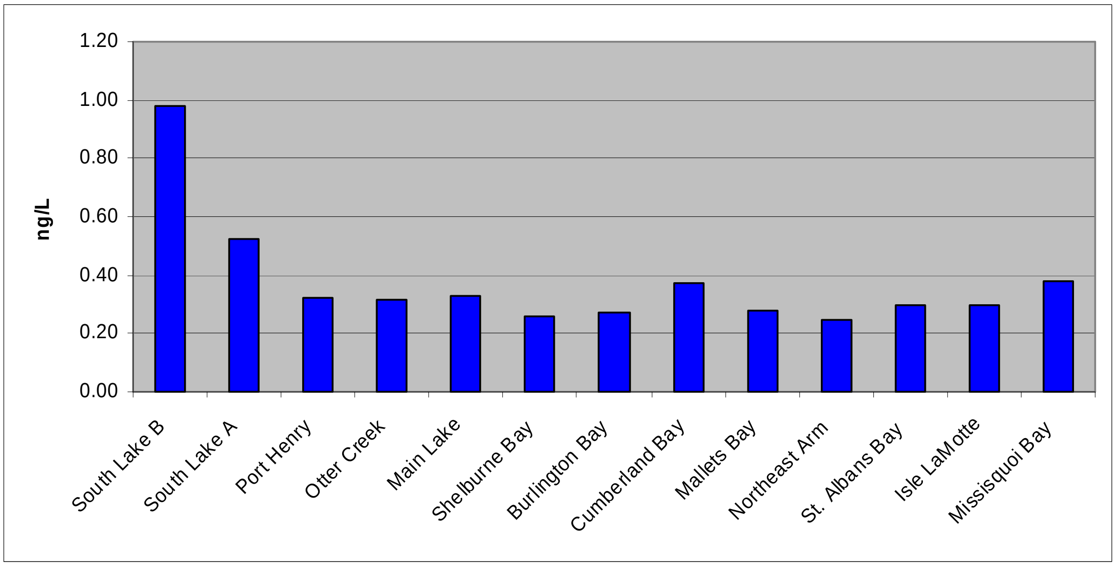



**Figure 28: Total mercury concentrations from single water column samples collected from Lake Champlain segments in September 2001.**

Because much of the mercury transported to Lake Champlain by the tributaries is associated with particulate matter that eventually settles to the bottom of the lake, mercury contaminated lake sediments may be an important source of mercury to the food web. Sediment concentration over a depth profile can also serve as an indicator of the historic load of mercury to the lake. In 1991, surface sediments from thirty sites across Lake Champlain were collected and analyzed for mercury and other toxic pollutants (McIntosh 1994). Some mercury contamination was found in the sediments at all the sample sites (Figure 29). The low sediment concentrations in the South Lake seems inconsistent with the high water concentrations, and suggests that the dissolved and suspended mercury in transported northward into the Main Lake.

Although there are no sediment quality standards specifically for Lake Champlain, sediment guidelines have been suggested by NOAA for freshwater sediments (Buchman 1999). For mercury, a threshold effects level (TEL) is suggested at 0.174 μg/g and an upper effects level at 0.560 μg/g. Using these levels the Main Lake, Burlington Bay, Cumberland Bay, St. Albans Bay, Isle LaMotte, and Missisquoi Bay could all have low-level toxic effects as result of mercury contamination. Background is estimated at 0.004-0.051 μg/g; all segments of Lake Champlain exceed background levels of mercury.

Total mercury is measured to gauge how much mercury is in the environment, but from a biological standpoint, it is the amount of methylmercury (Me-Hg) that is most important. Ionic mercury is transformed into Me-Hg by microbial and abiotic processes (Shanley, et al. 1999). Me-Hg, which is generally only 1-10% of the total mercury in the water column, is the form of mercury that bioaccumulates, making up more than 95% of the mercury found in fish tissue (Freidmann et al. 1996, Shanley et al. 1999).

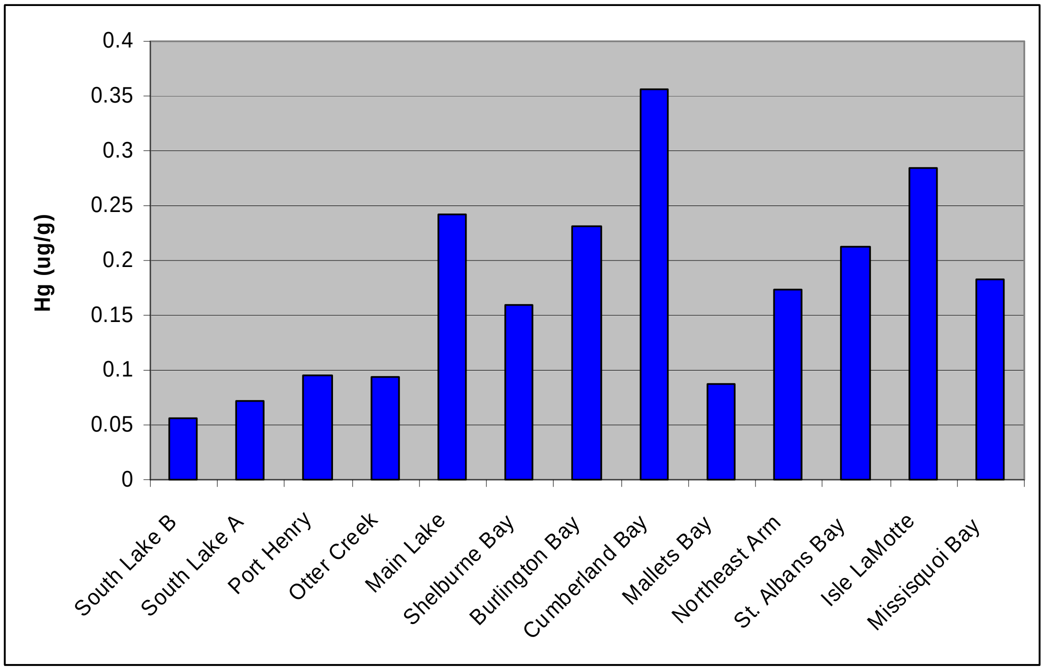



**Figure 29: Mean total mercury concentration in lake sediments (µg/g) from five replicate samples collected in June 1991. Data from the stations closest to the 13 long-term water quality monitoring stations are presented.**

Predicting bioaccumulation rates in lakes is not simple because both methylation rates and bioavailability of mercury is affected by numerous factors (Driscoll et al. 1994a, b, Kamman et al. 2003). The structure of the food web and productivity of the lake are particularly important factors. In northeastern lakes, Chen et al. (2000) showed that the total mercury in fish tissue is inversely correlated to both zooplankton food chain length and to cladoceran (a common component of the zooplankton) density. In highly eutrophic lakes, the mercury can be "diluted" by the phytoplankton biomass, resulting in lower mercury bioaccumulation in higher trophic levels. In experimental manipulations, Pickhardt et al. (2002) showed that in waters of equal mercury concentration, mercury accumulation in cladocerans is inversely proportional to phytoplankton density. This suggests that the fish in eutrophic sections of Lake Champlain like the South Lake and Missisquoi Bay may accumulate less mercury than the fish in mesotrophic or oligotrophic lake segments like the Main Lake.

To calculate bioaccumulation factors, data on the concentration of mercury in various levels of the food chain are needed. In 1997, dry weight mercury concentrations were determined for three samples of small (63-202 µm) and large (>202 µm) plankton collected by UVM from Lake Champlain. The average mercury concentration was 410 $\pm$ 110 ng/g in the small plankton and 760 $\pm$ 20 ng/g in the large plankton (Shanley et al. 1999). No data are available for other links in the food chain, however, in Lake Champlain, one of the major forage fish species for the sport fish is rainbow smelt (*Osmerus mordax*). This fish has been linked to increased mercury in native predatory fish in northwestern Ontario, where it has invaded in the last two decades, but the mechanisms of this increase are not known (Swanson et al. 2003).

Organisms that are largest and/or feed highest on the food chain are generally expected to have the highest concentrations of Me-Hg. The VT DEC analyzed 225 fish of sixteen different species for the concentration of mercury and other toxins in their tissues. These fish were collected from ten different lake segments over a 25-year period from 1975 to 2000. However, because the species, location, sample size and fish vary widely from year to year, it is difficult to determine trends in mercury contamination in the Lake Champlain fish community. In Lake Champlain, mercury concentrations are generally highest in walleye, a piscivorous top predator (Figure 30). For walleye, yellow perch and other species, the mercury concentration tends to correspond to the size of the fish sampled; bigger fish are older fish, and therefore, have bioaccumulated more mercury. Yellow perch has been used as a survey fish by VT DEC because it is such a broadly distributed fish. However, in order to track trends in mercury in fish tissue over time, samples must be repeatedly collected of fish of the same species and the same size.

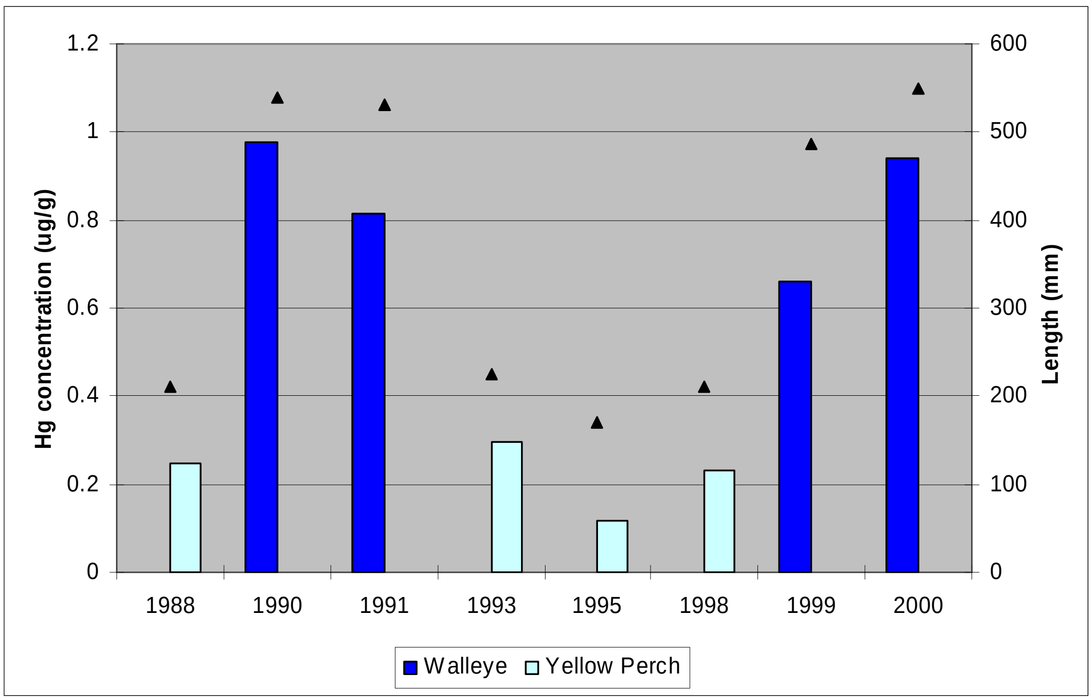



**Figure 30: Mean total mercury concentration (µg/g) in walleye and yellow perch (bars) with average length (mm) of the fish in the sample (triangles). Samples were collected from different lake segments in different years and the sample size for a given year varies from 3 to 20 individuals for walleye and 4 to 9 individuals for yellow perch.**

Mean total mercury concentrations for Lake Champlain walleye are approaching the Food and Drug Administration (FDA) limit for human consumption of 1 ppm (μg/g). This action level is set to provide an adequate margin of safety for fish consumption, and considers the types of fish people commonly eat, levels of methyl mercury found in each species, and average consumption amounts. It is designed to limit consumers' exposure to levels 10 times below the lowest methyl mercury level associated with adverse health effects. If fish tissue is found to exceed this action level, the FDA recommends that state health departments issue local or water-body specific advisories.

US EPA (2004) has compiled a database of tissue mercury concentrations in noncommercial fish as reported by states and tribes across the country. Mean concentrations for each species are reported as the arithmetic mean of the means from each sampling station, for species with at least 100 stations reporting. Fillet samples for adult fish of all lengths and weights are included in the database. These national mercury concentration data may be useful as a reference or comparison with Lake Champlain data. Mercury concentrations in yellow perch collected from 1987 – 2003, with 604 sampling stations reporting nationwide, averaged 0.22 μg/g. Yellow perch in Lake Champlain have mercury concentrations that are close to this national average (Figure 30). The US EPA reports that walleye tissues samples from 1,520 stations for the same time period averaged 0.4 μg/g. Annual mean concentrations in walleye from Lake Champlain range from 0.6 – 0.9 μg/g for the four years they were sampled from 1990-2000. Although these data are difficult to put in context because no size information is available in the US EPA data set, they suggest that Lake Champlain fish have mercury body burdens that are higher than those typically found throughout the nation.

The US EPA also published a human health water quality criterion for methylmercury of 0.3 μg/g fish tissue in 2001 (US EPA 2001). Water quality criteria are usually expressed as concentrations measured in the water column; but because the main pathway of human exposure is through consuming contaminated fish tissue, EPA issued this fish tissue value along with preliminary guidance for states on how to translate it into a mercury concentration in ambient surface water or effluent using bioaccumulation factors. States are expected to adopt this or a more stringent value as a water quality standard for mercury, and use it to control discharges through permit limits or to develop mercury TMDLs. It is not clear how this is being addressed in either Vermont or New York because the information necessary to calculate bioaccumulation factors is not available for Lake Champlain.

## RESPONSE INDICATORS

In order to protect human health, the management response to elevated levels of mercury in fish is the issuance of fish consumption advisories. Such advisories are issued by the federal government and state agencies, and they currently exist in both Vermont and New York for walleye, lake trout, and selected other fish.

Traditionally, the US FDA has issued advisories for fish and shellfish sold commercially and imported into the US, while US EPA issues advisories on recreationally caught fish. In March of 2004, the US FDA and US EPA issued a joint consumer advisory on methyl mercury in fish.

This revised advisory was intended to reduce mercury exposure to the most at-risk population – fetuses and young children. Women who may become pregnant, who are pregnant, and nursing mothers should not eat shark, swordfish, king mackerel or tilefish and eat up to 12 ounces (2 meals) per week of fish and shellfish that are lower in mercury (shrimp, canned light tuna, salmon, pollock and catfish). Albacore tuna has more mercury than canned light tuna. One meal per week of albacore tuna can be substituted for the 2 meals per week choice above. This advice also applies to young children, but they should be served smaller portions (FDA 2004). Consumers are advised to follow state advisories for fish caught locally. New York advises that women of childbearing age and children under 15 should not eat any fish from Lake Champlain (New York State Department of Health 2004). Vermont has advisories that are specific to particular fish species (Vermont State Department of Health 2000). New York and Vermont both define a meal as 8 ounces of fish, while the FDA and US EPA define an average meal as 6 ounces. Table 15 summaries the fish consumption advisories for Lake Champlain in the two states.

**Table 15: Fish consumption advisories for Lake Champlain (summarized from VT DOH 2000, NY DOH 2004).**

| | **New York** | | **Vermont** | |
|---|---|---|---|---|
| **Fish species** | Women of childbearing age and children under 15 | All other individuals | Women of childbearing age and children under 6 | All other individuals |
| Walleye | 0 meals | No more than 1 meal/week* | 0 meals | No more than 1 meal/month |
| Walleye > 19 inches | 0 meals | No more than 1 meal/month | | |
| Lake Trout >25 inches | 0 meals | No more than 1 meal/month | 0 meals (includes children under 15) | No more than 1 meal/month |
| Lake Trout ≤25 inches | 0 meals | No more than 1 meal/week* | 0 meals (includes children under 15) | No more than 3 meals/month (?) |
| Smallmouth Bass<br>Chain Pickerel<br>American Eel | 0 meals | No more than 1 meal/week *<br>(1 meal/month for eel from Cumberland Bay) | No more than 1 meal/month | No more than 3 meals/month |
| Largemouth Bass<br>Northern Pike | 0 meals | No more than 1 meal/week* | No more than 2 meals/month | No more than 6 meals/month |
| Other trout<br>Yellow Perch | 0 meals | No more than 1 meal/week *<br>(1 meal/month for yellow perch from Cumberland Bay) | Mo more than 3-4 meals/month | No advisory |
| Brown Bullhead | 0 meals | No more than 1 meal/week* (0 meals for brown bullhead from Cumberland Bay) | No advisory | No advisory |

| | | | | |
|---|---|---|---|---|
| Pumpkinseed | 0 meals | No more than 1 meal/week* | No advisory | No advisory |
| All other fish | 0 meals | No more than 1 meal/week* | No more than 2-3 meals/month | No more than 9 meals/month |

*NY State has a default advisory for individuals to eat no more than one meal per week of freshwater fish.

There are several local, regional and state-sponsored efforts to reduce sources of mercury to Lake Champlain. Chittenden Solid Waste District has a public education campaign to inform residents about mercury in common household products, and how to identify and purchase low- or mercury-free alternatives (www.cswd.net/hazardous waste/). Hazardous waste drop-off centers throughout the watershed accept mercury-bearing items such as batteries, fluorescent light bulbs, paints and thermometers. Vermont DEC has estimated the amounts of mercury removed from the waste stream through municipal household hazardous waste programs since 2000, one potential measure of recycling program success. With funding from the Lake Champlain Basin Program, the Northwest Vermont Solid Waste Management District is working with the Vermont Department of Agriculture to replace mercury-containing manometers on Vermont's dairy farms. As of October 2000, half of the known manometers in the basin were removed, each of which contained up to 1/2 pound of mercury (LCBP, personal communication). The National Wildlife Federation worked with dental clinics in the watershed to encourage proper mercury disposal practices, also with LCBP funding (LCBP, personal communication).

In 1998 the Vermont Legislature established the Advisory Committee on Mercury Pollution to work with Vermont DEC and Vermont Department of Health (VDH) to improve mercury reduction efforts. Non-regulatory programs include outreach on fish consumption advisories, elementary and middle school education programs, thermostat and fluorescent light bulb recycling outreach, and hospital and dental clinic mercury reduction programs (Vermont Advisory Committee on Mercury Pollution 2004). Regulatory efforts include Vermont's 1998 product labeling law, landfill disposal restrictions for labeled mercury-added consumer products, and pending legislation that requires comprehensive management of mercury in Vermont, including banning the sale of certain mercury-added products (Vermont Advisory Committee on Mercury Pollution 2004).

In 1998, the New England Governors and Eastern Canadian Premiers adopted a regional mercury reduction policy with the goals of reducing emissions by 50% in 2003, 75% in 2010, and to completely eliminate emissions thereafter. Reductions in mercury emissions from municipal waste combustion and industrial point sources in the region have surpassed the 50% goal (Vermont Advisory Committee on Mercury Pollution 2004). Coal-fired and other power plants remain the largest source of atmospheric mercury, with out-of-region sources accounting for one-third of mercury deposition in the region (Vermont Advisory Committee on Mercury Pollution 2004).

In January 2004, U.S. EPA released its first proposed rule to regulate mercury emissions from coal-fired power plants nationwide for public comment (U.S. EPA 2004). The rule caps power plant emissions, either through implementing maximum achievable control technology, or through a market-based "cap and trade" program. The EPA anticipates issuing a final rule in

March 2005. When fully implemented, EPA estimates that emissions from utilities would be reduced by 30-70 percent, depending on the regulatory approach selected.

Within the Lake Champlain Basin, the atmospheric mercury load to the watershed is greater than the load delivered to the lake in the surface water (Shanley et al. 1999). That means that some mercury is being stored in various compartments of the watershed, including the soils. Consequently, management actions aimed at reducing soil erosion and sediment transport could help reduce the mercury load to the lake, in addition to the phosphorus and bacteria load. Currently, efforts to reduce soil erosion are being implemented in agricultural areas where phosphorus concentrations are high, and through stormwater controls. Indicators for these management responses were previously discussed in the phosphorus issue area. Lake Champlain forests tend to accumulate more atmospheric mercury than other landscapes because the forest canopy provides more surface area for mercury deposition per unit of land area (Shanley et al. 1999), therefore, efforts to protect these land areas will also help prevent additional mobilization of mercury to the lake. Preliminary data from urban watersheds suggests that on a unit area basis, these watersheds may be among the largest sources of mercury to the lake (Shanley, personal communication). Efforts at stormwater control and management should help in this land use type.

## ASSESSMENT OF ADEQUACY AND ADDITIONAL MEASURES

Although the effort in this area has increased substantially in the last few years as a result of the funding coming to the US Geological Survey (USGS), key data are still lacking for many indicators. Atmospheric deposition was monitored consistently in the early 1990s using funding provided through NOAA, creating one of the best data sets available in the world for the PMRC site on Mount Mansfield (Shanley et al. 1999). However, this data collection has been less complete in the last several years, and the data are not readily available to the broader user community.

The surface water data that are currently being collected by the USGS should allow us to estimate tributary loads for most lake segments. Although it would be best to track these loads annually, in order to begin to develop an understanding of natural variability, an update every 5 years to track trends is a suggested minimum given the costs associated with this work. As analytical techniques allow, mercury concentrations in point sources should be measured and the point source loads re-estimated on a periodic basis.

We have proposed five state indicators that would help us track mercury in the major compartments in the lake. This will take a considerable new investment because currently we have incomplete data for all these indicators. Measures of the mercury concentration in water and sediment are necessary to begin to develop a mass balance for the lake and to develop a bioaccumulation factor for the food web. Mercury concentration in the sediment will change slowly, but the current data are now more than 10 years out of data, and new data are needed. We have only four data points, from samples collected by UVM in 1997, to indicate the concentration of mercury in the food web. A focused data collection effort in this area is critical.

Although both states collect fish samples for analysis of the mercury concentration in tissues, the sampling design currently will not allow analysis for trends over time. We recommend that both states agree on one or two species, and a size category, and that data in the future be collected according to this agreement. Walleye should be one those species.

Other states are also working to gather data on mercury concentrations in wildlife. For smaller lakes, loons can be good indicators of the level of mercury contamination in the watershed (Vermont Advisory Committee on Mercury Pollution 2004); however, loons do not use Lake Champlain as nesting habitat, so this species will not work for this large lake.

The response indicators we recommend include the number of species for which consumption advisories are posted, and the estimated load reductions that might be achieved by the LCBP partner efforts in the watershed, and national or continental efforts to reduce mercury in emissions. Data in all these areas are needed.

## D. Issue Area: Sport Fish Community

A healthy fish community in Lake Champlain is important from a recreational and ecological perspective. Although several species of sport fish have been stocked and otherwise managed for decades, there are many factors influencing these fish populations, not all of which are well understood by scientists and managers. Our PSR diagram takes a trophic interaction approach to the sport fish community (Figure 31). Because the effects of the parasitic sea lamprey have been dramatic, much of the fishery management effort in the basin has focused on this nuisance species, but other management activities probably also influence the sport fish through the food web interactions.

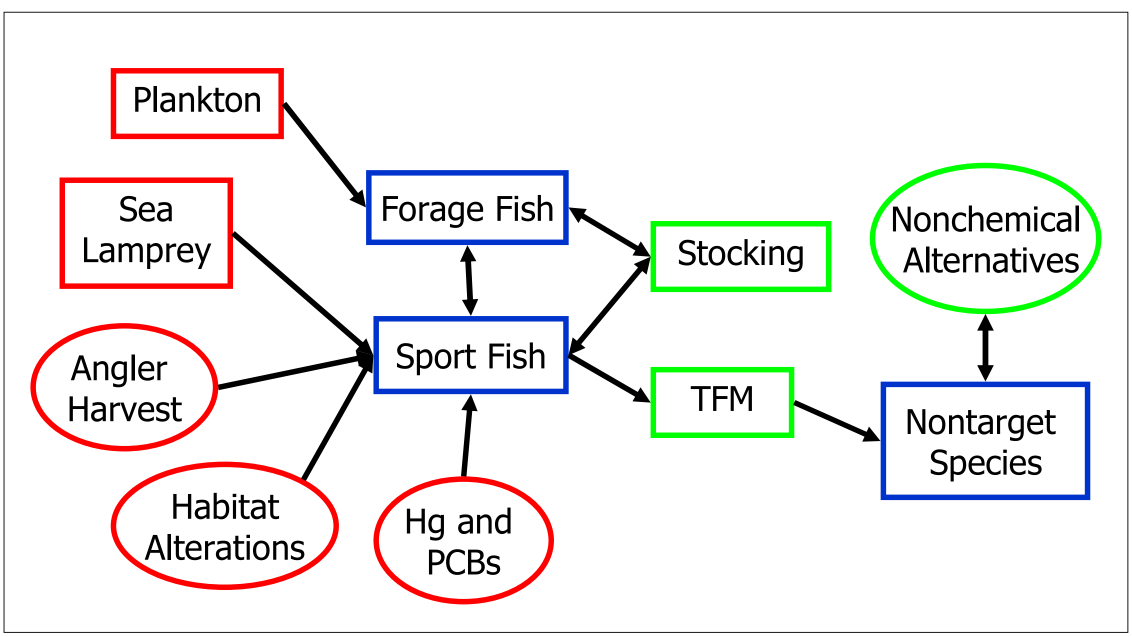



**Figure 31: PSR diagram for the sport fish community.**

## Table 16: Indicators for a healthy sport fish community.

| Indicator | P S R | Available Measure | Year | Source | Ref | Recommended Measure | Frequency (minimum) |
|---|---|---|---|---|---|---|---|
| Sea Lamprey | P | Mean number of wounds per 100 lake trout | 1982-2002 | LCFWMC; VT FW | Figure 32 | Mean number of wounds per 100 lake trout | Annually |
| Habitat Alterations | P | Data unavailable | | | Page 72 | Develop index of habitat quality for spawning areas | Update every 10 years |
| Angler harvest | P | Data unavailable | | | Page 72 | Creel surveys by selected lake segments | Annually |
| Hg and PCBs | P | Mean Hg concentration in walleye and yellow perch | 1988-2000 | VT DEC | Figure 30 | Body burden in key species by weight class | Updated every 5 years |
| | | Gonadosomatic index in juvenile walleye | 1995 | UVM | Page 72 | Gonadosomatic index for selected species | Updated every 5 years |
| Sport Fish | S | Mean number of lake trout per gill net lift | 1982-1997 | LCFWMC | Page 73 | Annual lake trout population abundance by lake segment | Annually |
| Forage Fish | S | Mean rainbow smelt catch per trawl | 1987-2002 | LCFWMC | Figure 33 | Mean rainbow smelt catch per trawl | Annually |
| | | Rainbow smelt mean length | 1984-2002 | LCFWMC | Figure 34 | Rainbow smelt mean length | Annually |
| Plankton and biodiversity | S | Phytoplankton and zooplankton taxonomic composition and relative abundance | 1991-2002, with some missing dates | LCBP long-term bio-monitoring; SUNY-Plattsburgh | Page 75 | Biomass and size distribution of zooplankton | Annually |
| | | Number of exotic species in the lake (fish and plankton) | 2000 | LCBP – ANS plan | Page 75 | Percent abundance of exotic species by taxa | Update every 2 years |
| Nontarget species | S | Abundance of selected taxa before and after TFM application | 1990-1995 | LCFWMC | Figure 37,38 | Abundance of selected taxa before and after TFM application | Annually |
| Stocking | R | Hatchery released smolt equivalents by lake segment | 1972-2002 | LCFWMC | Figure 35 | Hatchery released smolt equivalents by lake segment | Annually |
| TFM | R | Miles of stream exposed to TFM | 1990-2000 | LCFWMC | Figure 36 | Miles of stream exposed to TFM | Annually |
| Nontarget species | S | Abundance of selected taxa before and after TFM application | 1990-1995 | LCFWMC | Figure 37, 38 | Abundance of selected taxa before and after TFM application | Annually |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| Nonchemical alternatives | R | Data unavailable | | LCFWMC | Page 79 | Stream miles treated by nonchemical alternatives | Annually |

**PRESSURE INDICATORS**

The population of sea lamprey, a parasitic, eel-like fish, is probably the most important pressure on the sport fish in Lake Champlain. Sea lamprey attach to and prey upon salmonids, such as lake trout, landlocked Atlantic salmon, and other softer scaled fish in the lake causing unsightly sores, reduced growth, and mortality (Fish and Wildlife Management Cooperative Fisheries Technical Committee 1999). The impact of sea lamprey on this top level of the Lake Champlain food web has been monitored since the 1980s. An experimental lamprey control study conducted from 1990 through 1997 provided the basis for the current sea lamprey management strategy. The experimental study demonstrated that the sea lamprey population can be reduced with the application of lampricides in tributaries and river deltas (Fish and Wildlife Management Cooperative Fisheries Technical Committee 1999).

The impact of sea lamprey on Lake Champlain sport fish is commonly measured in terms of wounding rates. Although the wounding rate varies from species to species (species with softer scales are more susceptible) and from lake segment to lake segment (sea lamprey are more prevalent in deeper, colder lake segments), sea lamprey population reductions should result in decreases in wounding rates. The average wounding rate on lake trout is used to indicate the pressure on sport fish from sea lamprey (Figure 32).

Sport fish populations in Lake Champlain are also susceptible to alteration and degradation of fish habitat both in the lake and in the tributaries. Different species require different spawning grounds and nursery habitat. Indicators of both the quantity and the quality of habitat available to various fish populations in Lake Champlain and its tributaries are necessary. However, at this time, there are no data available to assess either the current condition of fish habitat or trends in the amount of habitat available. In the future, appropriate indicators might include hectares of “good” habitat available in the lake, and miles of stream habitat available in the tributaries.

Angler harvest is another pressure on Lake Champlain sport fish populations. Although creel survey data has been collected intermittently across the basin, we were unable to obtain any data on angler harvest for inclusion in this report. Because recreational fishing on Lake Champlain is both economically and ecologically important, the impact of harvesting on sport fish populations should be quantified and incorporated into our understanding of sport fish population dynamics.

Toxins in the lake also have the potential to influence sport fish population. As discussed above, it is difficult to determine the status and trends of fish toxicity in Lake Champlain because of the irregularity of the sampling regime. However, a number of recent studies have shown reproductive impairments when fish are exposed to methylmercury (Hammerschmidt et al. 2002, Drevnick and Sandheinrich 2003) or PCBs (Gutjahr-Gobell et al. 1999, Matta et al. 2001) in their diet at fairly low concentrations. A laboratory study conducted using juvenile walleye hatched from Lake Champlain parents has shown that mercury can impair the development of the reproductive system of 6-12 month old fish (Freidmann et al. 1996). Over the six month experimental period, both male and female fish fed a mercury-contaminated diet grew more slowly and showed lower gonad condition (measure using the gonadosomatic index) than those on an uncontaminated diet. Male fish showed atrophied testes, especially in the higher mercury treatment.

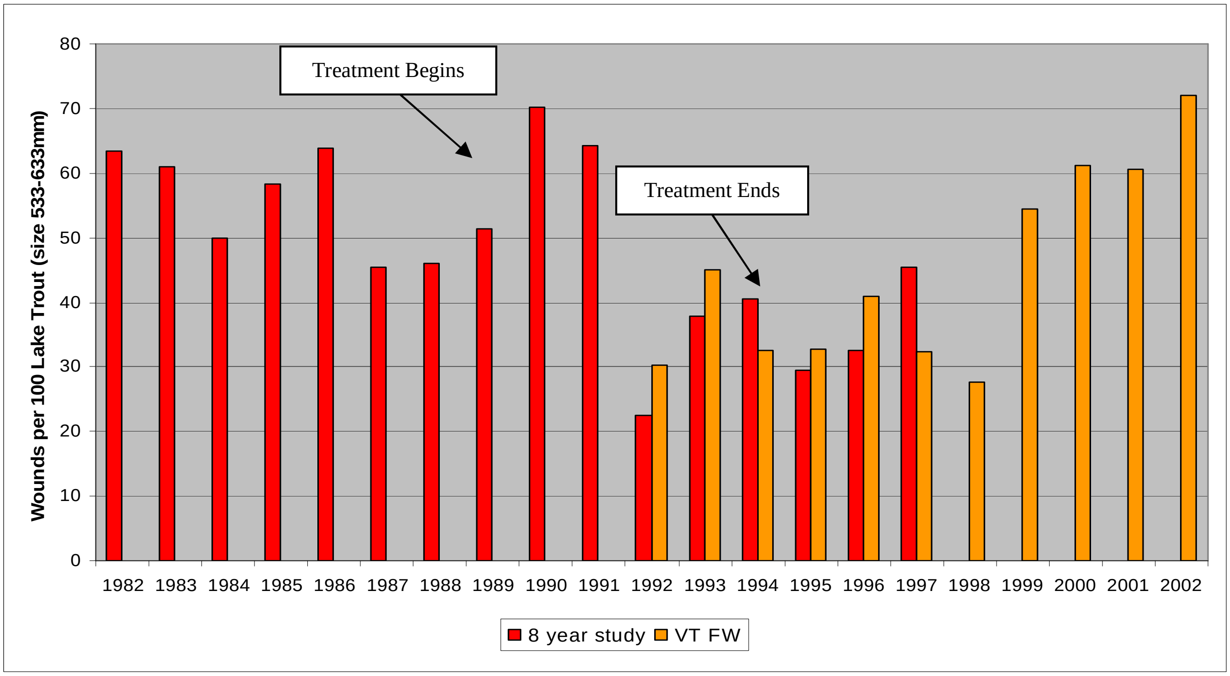



**Figure 32: Average number of wounds per 100 lake trout (533-633 mm). Lake trout wounding rates were calculated from two different datasets: data from the eight year sea lamprey control study and data from the VT Department of Fish and Wildlife. Treatment refers to the period where experimental lampricide treatments were applied.**

## STATE INDICATORS

Although lake trout population assessments are regularly conducted in the Main Lake and Inland Sea using gill nets, we were unable to obtain these data or the estimates of lake trout abundance that they produced. Over the monitoring period included in the eight year experimental sea lamprey control study (1982-1997), a significant increase in the average number of lake trout caught per net lift was reported for areas outside fishery management zones 3A and 3B, the central and southern portions of the Main Lake (Fish and Wildlife Management Cooperative Fisheries Technical Committee 1999).

Rainbow smelt, the primary forage fish for salmonid species in Lake Champlain, have been monitored continuously in Lake Champlain since 1984 using a midwater trawling technique developed by Kirn and Labar (1991). In 2001 and 2002, mid-water trawling was supplemented by a hydroacoustic survey method to monitor rainbow smelt in three sections of Lake Champlain (Parrish et al. 2004). At four of the five stations for which there are long-term data, catch per unit efforts (CPUEs) were higher in 2002 than in previous years. Over time, CPUEs from Juniper Island and the other long-term stations show a cyclical pattern (Figure 33) (N. Staats, U.S. Fish and Wildlife Service, Essex Junction, Vermont, unpublished data). Mean lengths of rainbow smelt have remained relatively constant (Figure 34).

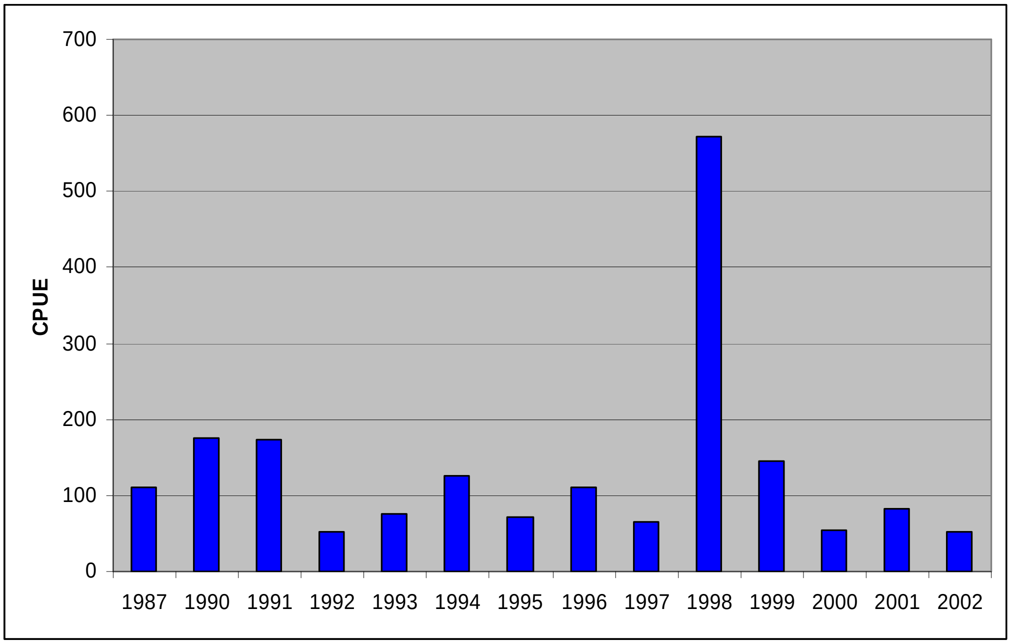



**Figure 33: Rainbow smelt mean catch per 55 minute trawl of rainbow smelt at the Juniper Island station.**

Smelt dynamics are the result of a complex set of density-dependent, density-independent, and environmental factors. Parrish et al. (2004) are currently working to refine smelt population models for the Main Lake, Inland Sea, and Malletts Bay that consider recruitment, cannibalism, and predation. Estimates of cannibalism, as well as zooplankton consumption are also being made based on diet analysis. Parrish et al. have concluded that smelt are highly cannibalistic, especially when the young-of-year density is high and when the thermal structure of the lake allows spatial overlap of the young-of-year with older fish. Smelt population cycles may be driven by this cannibalism. When smelt abundances are high, they may be able to limit the abundance of zooplankton, as least in late summer. This, in turn can cause individual smelt to grow more slowly.

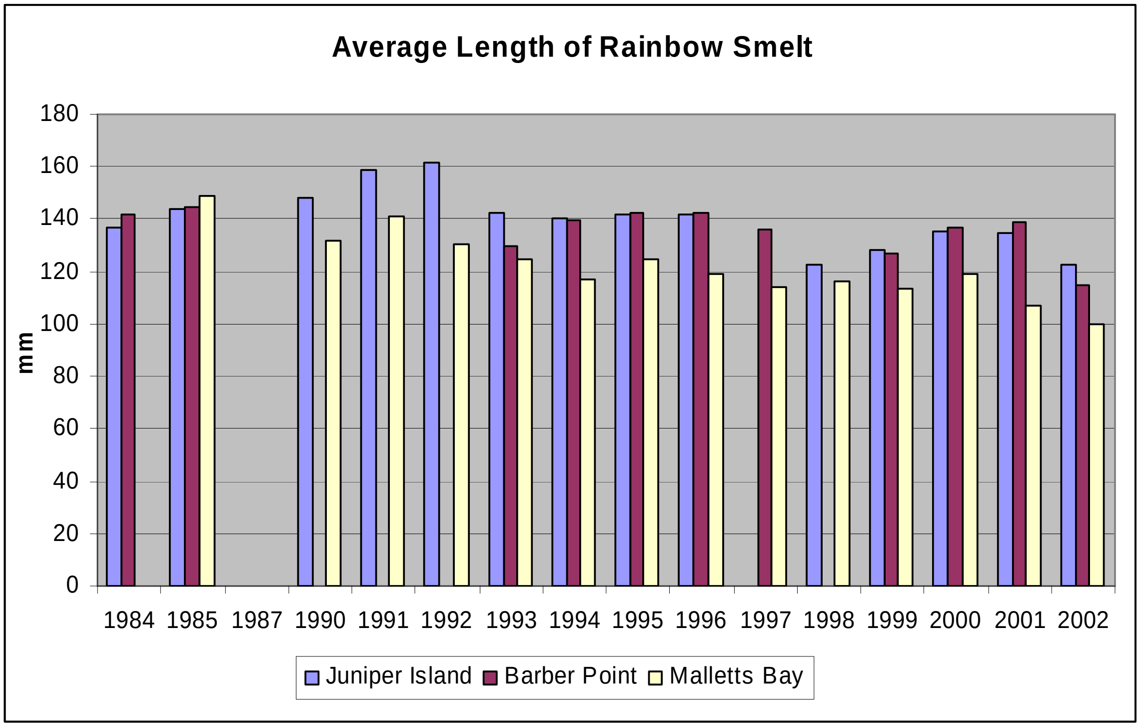



**Figure 34: Rainbow smelt lengths measured from 1984 through 2002 at three stations in Lake Champlain.**

Although Parrish et al. (2004) were able to develop density estimates for the smelt population in the Main Lake, Inland Sea and Malletts Bay using a combination of hydroacoustics and trawl surveys, there were wide confidence intervals around these estimates, suggesting that detecting trends in the data over time will be difficult. Additional work in this area will be needed to refine the model and improve its predictive ability.

Plankton is included as a state indicator to capture the bottom up controls of productivity (Schindler 1977, Wetzel 1983). Because phosphorus is featured in its own issue area, including links to the biomass of algae, we suggest using measures of the zooplankton community to represent this indicator in this issue area. Currently, plankton samples are collected by the LCBP Long-term Biomonitoring Program and analyses are underway at SUNY-Plattsburgh. Currently, the focus of this effort is on the taxonomic composition of the zooplankton, but biomass and size structure may say more about the grazing pressures in the lake, the allocation of phosphorus among the food web components, and the resilience of lake ecosystems in the face of phosphorus enrichment (Kitchell and Carpenter 1993, Carpenter et al. 1996). In the pelagic food web issue area, we discuss these linkages further.

The introduction of exotic species has accelerated in Lake Champlain and other aquatic ecosystems despite increasing awareness of the risks these species can pose for ecosystem (Lodge 1993, Mills et al. 1994). Once in the lake, these species can proliferate and cause a cascade of changes in the natural ecosystem. Like zebra mussels, white perch (*Morone* *americana*) is a relatively recent invader that is believed to have entered the lake through the Champlain Barge canal (Hawes and Parrish 2003). It was first reported in 1984 (Plosila and Nashett 1990) and has rapidly spread northward. It is a planktivore that may displace populations of native yellow perch and become a predominant forage fish for higher trophic levels. The implications of this for the sport fish community are not clear. Likewise, other potential invaders such as alewife (*Alosa pseudoharengus*), round goby (*Neogobius melanostomus*), and gizzard shad (*Dorosaoma cepedianum*), all of which are already found in or near the Lake Champlain Basin, could cause dramatic changes in the ecosystem. An indicator that tracks the relative abundance of exotic species in both the plankton and the fish community would help understand these changes and their implications for the sport fish populations in the lake. Ideally, data might be summarized by major taxonomic group.

## RESPONSE INDICATORS

Currently, there are two major management responses to the state of sport fish in Lake Champlain: stocking and sea lamprey control. Several species of sport fish, including lake trout, land-locked Atlantic salmon, walleye, steelhead trout and brown trout are grown in hatcheries across the basin and released into a wide range of tributaries and lake segments. These species are stocked in order to maintain sport fish populations at levels that will sustain the recreational fisheries in the basin, and because there does not appear to be sufficient natural reproduction to maintain the salmonids in the lake without assistance. A stocking database for lake trout and other salmonids is maintained by the NY DEC for the Lake Champlain Fish and Wildlife Management Cooperative (Figure 35).

Stocking rates for Lake Champlain are determined by the Lake Champlain Fish and Wildlife Management Cooperative based on a variety of factors. Early in the development of the strategic plan for restoration of salmonid fisheries in Lake Champlain, it was recognized that maintaining a healthy smelt population was critical to a successful trout and salmon fishery (Lake Champlain Fish and Wildlife Policy Committee and Technical Committee 1977). Beginning 1996, stocking rates were reduced because the Cooperative recognized that the higher salmonid survival rates resulting from sea lamprey control and other changes occurring in the lake might mean that the rate of consumption of smelt could threaten the long-term stability of this population of forage fish (Fish and Wildlife Management Cooperative 1995a). Estimates of the smelt available to lake trout and other predatory fish are based on a monitoring strategy and bioenergetics model also developed in 1995 (LaBar and Parrish 1995). The current stocking rate is about 4 smolt equivalents per acre, which is similar to the rate used in the Great Lakes (Fish and Wildlife Management Cooperative 1995a). The mix of lake trout, Atlantic salmon, brown trout, and steelhead is based on consideration of the preferences of Lake Champlain anglers (Fish and Wildlife Management Cooperative 1995b).

The other major management activity focused on sport fish is the effort to reduce the population of parasitic sea lamprey. The primary approach is the application of chemical lampricides. TFM (3-trifluoromethyl-4-nitrophenol) was applied in tributaries where sea lamprey are known to nest and river deltas are treated with Bayer 73 as part of the eight year sea lamprey control study from 1991-1997. The amount of TFM applied in the tributaries depends upon the tributary discharge and the miles of stream accessible to sea lamprey. Initially, the entire length of stream accessible to sea lamprey tended to be treated, however, in a few cases the treated length was reduced as the study proceeded (Figure 36).

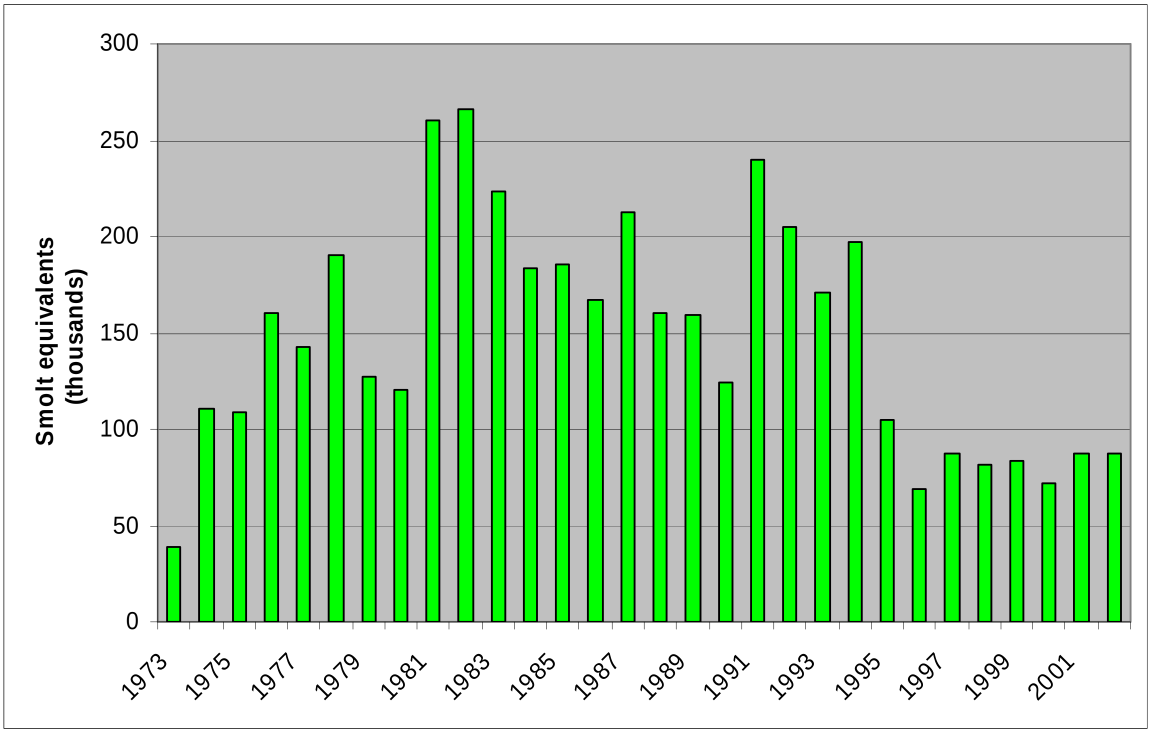



**Figure 35: Lake trout stocking to stations along the Main Lake from 1973-2002.**

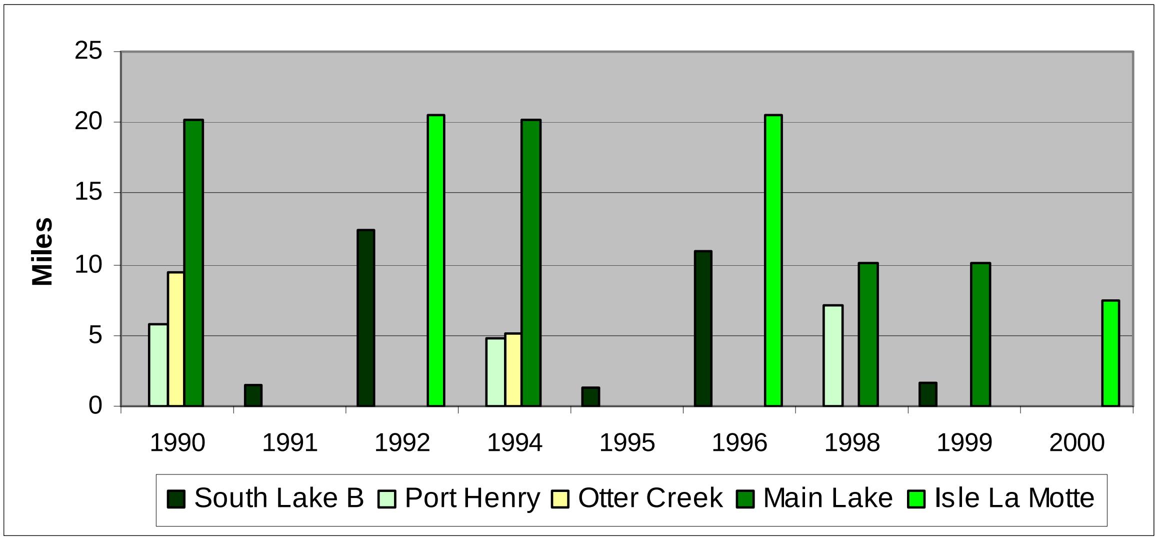



**Figure 36: Miles of stream exposed to TFM in lake segments with more than one TFM application.**

Although these chemicals have been formulated and adapted to act selectively on sea lamprey, there are nontarget impacts to other species in the lake and tributaries, including native brook lamprey, other fish species, macroinvertebrates, amphibians and mussels. For example, the effect of the application of Bayer 73 in the Little Ausable and Ausable River deltas on gastropods was examined over a five year period (Figure 37). There is a significant difference between in the number of gastropods pretreatment and the number immediately following treatment in 1991. These differences persisted in the 1992 samples, but had disappeared by the time that the 1995 samples were collected, suggesting that the gastropod population may naturally recover with time (Fish and Wildlife Management Cooperative Fisheries Technical Committee 1999). Total density and other measures of the fish community in Lewis Creek were made before a 1990 TFM application (Figure 38). Researchers concluded that there was "no undue adverse effect" on the fish community in Lewis Creek from the TFM application (Fish and Wildlife Management Cooperative Fisheries Technical Committee 1999).

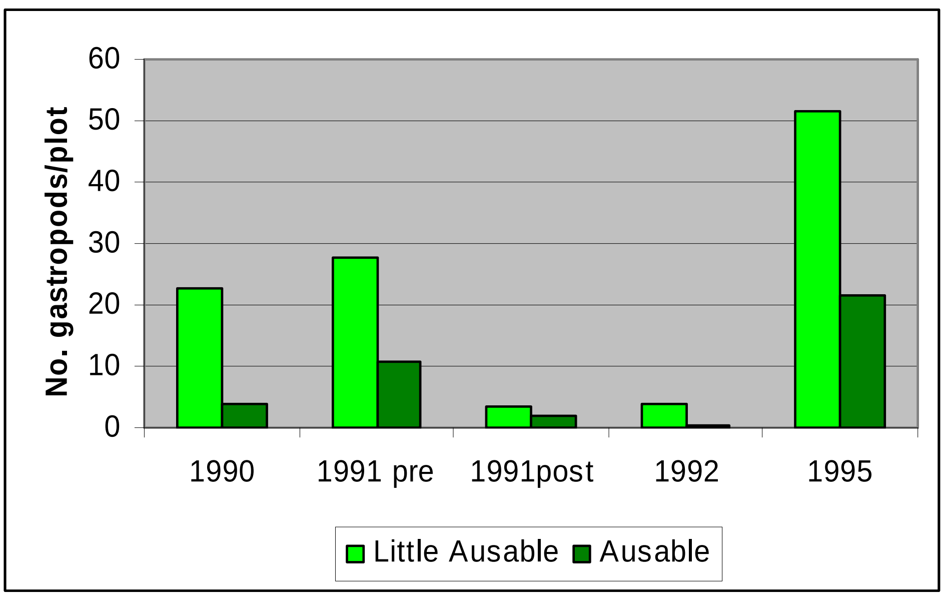



**Figure 37: Mean number of gastropods per plot before and after the application of Bayer 73 in the Little Ausable and Ausable River deltas. 1991 pre shows the mean number of gastropods prior to the 1991 treatment and 1991 post shows the mean number immediately after the treatment.**

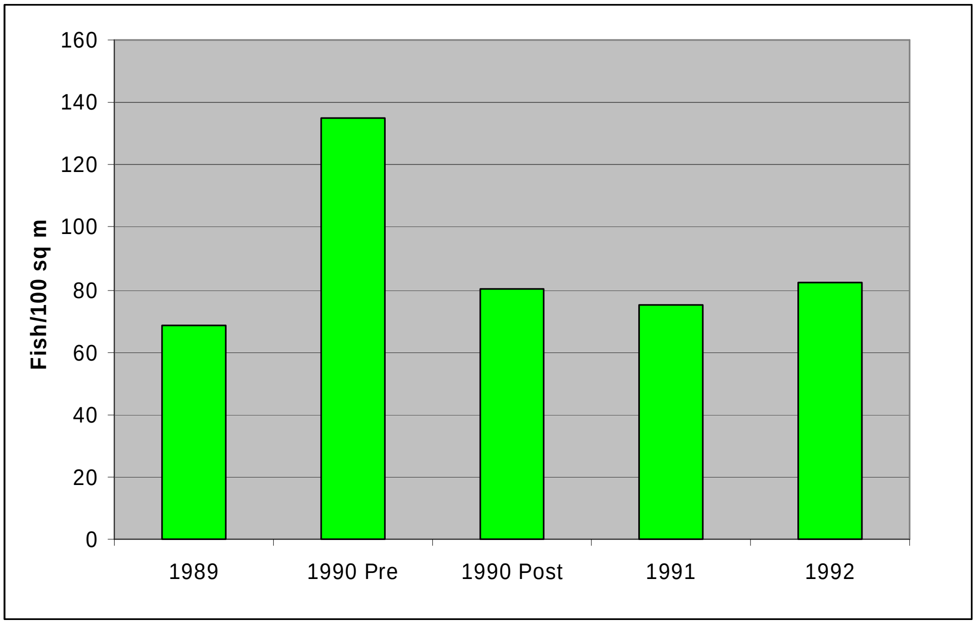



**Figure 38: Total density of the fish community at Lewis Creek before and after the 1990 TFM application. Densities were calculated based on number of fish collected from two electrofishing passes. 1990 Pre is the density prior to application and 1990 Post is the density after application.**

In an effort to reduce the need for chemical applications, various nonchemical alternative sea lamprey control measures have been tried. A long-term goal of the sea lamprey control program is to decrease the use of chemical treatment in favor of nonchemical alternatives (Fish and Wildlife Management Cooperative Fisheries Technical Committee 1999). However, we were unable to obtain data on the miles of stream treated with nonchemical alternatives.

Other factors that might influence the sport fish community in Lake Champlain include the introduction of exotic species, and phosphorus reduction, which may ultimately reduce the available phytoplankton at the base of the food web. Changes that might occur through the plankton are addressed in the next section.

## ASSESSMENT OF ADEQUACY AND ADDITIONAL MEASURES

A considerable amount of data is being collected in this area by the cooperators in the Lake Champlain Fish and Wildlife Management Cooperative; however, most of these data are not compiled and made available in electronic form to the user community. We recommend that these data be made more widely available and form the core of the measures used for the indicators in this issue area. We have proposed four pressure indicators. The Cooperative is collecting sea lamprey wounding rate data and angler harvest data, and these should be compiled and made available on an annual basis. Because habitat conditions and toxic contamination could also be driving the populations of sport fish, new data collection in these areas is needed.

Our state indicators relate to the population abundance of sport and forage fish. Currently, adequate data are being collected on the sport fish and smelt populations in the Main Lake, Inland Sea and Malletts Bay lake segments. However, little data is being collected outside of these areas. Also, a significant threat to the sport fish community is the invasion and expansion of exotic species of fish in the basin. A new state indicator is needed to track the population growth of these exotic species and their potential displacement of smelt in the food web. An additional plankton indicator is needed to tie the responses of the sport fish community to changes in the food web from the bottom up.

We have recommended response indicators that relate to sea lamprey control and stocking. Although good data are kept on the fish stocked into the lake, the data relating to sea lamprey control are incomplete and need additional attention, especially to track efforts in nonchemical control of lamprey.

## E. Issue Area: Pelagic Food Web

The plankton community responds directly to the nutrient levels and overall water quality in Lake Champlain. It is also the base of the food web ultimately leading to Lake Champlain's fish communities. The diversity and abundance of the plankton will determine much about both the ecological health of the lake, and its usefulness as a recreation and drinking water source.

The phytoplankton community in Lake Champlain historically has been dominated by cryptophytes, diatoms, and cyanobacteria, with cyanobacteria (also known as blue-green algae) densities highest in late summer and early fall (Myer and Gruendling 1979, Brown et al. 1992 and 1993, Shambaugh et al. 1999). Highest phytoplankton densities have consistently been observed in Missisquoi Bay and the South Lake.

Although cyanobacteria have always been a common part of the plankton of Lake Champlain, toxic blooms were not documented until 1999. Since that year, blooms have regularly occurred in portions of the lake, with the worst episodes occurring in Missisquoi Bay and St. Albans Bay. In these locations, visible surface scums have formed in each of the last four years (2000-2003). During the summer of 1999 and again in 2000, several dogs died after ingesting water from Lake Champlain containing large amounts of cyanobacteria. Beginning in 2000, a collaborative team from UVM, SUNY-ESF, and other institutions has documented the regular occurrence of toxin-producing cyanobacteria in the lake and presence of measurable and potentially hazardous amounts of toxin on several occasions (Rosen et al. 2001, Watzin et al. 2002, Watzin et al. 2003b).

The following PSR diagram (Figure 39) is preliminary and meant to stimulate discussion about both the causal relationships and the patterns of abundance of the plankton in Lake Champlain and their links to the rest of the food web and human uses of the lake.

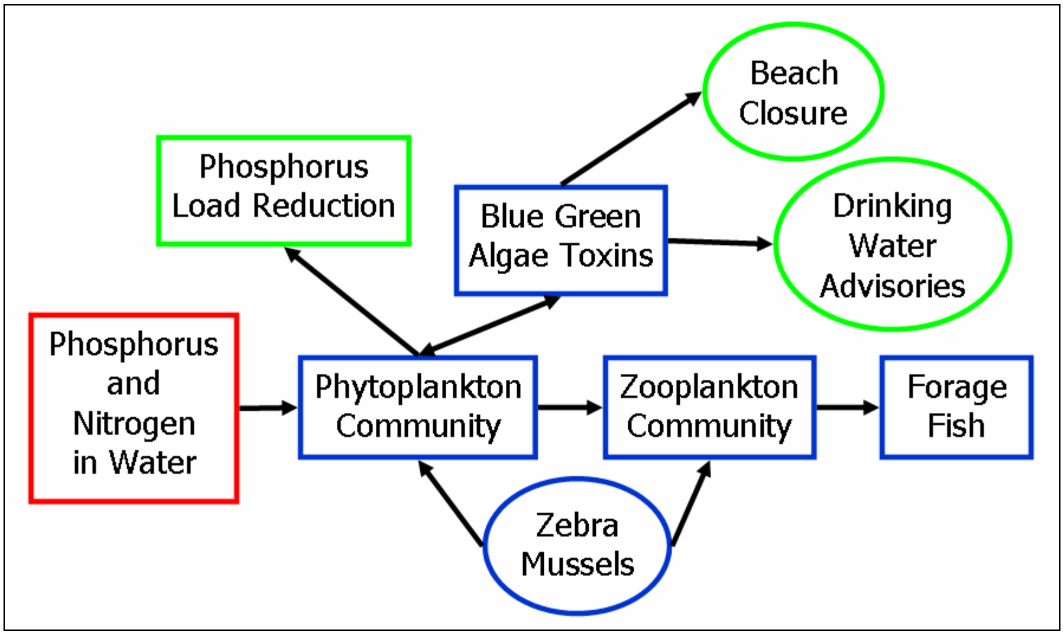



**Figure 39: PSR diagram for the pelagic food web.**

**Table 17: Indicators for the pelagic food web.**

| Indicator | P S R | Available Measure | Year | Source | Ref | Recommended Measure | Frequency (minimum) |
|---|---|---|---|---|---|---|---|
| P and N in water | P | Number of samples with N:P ≥ 50 | 1992-2002 | LCBP | Figure 40 | Number of samples with an N:P ≥ 50 by lake segment | Annually |
| Phytoplankton community | S | Taxonomic composition and relative abundance | 1992-2001 (some missing years) | LCBP and SUNY-Plattsburgh | | Taxonomic composition and relative abundance | Annually |
| | | Percent toxin producing cyanobacteria in selected locations | 2001-2003 | UVM | Table 18 | Percent toxin producing cyanobacteria by lake segment | Annually |
| Blue green algae toxins | S | Toxin concentrations by selected lake segment | 2001-2002 | UVM | Table 19 | Toxin concentrations by lake segment | Annually |
| Zooplankton community | S | Taxonomic composition and relative abundance | 1992-2002 | LCBP and SUNY-Plattsburgh | Figure 41 | Taxonomic composition and relative abundance; average size of the zooplankton; | Annually |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| | | | 1992-2002 | LCBP and SUNY-Plattsburgh | | Ratio of phytoplankton biomass to zooplankton biomass | Annually |
| Zebra mussels | S | Data for adults unavailable | | | | Biomass/$m^2$ for areas less than 30 m deep | Update every 5 years |
| Forage Fish | S | Mean rainbow smelt catch per trawl | 1987-2002 | LCFWMC | Figure 33 | Mean rainbow smelt catch per trawl | Annually |
| | | Number of exotic species in the lake | 2000 | LCBP – ANS plan | | Percent abundance of exotic species | Update every 2 years |
| Beach closure | R | | | | | Days of beach closure by lake segment | Annually |
| Drinking water advisories | R | | | | | Number of drinking water advisories by lake segment | Annually |
| Phosphorus load reduction | R | Annual mean tributary P load by lake segment | 1991-2002 | LCBP long-term biomonitoring | | Total phosphorus load reduction by lake segment | Every 2 years |

## PRESSURE INDICATORS

Phosphorus is considered the primary limiting nutrient in Lake Champlain, however, both phosphorus and nitrogen can limit the growth of the algal community and different phytoplankton groups have different specific nutrient requirements (Kilham 1990). Although there is generally a positive relationship between total phosphorus concentration and both total phytoplankton and cyanobacteria abundance, at least in the absence of zebra mussels (Huzsar and Caraco 1998, Paerl et al. 2001, Raikow et al. 2004), a more sensitive indicator of cyanobacteria abundance and dominance might be the relative quantity of available phosphorus and nitrogen (Smith 1983, Elser 1999).

In a study of 17 north temperate lakes, Smith (1983) found that cyanobacteria tended to dominant when the epilimnetic total N:P ratio fell below 29:1 by mass. Although a low N:P ratio alone does not mean a cyanobacteria bloom will occur, and several other studies have not found a relationship between N:P ratios and cyanobacteria abundance (Jensen et al. 1994, Scheffer et al. 1997) or found a weaker relationship than with either nutrient alone (Downing et al. 2001), the availability of N and P are still widely considered to be important factors contributing to cyanobacterial dominance (Hyenstrand et al. 1998b). Heterocystous cyanobacteria may have an advantage when N is limiting because they can fix atmospheric nitrogen (Howarth et al. 1988a, 1988b). The ability of many cyanobacteria to regulate their buoyancy, to be able to sink to where nutrients are available and then rise to where sunlight is optimal (Hyenstrand et al 1998a, Wallace and Hamilton1999) and the ability of some cyanobacteria to outcompete eukaryotic photoplankton for ammonium-nitrogen (Wilhelm et al. 2003) may also convey an advantage to the group.

Using the TN and TP data gathered by the LCBP Long-term Water Quality Monitoring Program, we found that the ratio of TN:TP does regularly exceed 29 in the summer in most years in the South Lake and the Main Lake (Figure 40). However, in St. Albans Bay, over the entire 10 year period from 1992 to 2002, the ratio only exceeded 29 once in 1997 and three times in 1999. In Missisquoi Bay, the ratio only exceeded 29 once in 2001 and four times in 2002. On average 15 samples are collected each year during the productive season.

Clearly, these patterns do not correspond with the cyanobacterial dominance we have seen in Lake Champlain. Based on data collected by Watzin et al. (2002, 2003b, 2004), the South Lake is never dominated by cyanobacteria; although the Main Lake is frequently dominated by cyanobacteria in the late summer, and was especially so in 1999 (Watzin et al. 2000), it does not regularly experience the noxious blooms observed in Missisquoi Bay and St. Albans Bay.

Although we have recommended that the ratio of N:P be used as an indicator, we also suggest that the relationship between the individual nutrient measures and cyanobacteria abundance continue to be explored. If additional data show one that the concentration of N or P alone provide a better indication of cyanobacterial dominance, then these could be substituted as indicators in the future.

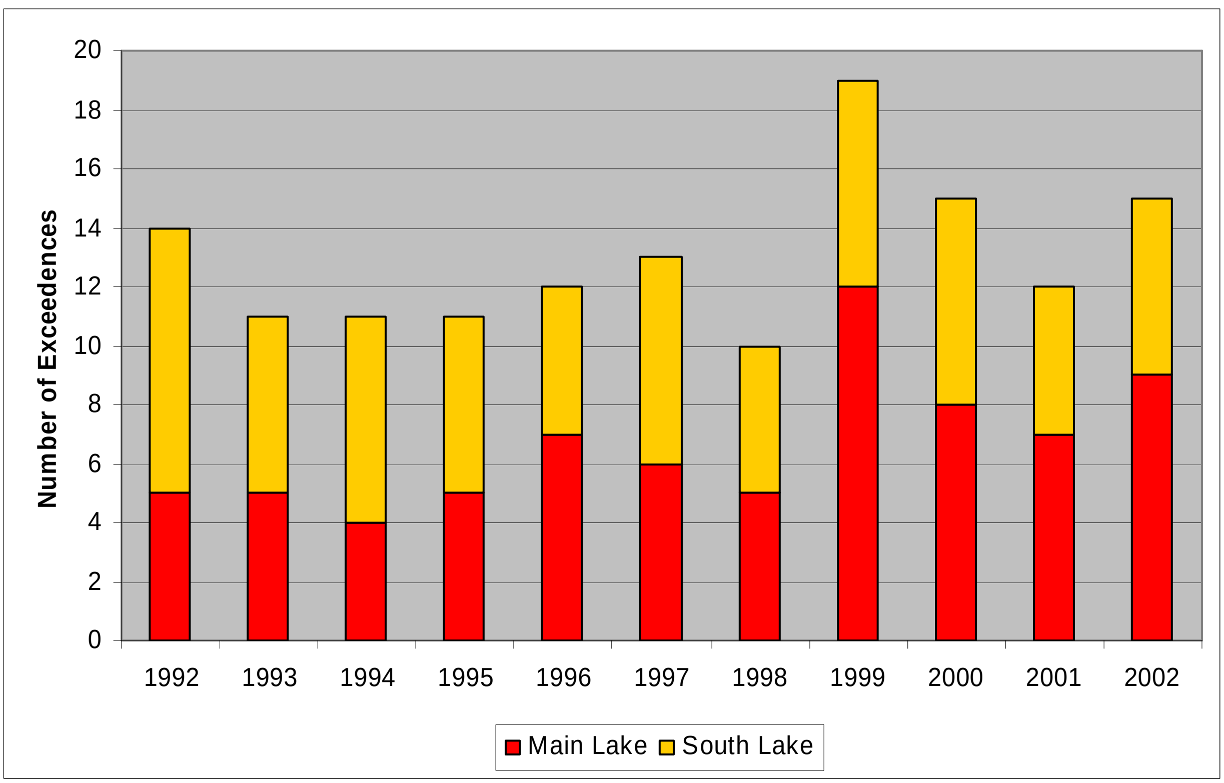



**Figure 40: Number of epilimnion and unstratified water quality samples taken from Main Lake and South Lake with a nitrogen to phosphorus ratio greater than or equal to 29 (on average, 15 samples wer e collected in each year in each lake segment).**

## STATE INDICATORS

Both nontoxic and potential toxin-producing cyanobacteria taxa have been observed in the plankton of Lake Champlain, but in the last four years, potential toxin producers have dominated (Table 18). The most common potential toxin-producers include several *Anabaena* spp., *Aphanizomenon flos-aquae*, and *Microcystis aeruginosa* (Rosen et al. 2001, Watzin et al. 2002, 2003b, 2004). Both anatoxin-a and microcystins have been detected throughout Lake Champlain and the range of concentrations seen is considerable (Table 19). In Missisquoi Bay and St Albans Bay, concentrations are well above the 1 μg/L limit recommended by the World Health Organization (1998) for the protection of human health and are a clear cause for concern.

**Table 18: Frequency of occurrence of potential toxin-producing species of cyanobacteria in Lake Champlain (Watzin et al. 2004).**

| | 2003 Frequency of Occurrence – Percent of Samples | | | |
|---|---|---|---|---|
| Taxon | Main Lake | South Lake | Missisquoi Bay | St Albans and other Northeastern Bays |
| *Anabaena flos-aquae* | 83 | 29 | 57 | 53 |
| *Anabaena* spp | 51 | 50 | 43 | 58 |
| *Microcystis aeruginosa* | 47 | 43 | 94 | 56 |
| *Coelosphaerium* spp. | 34 | 29 | 4 | 44 |
| *Gloeotrichia* spp. | 1 | 0 | 5 | 9 |
| *Aphanizomenon flos-aquae* | 73 | 43 | 22 | 49 |
| Samples Analyzed | 102 | 14 | 175 | 45 |

**Table 19: Range of toxin concentrations seen in Lake Champlain, 2001-2003 (Watzin et al. 2002, 2003b, 2004).**

| **Location** | **Microcystin (μg /L)** | **Anatoxin (μg /L)** |
|---|---|---|
| Main Lake | n.d. – 1.15 | n.d. – 1.2 |
| St. Albans Bay | n.d. – 114 | n.d. – 0.3 |
| Missisquoi Bay | n.d. – 2,500 | n.d. – 0.1 |

Zebra mussels are voracious filter feeders and as they have invaded aquatic ecosystems in North America, there have generally been significant changes in available nutrients and the plankton community, including changes in nutrient availability, decreases in phytoplankton density, a change in the dominant phytoplankton species, and reductions in small zooplankton (MacIsaac et al. 1995; Pace et al. 1998; Baker et al. 1998; Idrisi et al. 2001; Vanderploeg et al. 2001; Dionisio

Pires and van Donk 2002, Raikow et al. 2004, Brines 2004). What is causing this shift is not yet clear, although selective grazing and changes in nutrient availability are two potential hypotheses.

Herbivorous zooplankton may also influence phytoplankton community composition. Cladocerans are perhaps the most important grazers among the zooplankton, and several studies have shown positive correlations between *Daphnia* abundance and lower densities of cyanobacteria (Smith 1983, Elser 1999, Jacoby et al. 2000). Other studies have suggested that the toxins that cyanobacteria produce may make them undesirable or even toxic to zooplankton (Haney 1987, Demott 1999), and there is evidence that microcystin production is at least in part a response to grazing pressure (Jang et al. 2003).

The complexities of the food web in Lake Champlain are still largely unknown. In a research study conducted prior to the establishment of zebra mussels, Levine et al. (1997a and b) found that both nitrogen and phosphorus could control phytoplankton abundance in the Main Lake and nutrients generally influenced the phytoplankton more than grazing by zooplankton. In a laboratory study with a natural plankton community from the Main Lake, Brines (2004) found that phosphorus concentrations increased and N:P ratios decreased following the addition of zebra mussels to her experimental systems, a condition that could favor the cyanobacteria. Zebra mussels also caused significant declines in all phytoplankton groups, protozoa, rotifers, and copepod nauplii. This study suggests that there may be substantial changes occurring in the plankton of Lake Champlain through both direct and indirect pathways.

In an analysis of the trends in zooplankton composition and abundance at the Long-term Biomonitoring sites, Carling et al. (2004) found significant reductions in the rotifers in many Lake stations (Figure 41), including those where zebra mussels had not yet invaded (for example, Station 50 in Missisquoi Bay). They suggest, however, that because of the hydrologic connections between lake segments, these reductions might be the result of the high abundances of zebra mussels in the Main Lake. Although it is impossible to predict how the planktonic foodweb might adjust to the continuous presence of zebra mussels, clearly zebra mussels have the potential to strongly influence the plankton community and possibly those higher trophic levels that depend on the plankton.

There is an ever-growing body of literature that has examined the food web linkages in lakes and how they respond both to phosphorus enrichment from the bottom up and grazing pressure from the top down. This literature clearly shows that both pathways can have significant influences on biota that dominate and the resilience of the lake ecosystem (e.g., Sarnelle 1992, Kitchell 1992, Rudstam et al. 1993, Carpenter and Cottingham 1997, Carpenter et al. 2001, Jeppesen et al. 2003). While zooplankton biomass is necessary in bioenergetics models linking the plankton to planktivorous fish, the size of the zooplankton correlates strongly with its ability to control phytoplankton growth, with larger zooplankton exerting much stronger control. Therefore, we propose that zooplankton size be used as one measure for the zooplankton indicator.

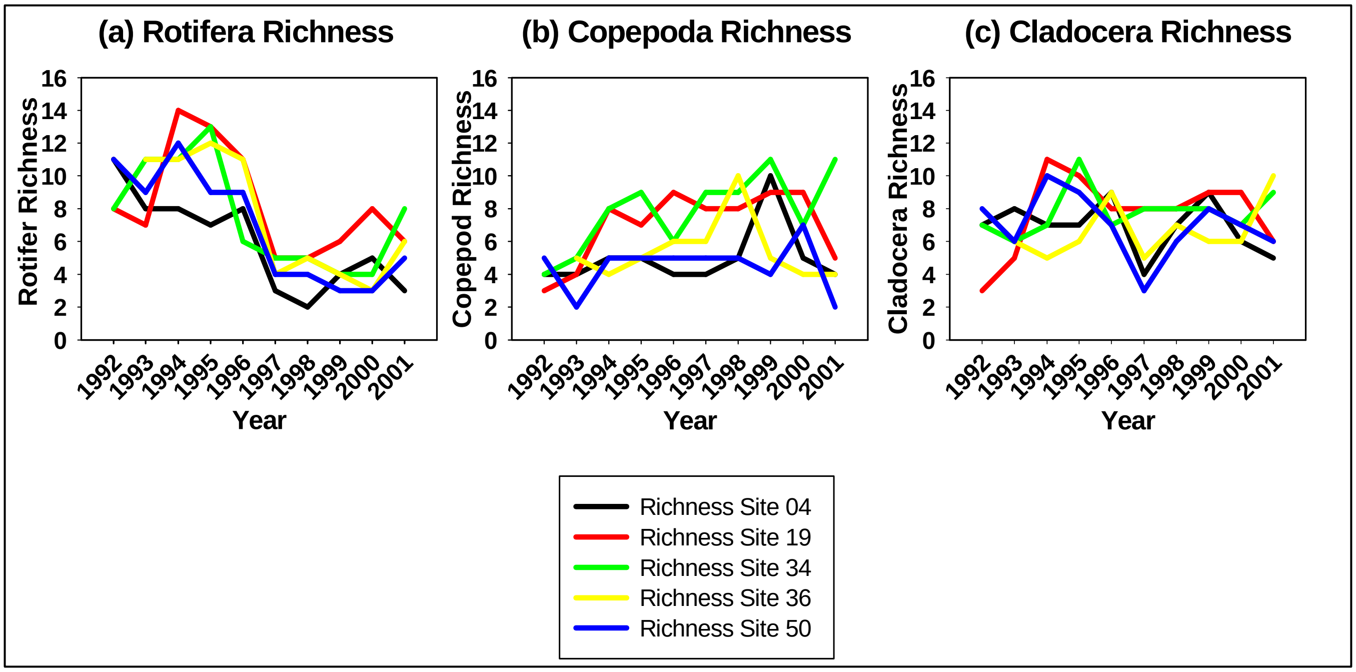



**Figure 41: Trends in selected zooplankton biodiversity measures at sites included in the Long-term Biomonitoring stations in Lake Champlain (from Carling et al. 2004).**

In an elegant study of 466 lakes, Jeppesen et al. (2003) showed that nutrient state and lake depth influence the ability of grazing fish to control phytoplankton biomass. In their analyses, fish predation was significant in both nutrient rich and nutrient poor lakes, but the cascading, top down effects of fish on phytoplankton were greatest in eutrophic lakes because of the lack of nutrient controls on phytoplankton and the low zooplankton to phytoplankton biomass ratio. In shallow lakes, the high benthic production also tended to support a higher planktivorous fish community which can increase grazing pressure. We suggest that a ratio of zooplankton to phytoplankton biomass could be developed as an indicator of the cascading effects through the food web in the various lake segments. Because of the substantial differences in lake segments, all the indicators in this issue area must be examined on a lake segment basis.

The literature discussed here, and many other studies clearly suggest that the state of the lake's aquatic community depends on both nutrient and fisheries management, and that these two management emphases should be considered in tandem. We are not yet doing that in the LCBP program, but as data are collected in this issue area, they might support an effort to move in this direction.

## RESPONSE INDICATORS

The primary management strategy to reduce nuisance blooms of cyanobacteria in Lake Champlain is phosphorus load reductions, therefore, the phosphorus load reductions achieved by lake segment are a good response indicator. In order to protect people from the hazards associated with exposure to cyanobacteria toxins, health advisories have been posted in Vermont and Quebec in the last several years. Both beach closings and drinking water restrictions have been necessary. Although these data have not been collected, the number of days of beach closure and the number of drinking water system restrictions would be useful indicators of these management actions. As cyanobacteria densities are controlled, these indicators should also decline.

**ASSESSMENT OF ADEQUACY AND ADDITIONAL MEASURES**

Currently, we have proposed just one pressure indicator in this issue area. Although from a bottom-up perspective, this indicator should capture the major human impact on the plankton, changes occurring from the top down are probably also important. It is also possible that other water quality changes may be contributing to the patterns in the plankton. These possibilities should be considered as more data are gathered in this issue area.

We have proposed six state indicators in this issue area that relate to the major compartments in the food web. Although phytoplankton and zooplankton data have bee collected since the early 1990s as part of the LCBP Long-term Biomonitoring Program, there have been problems in completing the enumeration of these samples and making the data available. As these data become more available additional effort should be expended to develop metrics that might relate to the taxonomic composition, relative abundance, and biomass of these two groups of plankton. Because food web structure varies tremendously across the lake segments, the indicators in this issue area should be followed on a lake-segment by lake-segment basis. Although toxic blooms have been most prevalent in the Missisquoi Bay lake segment, because toxic cyanobacteria blooms have become a major public concern and a priority for the LCBP, the percent potential toxin-producing cyanobacteria species should be tracked in all lake segments. Toxin concentrations should likewise be tracked in all lake segments. This will require an increased level of investment by the LCBP.

As previously discussed in the phosphorus issue area, estimates of the biomass of zebra mussels in each lake segments are needed to track the influence of this exotic species on the plankton. Although smelt populations are being monitored in some lake segments, additional indicators that relate to forage fish and the upper compartments of the food web should be developed.

The three response indicators that we recommend relate to the current management focus on toxic cyanobacteria blooms and reductions on phosphorus. The information necessary for tracking beach closures and drinking water advisories exist and should be compiled as recommended.

## F. Issue Area: Water Chestnut

According to the recent draft of the Aquatic Nuisance Species plan for Lake Champlain, at least 40 aquatic nuisance species (ANS) have been introduced into waters of the Lake Champlain

Basin. As part of the Long-term Biomonitoring Program, the LCBP had been keeping informal track of these invasions since 2001 (Eliopoulos and Stangel 2001).

Water chestnut (*Trapa natans* L.) is an invasive plant that can impede recreational and commercial water uses and negatively impact native plant communities (Hunt and Crawford 2002). It was first documented in Lake Champlain in the early 1940s in shallow bays in the southern end on both the Vermont and New York shores. It is generally assumed that water chestnut seeds were brought to Lake Champlain on boats traveling through the Champlain Barge Canal from the Mohawk or Hudson River (Countryman, 1970).

Water chestnut is one of the few manageable aquatic nuisance species found in Lake Champlain. Because of this, and the highly significant implications of this infestation for human use and enjoyment of the lake, we have developed a PSR model for it (Figure 42). Water chestnut is managed by harvesting the plants prior to seed formation, thereby inhibiting further spread of the infestation.

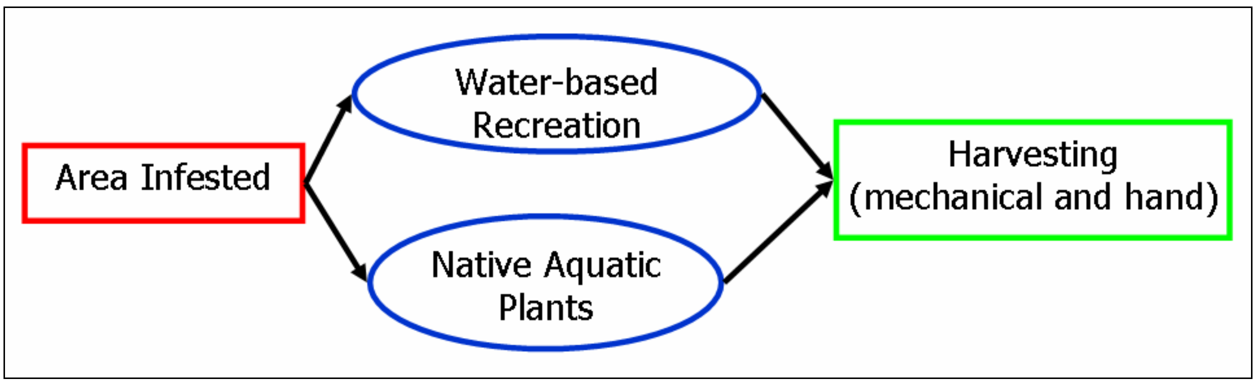



**Figure 42: PSR diagram for water chestnut management.**

**Table 20: Indicators for water chestnut management.**

| Indicator | PSR | Measure | Year | Source | Ref | Recommended | Frequency (minimum) |
|---|---|---|---|---|---|---|---|
| Area Infested | P | Miles north of Whitehall, NY | 1982-2002 | VT DEC and LCBP | Figure 43 | Miles north of Whitehall, NY | Annually |
| Water-based Recreation | S | Data unavailable | | | | Develop measure of recreational use | Update every 5 years |
| Native Aquatic Plants | S | Data unavailable | | | | Native species present and percent cover in shallow water by affected lake segment | Update every 5 years |
| Harvesting | R | Dollars spent on harvesting | 1991-2002 | LCBP | | Dollars spent on harvesting | Annually |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| | | Number of mechanical harvester loads | 1982-2002 | LCBP | Figure 43 | Number of mechanical harvester loads | Annually |
| | | Biomass removed by hand-pulling | ??? | LCBP | | Biomass removed by hand-pulling | Annually |

## PRESSURE INDICATORS

The area of the lake infested by water chestnut, measured in miles north from the southern end of the lake in Whitehall, NY and indicates the extent of the pressure on the lake ecosystem. The northern most point of the infestation, measured in miles from Whitehall, NY has been tracked since the early 1980s (Figure 43). This indicator is directly responsive to management, which has focused on keeping the area of infestation from moving north in the Main Lake.

## STATE INDICATORS

Although surveys of the location and density of water chestnut and native aquatic plants were conducted by the VT DEC in 1994 and from 1998 to 2003, these data only exist in paper records and are not generally available. However, because water chestnut displaces other aquatic plant species and is of little food value to wildlife, an appropriate state indicator would be the relative dominance of the plant cover by water chestnut.

Water chestnut is known to be a nuisance for water-based recreationalists. However, we are not aware of any systematic data that quantifies water-based recreation on Lake Champlain. If, in fact, a justification for management is the desire to use the lake for recreation, then an indicator of recreational use in areas of historic infestation should be developed.

## RESPONSE INDICATORS

Water chestnut has been harvested from Lake Champlain with varying intensities since 1982 (Figure 43). There is a noticeable decline in the northern extent of water chestnut following years of intensive harvesting. Because the seeds remain viable in the seed bed for up to twelve years (Countryman 1970), harvesting needs to occur every year in order to prevent reinfestation. With persistent management year after year, the infestation of water chestnut could be contained if not eradicated in some areas

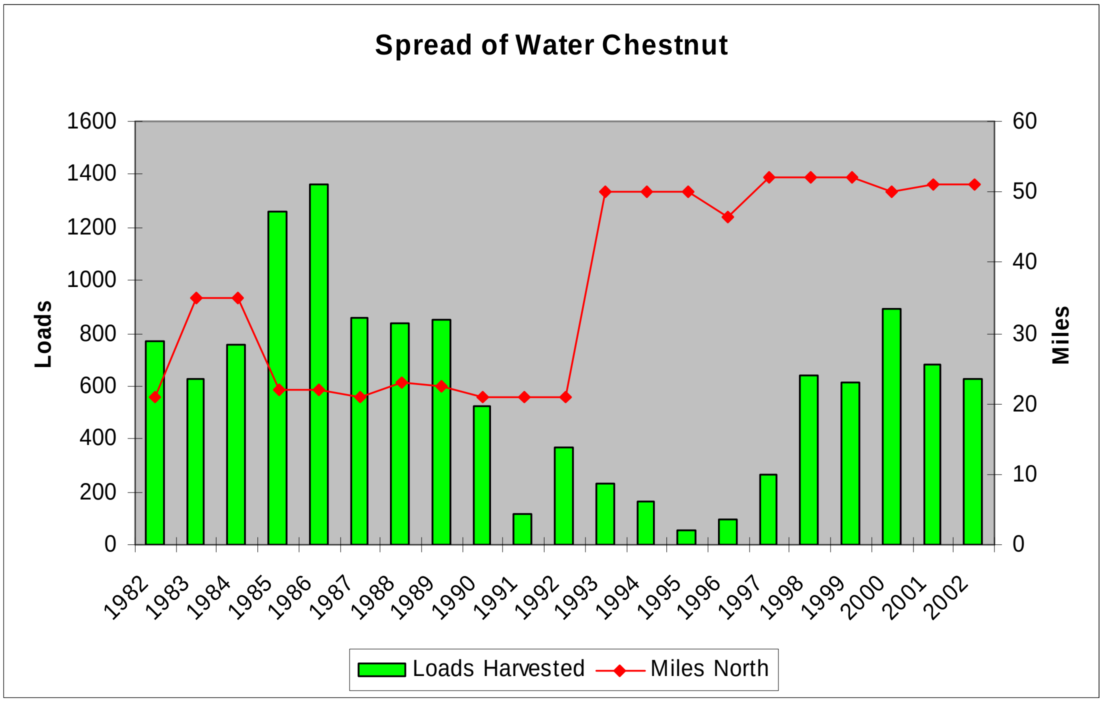



**Figure 43: Spread of water chestnut north of Whitehall and loads of water chestnut harvested from 1982 – 2002 (data from LCBP).**

### ASSESSMENT OF ADEQUACY AND ADDITIONAL MEASURES

Our pressure indicator builds on a long-term data set, but ignores water chestnut in northern sections of the lake. In 1998, the first population of water chestnut in the north was found in Quebec, Canada. This new infestation was located in the South River approximately 9 miles northwest of Missisquoi Bay, Lake Champlain (LCSC 2003). Plants have since been found in the Richelieu River and Pike River, Quebec (Hunt and Crawford, 2003). A pressure indicator for this infestation should also be developed.

Neither of our proposed state indicators is currently being measured. Both these indicators relate directly to the reasons so much management attention is focused on this species, and thus should be added as soon as practical.

## G. Issue Area: Recreation and Cultural Heritage Resources

One of the goals in the management plan *Opportunities for Action* is to manage Lake Champlain, its shoreline and tributaries for diverse recreational uses. Lake Champlain related tourism and recreation are major contributors to the economy of the region; tourism brought an estimated $3.8 billion to the basin in 1998/99 (LCSC 2003). This demand is based, in part, on the condition of the ecosystem. If the condition of the lake were to degrade substantially, a corresponding drop in recreational and tourism activity could be expected. However, as the number of people residing, visiting and recreating in and around Lake Champlain increases, the potential for ecosystem degradation also increases. Therefore, ecosystem management efforts in other issue areas to mitigate the impact from an ever increasing number of people can also be considered a response indicator in this area as well. The diagram presented here (Figure 44) is preliminary and meant to stimulate discussion about the linkages between recreation management and the other issues areas.

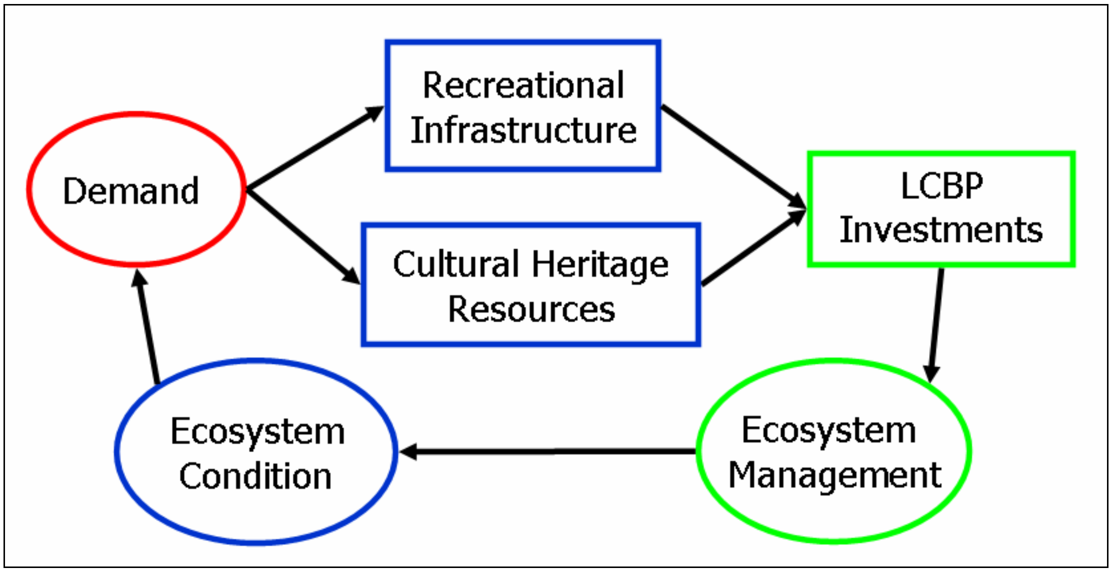



**Figure 44: PSR diagram for recreation and cultural heritage resources.**

**Table 21: Indicators for recreation and cultural heritage resources.**

| Indicator | PSR | Measure | Year | Source | Ref | Recommended | Frequency |
|---|---|---|---|---|---|---|---|
| Demand | P | | | | | Population and tourism growth rates | Update every 5 years |
| | | | | | | Lake-related recreation interests, both residents and tourists | Update every 10 years |
| Ecosystem Condition | P | State measures in other issue areas | | | | | Update every 5 years |
| Recreational Infrastructure | S | Miles of bikeway around Lake Champlain | 1995-2003 | LCBP | Figure 45 | Miles of bikeway around Lake Champlain | Annually |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| | | Number of public access sites or improvements around Lake Champlain funded by LCBP | 1995-2003 | LCBP | | Number of public access sites around Lake Champlain per capita (residents and tourists) by lake segment | Update every 5 years |
| | | | | | | Congestion and adequacy of harbor facilities by lake segment | Update every 5 years |
| Cultural Heritage Sites | S | Number of interpreted wayside exhibits in the Lake Champlain basin | 2003 | LCBP | | Number of interpreted wayside exhibits in the Lake Champlain basin | Update every 5 years |
| | | | | | | Number of interpreted cultural heritage sites in the Lake Champlain basin | Update every 5 years |
| LCBP Infrastructure Investments | R | Dollars granted by LCBP for recreation and cultural heritage projects | 1993-2001 | LCBP | Figure 46 | Dollars granted by LCBP for recreation and cultural heritage projects | Annually |
| Ecosystem Management | R | Response measures in other issue area | | | | Response measures in other issue area | As appropriate |
| | | Dollars spent on environmental education and outreach | 1994-2003 | LCBP | Figure 47 | Dollars spent on environmental education and outreach | Annually |

## PRESSURE INDICATORS

Although not everyone living in and visiting the watershed engages in recreational activity associated with Lake Champlain, it is reasonable to assume that the recreational demand associated with Lake Champlain will increase in proportion to the increase in the total number of people in the basin. The basin resident population has been increasing steadily for the last 50 years (Figure 4). The tourist population is also large and growing although data specific to tourism on Lake Champlain are lacking.

Demand is a complex function of population and interests. There is currently little data on recreation interests that are tied directly or indirectly to Lake Champlain. These data could be obtained using an appropriately designed survey instrument.

## STATE INDICATORS

Public access is a prerequisite for boating, swimming, fishing, SCUBA diving, sightseeing, bicycling, wildlife watching, camping and the myriads of other ways residents and tourists enjoy Lake Champlain and its shoreline. Anecdotal evidence about competition for boat mooring and docking slips throughout the Lake and a 1995 study on harbor congestion and boater conflicts in Malletts Bay (T.J. Boyle and Associates 1995) suggests that the recreational infrastructure to support some activities may be inadequate. However, without more data on demand for particular activities at particular locations in the basin, it is difficult to develop a quantifiable measure or indicator of the adequacy of public access to the lake. One possibility is to array the number of public access points in each lake segment, along with the number of people they are designed to support in terms of parking spaces, boat trailer parking, restroom facilities, etc. with the resident and seasonal or tourist populations for those lake segments.

Because of increased interest in bicycling, the LCBP has provided seed money and support to efforts to facilitate the development of bike paths and maps of biking routes around the lake. The “Lake Champlain Bikeway” was created in 1995. Since then miles have continued to be added to the network (Figure 45). There are currently 1,100 miles in the Lake Champlain Bikeways, including a 350-mile loop around the lake and 27 shorter theme loops scattered throughout the basin.

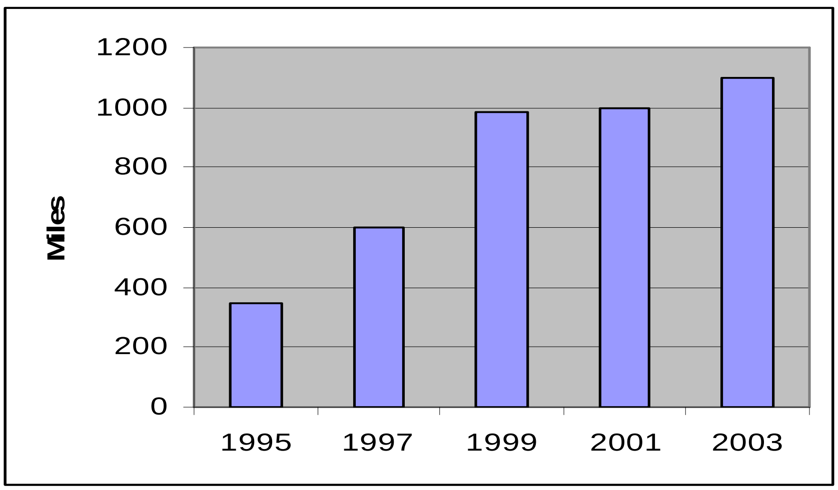



**Figure 45: Miles of bikeways included in the Lake Champlain Bikeway.**

Because of Lake Champlain’s role in early American history, the region is rich with cultural heritage resources including underwater shipwrecks, national historic sites and museums. There are now six underwater Historic Preserves that are maintained by Lake Champlain Maritime Museum. These are located at shipwreck sites that SCUBA divers can visit. In 2001, the LCBP designed a template for a unified system of outdoor wayside exhibits. In 2003, there were 62 exhibits in that system (LCBP 2003), and more are continually added. The number of interpreted wayside exhibits would be a useful indicator in the future.

## RESPONSE INDICATORS

Through the Public Access Grants Program and Partnership Program grants, the LCBP has invested in creating new public access sites, enhancing existing public access sites around the

Lake Champlain shoreline, and supporting a variety of recreation and cultural heritage projects for both residents and tourists (Figure 46). The level of investment increased significantly in the late 1990s but declined in 2001 because funding to support these activities from the National Park Service was interrupted. To date, the LCBP has spent $140,569 for public access.

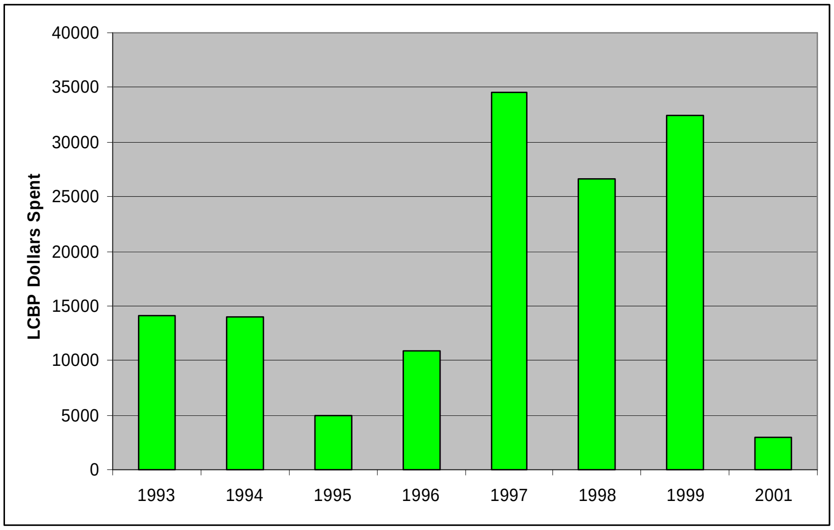



**Figure 46: LCBP Grants for recreation and cultural heritage projects.**

A variety of education efforts have been undertaken by the LCBP to raise awareness about the condition of the Lake Champlain ecosystem and to cultivate environmental stewardship in the basin. Public awareness and education are essential in order to help support the changes in personal behavior that are prerequisite to better conditions across all issue areas, and thus the level of activity in this area can serve as one indicator of management response (Figure 47).

Increases in recreation and tourism in the Lake Champlain Basin depend on the condition of the lake and but also influence the condition of the lake, therefore, additional information about demand and public perceptions about lake condition is needed.

## ASSESSMENT OF ADEQUACY AND ADDITIONAL MEASURES

Because it is a combination of demand and ecosystem condition that drive recreational pursuits in the Lake Champlain basin, we have recommended that indicators be developed in these areas. Although population data exist for the Lake Champlain Basin, we were not able to find lake-specific tourism data. Along with specific data on lake-related recreation interests in the watershed, these data are a critical need.

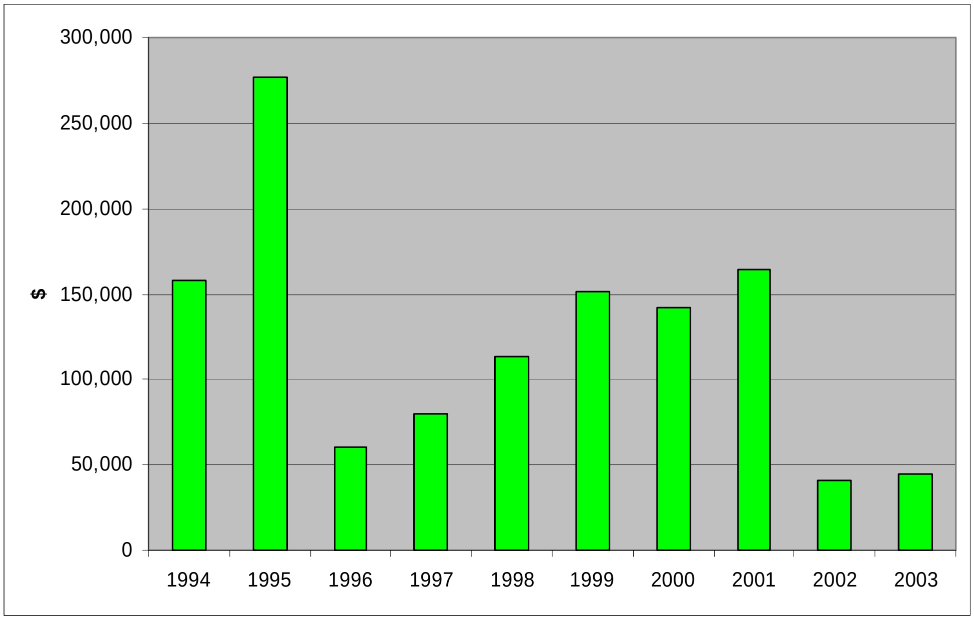



**Figure 47: LCBP funds spent on public outreach and education.**

The state indicators we have recommended are based on the infrastructure and resources available to support recreation and cultural heritage tourism in the basin. Clearly additional measures that capture these indicators could be developed.

There are two major categories of management actions that are undertaken to support lake related recreation and heritage tourism. One category is the investments made by the LCBP to develop the appropriate infrastructure, and the other is investments in ecosystem management activities. Additional measures in both categories could clearly be developed over time.

## H. Overall Assessment of PSR Models

An overall assessment of the condition of the Lake Champlain ecosystem and the effectiveness of management can be derived from a collective analysis of all the PSR diagrams. In such an analysis, it immediately becomes clear that the human population residing in or visiting the basin, and the activities that this population undertakes, are the penultimate stressors in the Lake Champlain ecosystem. As the human population grows, more land is consumed, more waste and pollution are generated, and additional stresses are placed on the Lake Champlain ecosystem. In order simply to maintain the current condition of the ecosystem, more aggressive management is needed. To improve the condition, an extraordinary level of effort will be required.

The changes that are occurring throughout the lake's ecosystem are occurring through diverse and complex pathways. The more we understand about these pathways, the more effectively management can target responses to control unwanted changes. An ecosystem management approach will required an explicit consideration of the position and role of people as part of the Lake Champlain ecosystem. A proactive approach to management will require a comprehensive information base that can be used in adaptive management over the long-term.

## III. DEFINING ACCEPTABLE LEVELS

In order to interpret the indicators we have developed, the LCBP must define acceptable levels for these indicators. Those levels then become the management targets. As introduced previously, acceptable levels can be defined from both a social and an ecological perspective. The following sections present the analyses we undertook to inform the definition of acceptable levels for a core set of indicators. In these analyses, we focused on the state indicators. If an acceptable level of these indicators can be defined, then based on the linkages in the ecosystem, definition of the acceptable levels of the pressure indicators and response indicators will follow.

From the ecological perspective, we concentrated on statistical approaches to uncertainty analysis. We also discuss the notion of ecological thresholds and outline some ways to approach acceptable ranges based on this concept. From the social perspective, we conducted a choice experiment using five key characteristics of the Lake Champlain ecosystem. We also generated social norm curves for selected indicators.

### A. Acceptable Levels from an Ecological Perspective

**Introduction to Uncertainty**

While there are many sources of uncertainty in ecological indicator projects, most of the case studies we reviewed do not acknowledge or attempt to quantify uncertainty. Uncertainty is the condition of being in doubt. MacIntosh et al. (1994) defined two major types of uncertainty; knowledge uncertainty and stochastic variability. Knowledge uncertainty is the result of incomplete understanding or inadequate measurement of system properties and is a property of the analyst. Knowledge uncertainty can be further partitioned into model and parameter uncertainty. Stochastic variability is the result of unexplained random changes in the natural environment and is a property of the system under study. Stochasticity can be further subdivided into temporal and spatial variability.

Quantifying uncertainty amounts to "intellectual honesty" (Reckhow 1994, Haan 1995) and can provide valuable information for decision makers (Hession and Storm 2000). First, quantifying uncertainty provides information on the reliability of the predictions. In other words, a low prediction uncertainty indicates a high value of information contained in the prediction (Chapra and Reckhow 1979). Second, uncertainty analysis allows for designing a management program or pollutant reduction strategy to achieve a desired ecological state with a specified probability, thus accounting for the possibility of success and failure (Hession and Storm 2000). A particular management decision does not result in a single environmental response to be realized year after year but a whole range of responses to which probabilities can be assigned (Haan 1995). Water quality managers want to avoid concluding that a water body is impaired when it is not (false positive) or that it is not impaired when it is (false negative) (Shabman and Smith 2003).

Ecosystems are complex, variable, and diverse in nature (Kelly and Hartwell 1990), and there are many sources of uncertainty in the data used to develop indicators (Hunsaker et al. 1990). Interpreting indicator levels requires an understanding of the natural variation (Landres 1992) and a recognition and quantification, when possible, of the inherent uncertainties (Hunsaker et al. 1990). Regional indicator programs may be subject to uncertainty as a result of differences between sampling protocols as well as spatial heterogeneity in the landscape over a large geographic area (Hunsaker et al. 1990). Measuring indicators at discrete time intervals is another source of uncertainty because the timing of measurements may not coincide with the timing of environmental stressors or changes (Hunsaker et al. 1990).

The natural variability of indicators can be so high that their use in detecting changes in the ecosystem as result of the pressures on it can be very low. Kelly and Harwell (1990) define the signal-to-noise ratio of an indicator as the combined sensitivity of the indicator to respond to a given perturbation compared to the normal variation of the indicator over time and space. A high signal-to-noise ratio is required for sensitive, stress-specific indicators, while a low signal-to-noise ratio is acceptable for screening indicators, especially involving inexpensive or easily measured variables. The signal-to-noise ratio of indicators cannot be evaluated without an explicit quantification of the variability inherent in the indicators. Identifying and understanding the full range of possibilities, as presented stochastically through statistical procedures or a quantitative uncertainty analysis, provides useful information for planning and management (Hession et al. 1996, Hession and Storm 2000).

**Example Analysis**

As an example of how uncertainty can be addressed, we analyzed the phosphorus data collected as part of the LCBP Long-term Biomonitoring Program. In 2003, the Vermont and New York Departments of Environmental Conservation developed the phosphorus TMDL for Lake Champlain (VTDEC and NYSDEC 2002). The TMDL was developed based on the phosphorus concentration criteria for each lake segment that the two states have adopted for Lake Champlain (Smeltzer 2003). Phosphorus concentration samples have been taken somewhere between biweekly and triweekly from April through November from 1992 through 2004. Sampling frequency was developed based on a power analysis to "achieve adequate power in detecting environmental change over time in Lake Champlain (VTDEC and NYSDEC 2004)." The procedure assumes that environmental change will be analyzed using a t-test for the difference in the mean values for a water quality variable between two time periods (e.g. a baseline period vs. a post-treatment period). The statistical power should be adequate to allow for detecting a 15% change when comparing a 10-year baseline period and a 10-year post-treatment period assuming triweekly sampling. Given that the sampling frequency is often closer to biweekly, the states suggest that it might be possible to evaluate change between 4 years of baseline and 4 years of post-treatment data adequately (VTDEC and NYSDEC 2004). In either case, the states recommend that change will be detected by comparing the annual mean after some baseline and post-treatment period (not clearly specified in VTDEC and NYSDEC 2004). The paired t-test does not determine compliance with the criteria; it only compares the means for two time periods. Also, although the t-test would be performed assuming there actually was a baseline and a post-treatment period, this is really not the case as changes in water quality (both good and bad) have been occurring continually since 1992.

Until recently, US EPA guidance suggested that if 10% of the samples violated a criterion, then the water body should be listed as impaired (Smith et al. 2001). If a very large data set exists, determining if the criterion is exceeded 10% of the time would be a fairly simple statistical procedure. For Lake Champlain, we do not have large sample sizes (generally, 10 to 20 samples per year). If one of the 10 samples exceeds the P concentration criterion for a given lake segment, the criterion is equaled and the lake segment would be listed as impaired. This decision has a probability of about 9-24% of being a false positive (in other words, the segment is listed as impaired when it actually is not) and a probability of about 26-57% of being a false negative (not listed as impaired when it actually is impaired), calculated following Shabman and Smith (2003). This illustrates how uncertain our management actions could be based on this data set. Autocorrelation of the sample data, which obviously is the case the this data set, can increase the uncertainties involved (Smith et al. 2003)

We analyzed the Main Lake, Missisquoi Bay, South Lake B, South Lake A, and Port Henry segment P concentration data in order to determine what the current probabilities that they are meeting their water quality criteria might be. These lake segments had the most data available for analysis. To illustrate, we selected the baseline period to be from 1992 through 1996 (5 years) and the post-treatment period to be from 1997 through 2001 (5 years). Each data set was fit to a lognormal distribution and is presented in the following figures as complementary cumulative distributions functions (CCDFs) that define a probability of exceedence (Helton 1994). In addition, the median concentrations for each time period are plotted along with the CCDFs and we performed paired t-tests to evaluate if the arithmetic means or the log-tranformed means between these time periods were significantly different (Table 22).

**Table 22: T-test results for comparing phosphorus concentrations for two time periods, 1992-1996 and 1997-2001.**

| Lake Segment | Phosphorus Concentration Mean (+ std. dev) (μg/l) | | T-Test Results (p value) | | | |
|---|---|---|---|---|---|---|
| | | | Raw Data | | Log-Transformed Data | |
| | 1992-1996 | 1997-2001 | 1-Tail | 2-Tail | 1-Tail | 2-Tail |
| Main Lake | 0.012 (0.004) | 0.011 (0.003) | **0.023** | **0.047** | **0.043** | **0.086** |
| Missisquoi Bay | 0.046 (0.017) | 0.047 (0.018) | 0.34 | 0.67 | 0.35 | 0.70 |
| Port Henry | 0.014 (0.004) | 0.014 (0.003) | 0.19 | 0.39 | 0.09 | 0.17 |
| South Lake A | 0.035 (0.012) | 0.034 (0.012) | 0.32 | 0.65 | 0.23 | 0.47 |
| South Lake B | 0.053 (0.018) | 0.048 (0.019) | **0.05** | **0.09** | **0.03** | **0.06** |

Note: QAPP states that alpha = 0.05 for 1-tail and alpha = 0.10 for 2-tail t-tests. Significant differences are in **bold**.

The Main Lake segment water quality criterion has been set at 10 µg/L (VTDEC and NYSDEC, 2002). While there is a significant different between the means P concentrations for the two time periods, the second period (1997-2001) still would not meet water quality criteria (Table 22). However, based on the fitted lognormal distributions (Figure 48), there is a 70% probability that the Main Lake exceeded the criterion from 1992-1996 and a 65% probability of exceedence from 1997-2001. The more recent time period (1997-2001) has a lower probability of exceeding the standard, but still has a high probability of exceedence. The CCDFs provide more information than a simple t-test.

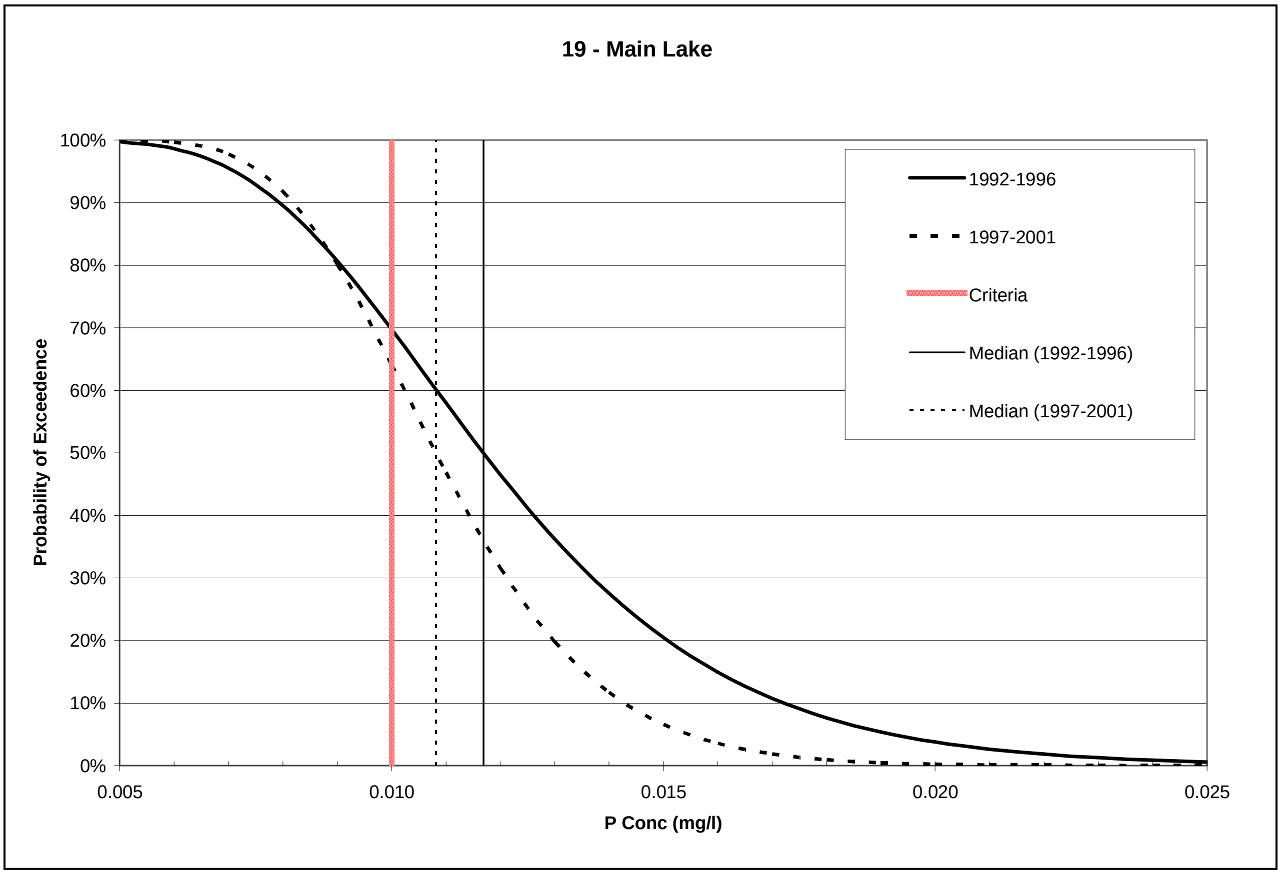



**Figure 48: Complementary cumulative distribution functions of phosphorus concentration in the Main Lake segment: 1990-1995 versus 1996-2001.**

The Missisquoi Bay segment water quality criterion has been set at 25 µg/L (VTDEC and NYSDEC, 2002). According to the t-test results (Table 22), there is no significant difference between the two time periods tested. In addition, inspection of the fitted lognormal distributions (Figure 49) suggests that there is very little change in the phosphorus concentration in Missisquoi Bay, there is about a 95% probability that the Missisquoi Bay lake segment exceeded the criterion from 1992-1996 as well as from 1997-2001.

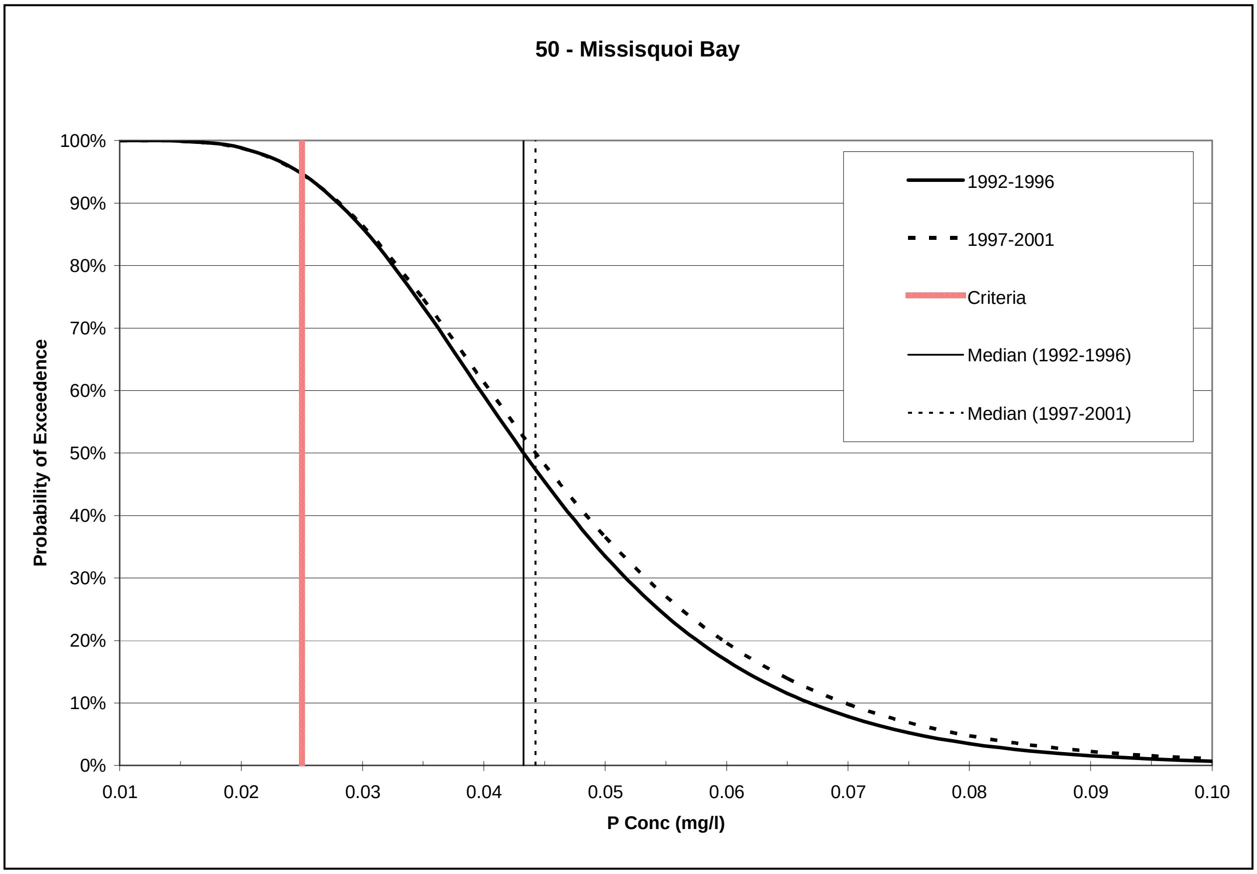



**Figure 49: Complementary cumulative distribution functions of phosphorus concentration in the Missisquoi Lake segment: 1990-1995 versus 1996-2001.**

The South Lake B segment water quality criterion has been set at 54 μg/L (VTDEC and NYSDEC 2002). The t-test does suggest that there is a significant difference in P concentration between the two time periods analyzed. The mean P concentration has gone from 53 μg/L to 48 μg/L. At this point, it appears that the lake segment is meeting its criteria. However, the current Quality Assurance Plan for the Long-term Biomonitoring program (VTDEC and NYSDEC 2004) does not specifically state how such a determination will be made (e.g., how many years of data must be compared). Inspection of the fitted lognormal distributions (Figure 50), suggests that there is still a 30% probability that the South Lake B segment exceeds the criterion prior from 1997-2001, down from a 42% probability of exceedence from 1992-1996. Again, the CCDF provides additional information over the simple t-test comparison or comparing the mean to the criteria.

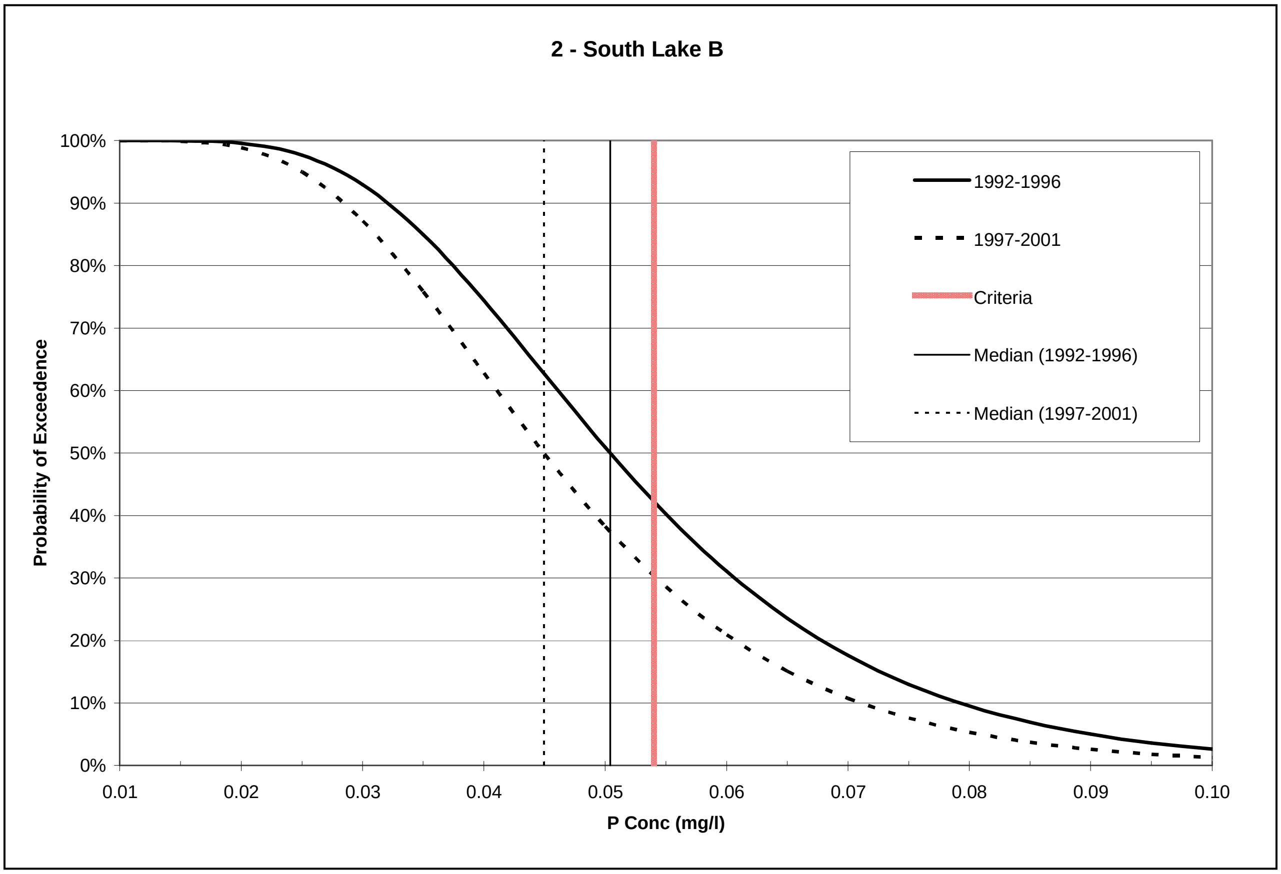



**Figure 50: Complementary cumulative distribution functions of phosphorus concentration in the South Lake B segment: 1992-1995 versus 1996-2002.**

The South Lake A segment water quality criterion has been set at 25 µg/L (VTDEC and NYSDEC 2002). There is no significant difference in P concentration between the two periods tested (Table 22). The fitted lognormal distributions (Figure 51) suggest there is very little change in the distributions of P concentration for the two periods. There is an 82% probability that the South Lake A segment exceeded the criterion from 1992-1996 and a 77% probability of exceedence from 1997-2001. In this case the more recent time period (1996-2002) has a lower probability of exceeding the standard, but there is basically no difference in probabilities at the higher P concentrations.

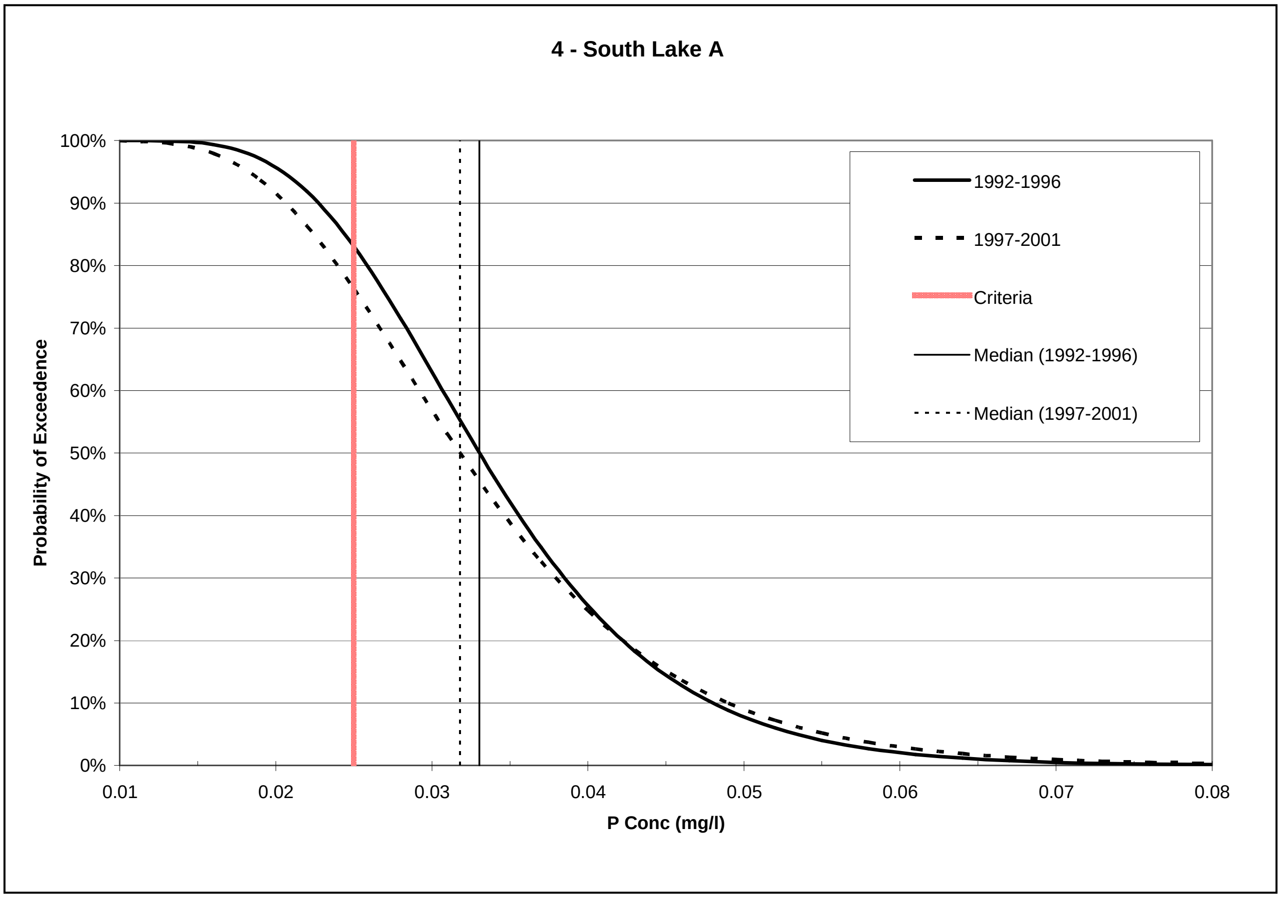



**Figure 51: Complementary cumulative distribution functions of phosphorus concentration in the South Lake A segment: 1992-1995 versus 1996-2002.**

Finally, the Port Henry segment water quality criterion has been set at 14 μg/L (VTDEC and NYSDEC 2002). Again, the t-test suggests there is no significant difference in P concentration between the two time periods tested (Table 22). However, based on the fitted lognormal distributions (Figure 52), the distributions are different and there is higher probability (50%) that the Port Henry segment exceeded the criterion in the more recent time period (1992-1996) than the earlier time period's 42% probability of exceedence.

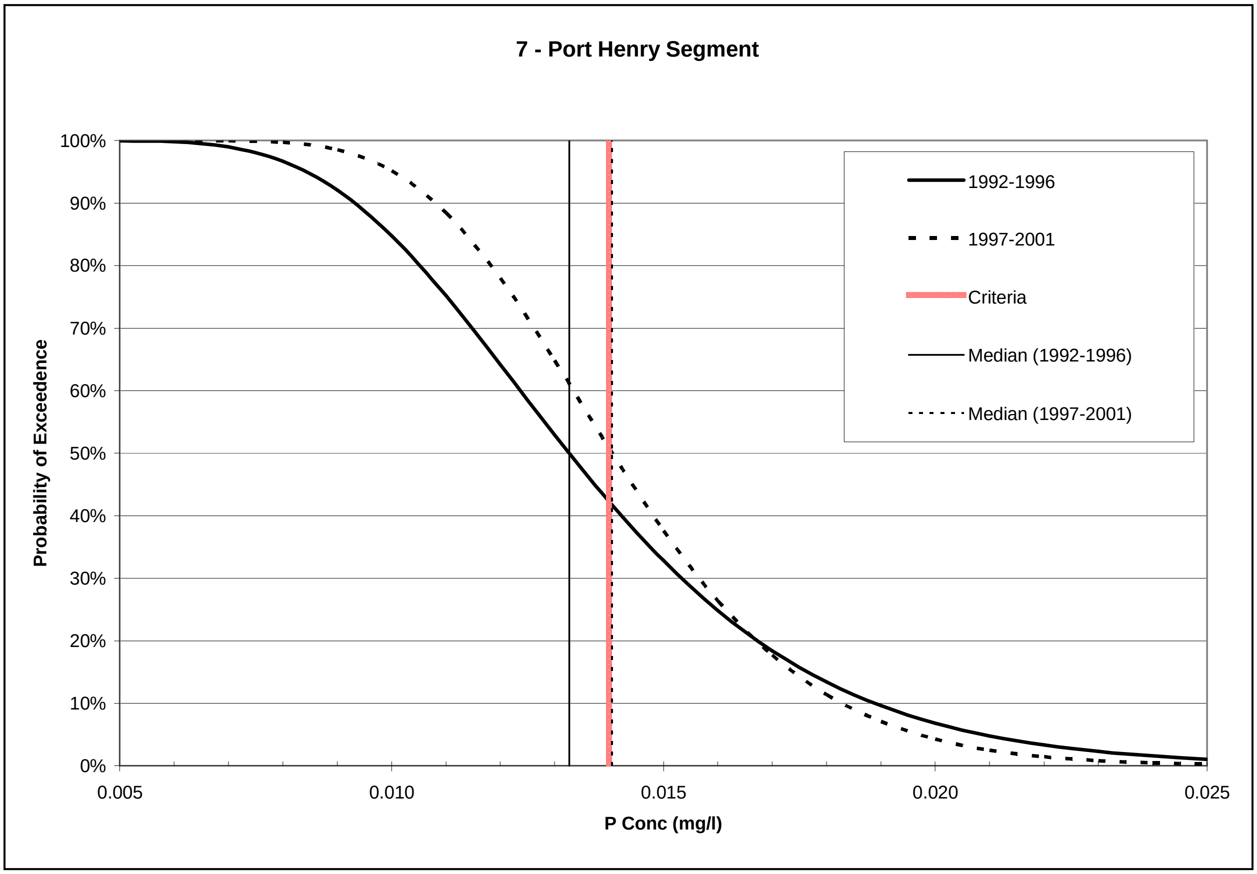



**Figure 52. Complementary cumulative distribution functions of phosphorus concentration in the Port Henry Lake segment: 1992-1995 versus 1996-2002.**

In summary, when we analyzed the data using 1992-1996 as the baseline period and 1997-2001 as the post-treatment period, there is a decreased probability that the standards are being exceeded in the Main Lake, South Lake A, and South Lake B, while the probability of exceedence has increased since 1996 for the Missisquoi Bay and Port Henry segments. It is important to note that this analysis is based on fairly sparse data for a relatively short time period. In addition, we have assumed that lognormal distributions fit the data sufficiently well; more data are needed to test this assumption.

More data are also needed to evaluate spatial variability within lake segments. Although the water quality criteria specify that they apply to "central, open-water areas of each lake segment," and all the monitoring stations are so located, in reality, there are variations from place to place that may not be captured by a single site, especially in large lake segments.

Our presentation of the data provides an alternative for judging trends over time and a more scientifically defensible way of informing decision-making. We recommend that such a statistical approach be commonly used for all quantitative data when sufficient data are collected to make this approach practicable. We fully recognize that regulatory compliance for Lake Champlain will be judged using annual mean concentrations; however he annual means themselves give us just one yes or no answer about compliance for each year. We believe that the definition of a more defensible approach to judgments about trends and significant difference from standard probably deserve additional attention. Unfortunately, the annual means themselves cannot be used in the CCDF approach because there will not be sufficient data to generate a useful distribution for several decades. We will want and need additional information on a much more frequent basis to make management decisions.

In the broadest sense, the purpose of monitoring is to generate data for evaluating the state of the environment. Monitoring data are derived from samples collected discretely over space and time. Therefore, from a statistical perspective, monitoring data are samples from the total population of environmental conditions (Shabman and Smith 2003). Statistical techniques can be used to determine whether the actual state of the environment is within the range of acceptable levels. The use of appropriate statistical techniques can help managers to avoid both false positives (i.e., determining the state is outside the acceptable range when in reality is it not) and false negatives (i.e., determining the state is within the acceptable range when it is not).

Acceptance sampling is hypothesis driven and incorporates both the frequency of samples in violation of the standard, as well as the magnitude of those violations (Shabman and Smith 2003). A mean and probability distribution are used in a t-test to judge whether the sample mean differs from the acceptable level. The concept of a probability distribution is familiar to scientists of all types. Such scientists would never define an environmental variable as a single value, but would define a central tendency, or mean, with an expected distribution around that mean, expressed as a confidence interval or standard error. With such an approach, acceptable levels might be defined as ranges rather than single values. Although such an approach might present challenges in the regulatory arena, where either a characteristic achieves the regulation or does not, we suggest this is a more environmentally realistic and meaningful approach.

**Thresholds**

It can be attractive to think about approaching acceptable levels using the concept of a threshold. For our purposes, a threshold might be envisioned as the breakpoint between an acceptable level of pressure, one to which the ecosystem can adapt, and an unacceptable level of pressure, one that drives the ecosystem to change to an undesired state. In theory, setting the acceptable level a little below (or above) the ecological threshold would maintain the ecosystem in a healthy condition (Schaeffer et al. 1988). The danger in this approach is that frequently, ecological thresholds only become apparent after disastrous shifts in community composition have occurred (Paine et al. 1998, Rapport and Whitford 1999, Scheffer et al. 1993, 2001).

Ecosystems that are in good condition have resilience, they maintain the ability to bounce back or recover from both natural disturbances and some human perturbations, or pressures (Holling 1973). This resilience depends on a variety of self-reinforcing mechanisms, or feedbacks that prevent the ecosystem from shifting to an alternative stable state. It can also be considered the adaptive capacity of the system. There is increasing evidence that ecosystem resilience may be one of the most important characteristics that must be in place in order to sustain the human uses and ecosystem services that a growing population requires (Gunderson and Holling 2002, Elmqvist et al. 2003).

Recently, Gunderson and Holling (2002) proposed an approach to natural resource management that is based on evaluating the types and the timing of management interventions in light of their ability to increase resilience. Walker et al. (2002) built on this approach and propose a four step process for resilience management. By increasing understanding of resilience, the goal is to identify actions that will strengthen the feedback relationships that maintain the desired state of the ecosystem. Although this approach is intriguing and deserving of additional investigation, the scientific understanding to support its application is still quite preliminary.

Using a combination of modeling and data review, Carpenter et al. (1999) showed that cultural eutrophication in some lakes may be reversed by phosphorus input reductions alone, but in other lakes, a threshold had been crossed and the changes may be irreversible if phosphorus reductions are the only management approach taken. If the inputs of phosphorus are stochastic, as they are from nonpoint sources, and there are uncertainties about lake response, there is an increased likelihood that irreversible change will occur without early and aggressive management intervention.

Some useful thresholds have been identified at the level of the ecological component. A recent review shows that phosphorus is accumulating in soils around the world (Bennett et al. 2001). Any factor that increases soil erosion increases the potential for phosphorus runoff to surface waters (Sharpley et al. 1994). In agricultural watersheds, phosphorus frequently accumulates in soils for a number of years, but export can abruptly increase once a threshold level is passed (Heckrath et al. 1995). Through an understanding of soil chemistry, the loss concentration or threshold can be predicted and serve as a basis for defining an acceptable level of phosphorus in soils as an agricultural management target. These concentrations are currently being estimated for Vermont soils (Jokela 2000, personal communication).

The US EPA water quality criteria represent another kind of threshold that can be used as acceptable levels for toxic substances. The criteria represent the concentration at which significant biological impacts begin to occur (US EPA 1986). Although these concentrations are often established based on probabilities (for example, the $LC_{50}$), they also generally represent the concentration that is the threshold between healthy conditions and potential impairment. One problem with these criteria, however, is that they are based primarily on single species laboratory toxicity tests, not exposure under ambient conditions or to a mixture of chemical pollutants (Spehar and Fiandt 1986, Watzin et al. 1994, Masnado et al. 1995). Recently, King and Richardson (2003) have proposed an approach to identifying potential ecological thresholds using assemblage-level attributes and the Index of Biotic Integrity as endpoints to measure responses to numerical changes in water quality. This approach would provide a greater weight of evidence of impairment in the environment because it is based on a multi-species assemblage. Such approaches are much more likely to predict environmentally meaningful concentrations of concern (Watzin et al. 1997).

For living resources, thresholds are most apparent in ecosystems where overfishing has occurred, and the community composition has been dramatically altered (Pauley et al. 1998, Jackson, et al. 2001). In Lake Champlain, a threshold was probably passed when native salmonids were extirpated from the lake in the late 1800s, and the fish community now will likely never be the same. The increase in the sea lamprey population probably also significantly altered the potential stable state (Aron and Smith 1971). Additional research will be necessary to determine what community composition will be stable and resilient given the lake's current trophic state and the suite of exotic species that are now part of the ecosystem. Because exotic species alter resource flows, trophic structure, and, potentially, the disturbance regime, they also have the potential to alter the stable state of a community dramatically (Vitousek 1990).

It is now commonly accepted that biodiversity enhances ecosystem resilience, however, more than one mechanism is likely involved (Schindler 1990, Tilman 1996, Peterson et al. 1998). Redundant species provide multiple pathways for trophic transfer and provide insurance against unexpected disturbances. To the extent that management focuses on one component of the ecosystem at the expense of others, it can reduce adaptive capacity and leave the ecosystem more vulnerable to catastrophic changes.

Ecological economists are also intrigued by ecological thresholds because if they can be predicted, they would be significant factors in environment-economic models (Muradian 2001). Although threshold effects and alternative stable states have been identified in a variety of communities, the ability of ecologists to predict threshold levels is limited, and prices also cannot yet capture the closeness of an ecosystem to transition (Muradian 2001). Although we recognize that managing for resiliency is prudent, we also recognize that the data bases necessary to recognize critical thresholds are not yet available. Therefore, it is probably not practical to use thresholds as acceptable levels for indicators of the state of Lake Champlain in the short term.

## B. Acceptable Levels from a Social Perspective

The social analyses we conducted were based on a survey of Lake Champlain stakeholders. The survey was designed to provide a better understanding of the social values and preferences of these stakeholders so that acceptable levels might be developed based on these perspectives. The survey included two quantitative techniques that were adapted for use in the context of Lake Champlain ecosystem management, namely choice experiments and social norm curves. In this chapter, these techniques and the survey results are discussed.

### Choice Experiments

A choice experiment is a stated preference technique that requires survey respondents to make repeated discrete choices among competing options (Louviere and Timmermans 1990; Boxall, et al. 1996). Choice experiments were first used in economics and marketing to determine consumer preferences for multiattribute goods. The economic theory of random utility is the theoretical foundation of choice experiments. According to this theory, consumers choose to buy products that maximize their utility (an economic measure of satisfaction) and utility is derived from the attributes of the products, not the actual products themselves (Louviere, et al. 2000).

As part of the indicators project, a choice experiment was designed to examine stakeholder preferences for alternative Lake Champlain management scenarios. In order to meet the assumptions of utility maximization, the attributes in the choice experiment must reflect the characteristics of the ecosystem from which stakeholder preferences are derived (Boxall, et al.

1996). For Lake Champlain, there are many ecosystem characteristics that are of known to be of concern to the public. Because the number of characteristics that can reasonably be accommodated in a choice experiment is limited, only ecosystem characteristics that are both relevant to public opinion and responsive to management actions were considered. For example, the spread of zebra mussels, a socially relevant issue, was excluded from the choice experiment because there is currently no technique available for the management of zebra mussels. Given these criteria, a series of potential attributes was developed; each attribute was assigned three levels based upon existing conditions and potential future conditions that could result from management. A prototype choice experiment was designed using these potential attributes.

Two focus group meetings, one in Vermont and one in New York, were held to evaluate the salience of the potential attributes. Focus group participants were asked to describe and evaluate the current condition of Lake Champlain and to specifically discuss the aspects of the ecosystem that are most important to them. They were then asked to discuss human activities impacting the lake and what they thought should be done to improve the condition of the lake. Following this general discussion, participants completed and discussed the prototype choice experiment. Participants were asked about the relevance and comprehensibility of the attributes in the choice sets, the terminology used and the appropriateness of the range of levels described. They were also asked to consider how many attributes they could reasonably trade-off and how many paired comparison choices they could make in one sitting. Potential attributes were refined following each of the focus group meetings and through discussions with professionals familiar with the specific attributes.

The five ecosystem attributes used in the choice experiment were algae blooms, public beach closures, fish consumption advisories, basin land use distribution and spread of water chestnut. As discussed previously, algae blooms interfere with recreational use and enjoyment of Lake Champlain and are attributed to elevated phosphorus concentrations (Smeltzer and Quinn 1996). Phosphorus reduction has remained amongst the highest priorities of the LCBP and with 80% of the current phosphorus load coming from nonpoint sources (LCSC 2003), substantial management resources will have to be directed towards phosphorus to attain the load reductions necessary to achieve the in-lake criteria. As with algae blooms, public beach closure is a problem affecting human use and enjoyment of Lake Champlain, and this problem is also a high priority for management.

Safe fish consumption was emphasized as an important concern during focus group discussions. PCBs and mercury are the major bioaccumulating toxins triggering fish consumption advisories for Lake Champlain (LCSC 2003). Although the advisories issued by the Health Departments of the State of New York and the State of Vermont for Lake Champlain differ, both agree that walleye and lake trout are the species posing the greatest risk and that everyone, especially children and women of childbearing age, should limit their consumption of fish, particularly walleye and lake trout, from Lake Champlain (NYSDOH 2002; VTFWD 2002).

Basin land use distribution was included in the choice experiment because it is a pressure related to many Lake Champlain ecosystem characteristics of concern. Agricultural land, approximately 16% of the Lake Champlain Basin, contributes phosphorus, bacteria and pesticide waste to Lake Champlain (Hegman et al. 1999). Urban and suburban land, at least 6% of the basin and growing, contributes more phosphorus per acre than agricultural land and is also a source of bacteria and toxin-laden stormwater (Budd and Meals 1994).

Water chestnut is the one major nonnative nuisance species that can be managed in Lake Champlain. It is primarily found in the South Lake, but historically, has moved northward from Whitehall, NY. Twenty years of data show that the spread of water chestnut can be adequately controlled given sufficient management expenditure for harvesting (Hunt and Crawford 2002). The spread of water chestnut was included in the choice experiment to examine whether respondents prefer water chestnut management more or less than other potential uses of limited management funds.

Sea lamprey control is a major focus of fisheries management efforts on Lake Champlain (Lake Champlain Fish and Wildlife Management Cooperative Fisheries Technical Committee 1999). It attaches to and preys primarily upon salmonids such as lake trout and landlocked Atlantic salmon causing unsightly sores, reduced growth and mortality of these fish. At this time, sea lamprey are primarily managed with a chemical lampricide, known as TFM, which is applied in lamprey spawning streams in the Champlain Basin. The use of TFM is controversial because of potential impacts to non-target species.

Stormwater is a source of water pollution for Lake Champlain and its tributaries. While stormwater discharge permits are required of all new development, many stream reaches are already impaired by development that pre-dates the permitting system or current have developments with stormwater BMPs that are not in compliance with their original permits (VTDEC 2002). Due to the variety of pollutants found in stormwater, streams impaired by stormwater pose a threat to Lake Champlain. As forest, other natural land types and agricultural land are converted into urban and suburban land cover, the likelihood of stream impairment resulting from stormwater runoff will increase.

Lake Champlain is a popular recreational site for both basin residents and tourists and generated an estimated $3.8 billion from tourism in 1998-99 (LCSC 2003). Developing and enhancing public access to Lake Champlain for diverse recreational uses is a high priority of the LCBP. Public access with minimal congestion and user conflict is an important element of sustainable tourism and resident satisfaction with Lake Champlain.

The five ecosystem attributes used in the choice experiment, algae blooms, public beach closures, fish consumption advisories, basin land use distribution and spread of water chestnut, were then varied systematically. Three levels were defined for each attribute based on existing condition and potential future conditions using responses from the focus group discussions. The attribute levels were arrayed into paired comparisons following an orthogonal fractional factorial design that allows for independent estimation of all main effects and two-way interactions. The resulting choice experiment consisted of five blocks of nine paired comparisons. Each block corresponds to a different version of the mail survey questionnaire.

Respondents were instructed to examine and weigh the levels of the five attributes in two profiles, Lake Champlain A and Lake Champlain B, then choose the profile they preferred. An example question with instructions was followed by nine paired comparisons (Figure 53) in each survey. The choice task presented in each paired comparison was likened to the tradeoffs that managers have to make when prioritizing management activities and allocating management resources.

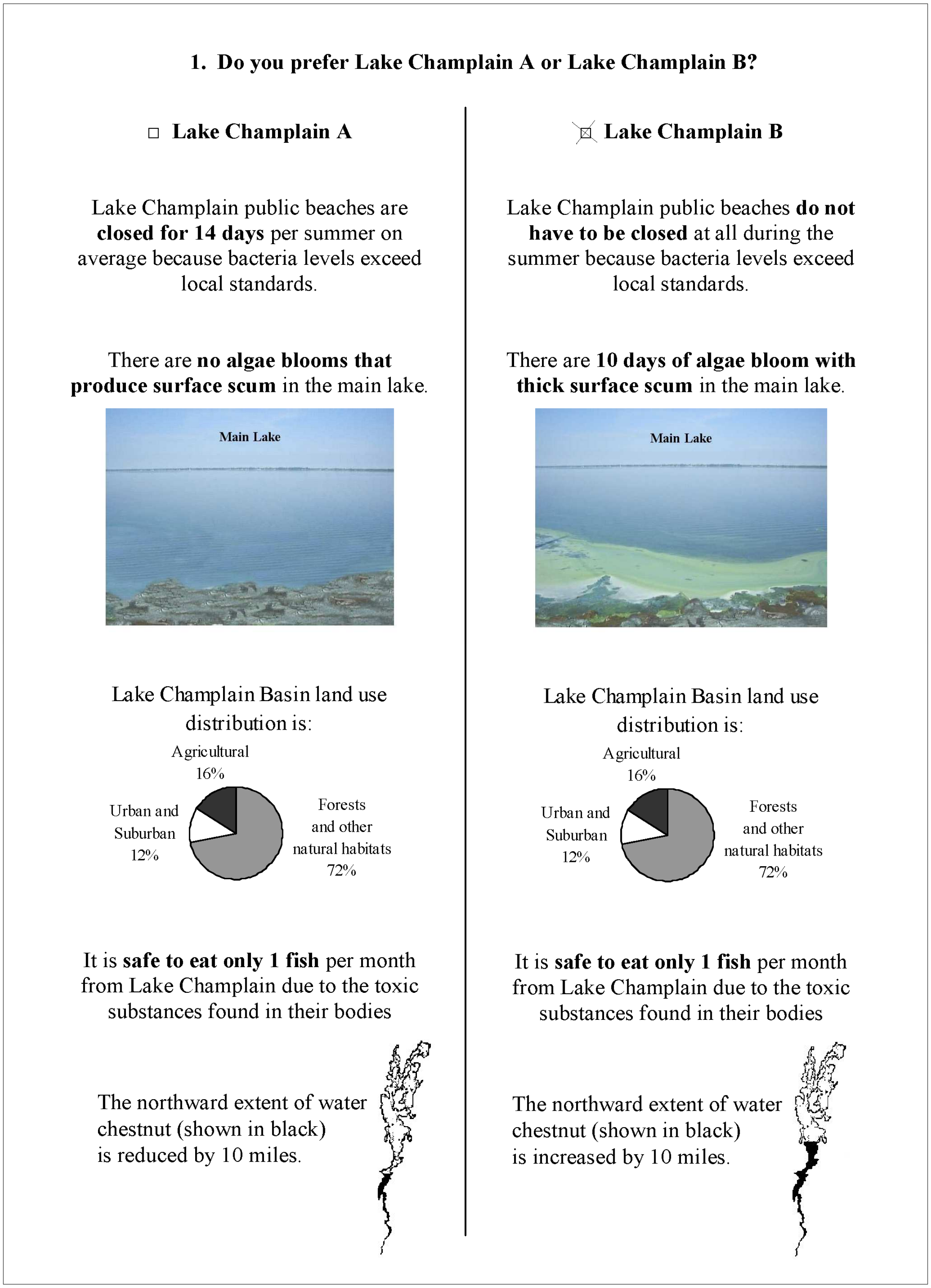



**Figure 53: Example choice experiment question**

Respondents had to trade-off two, three or four attributes for each comparison. The questions were ordered such that the simpler, two attribute trade-off questions were first, followed by the more complex tradeoffs. Respondents were instructed to make a choice even if they did not "like" either profile in the paired comparison; they were not given a no-choice option. The choice questions were followed by questions to evaluate the amount of information provided and the difficulty of the choice task.

**Measuring Social Norms**

Choice experiments measure preferences for multiple ecosystem characteristics simultaneously. Although this technique provides a context that is more representative of real life decision-making, there is a limitation in the number of discrete levels that can be examined because of the statistical complexity in the choice experiment design. In park and wilderness management research, social norms curves are used to examine preferences for a larger range of indicators of quality to help define standards of quality (Manning and Lawson 2002). This technique follows an approach for defining social norms developed by Jackson (1965). Individual norms are measured with questions that ask respondents to rate a range of potential conditions on a scale representing acceptability, satisfaction, tolerance or other evaluative dimension (Manning and Lawson 2002).

Social norm curves were estimated for the five characteristics of Lake Champlain included in the choice experiment and for three additional characteristics. These additional characteristics were number of sea lamprey wounds per fish caught, stream impairment by stormwater (Figure 54), and public access. Two norm curve questions were included in each version of the questionnaire.

22. Urbanization in the Lake Champlain Basin can have negative impacts on streams which discharge into Lake Champlain. Currently, there are at least 27 streams in the Lake Champlain Basin impaired by stormwater. We would like to know what you think is an acceptable number of impaired streams. Please rate **each** of the number of impaired streams in the table below by circling a number on the scale from -3 to +3. A rating of -3 means that the number of impaired streams is very unacceptable and a rating of +3 means that the number of impaired streams is very acceptable.

| | Very Unacceptable | | | | | | Very Acceptable |
|---|---|---|---|---|---|---|---|
| 0 impaired streams | -3 | -2 | -1 | 0 | +1 | +2 | +3 |
| 9 impaired streams | -3 | -2 | -1 | 0 | +1 | +2 | +3 |
| 18 impaired streams | -3 | -2 | -1 | 0 | +1 | +2 | +3 |
| 27 impaired streams | -3 | -2 | -1 | 0 | +1 | +2 | +3 |
| 36 impaired streams | -3 | -2 | -1 | 0 | +1 | +2 | +3 |
| 45 impaired streams | -3 | -2 | -1 | 0 | +1 | +2 | +3 |

**Figure 54: Example social norm curve question**

For each question, respondents were asked to rate the acceptability of a range of potential conditions represented by five or six discrete levels of each of these characteristics on an ordinal scale from -3 to +3, where -3 is very unacceptable, zero is neutral and +3 is very acceptable. A mean acceptability rating and standard deviation was calculated for each level of the characteristics by averaging all the individual ratings for that level. Aggregated, social norm curves were created by plotting these mean acceptability ratings for the ranges of discrete levels for the eight characteristics.

**Survey Implementation**

For a choice experiment to be valid, respondents must have preferences for the characteristics that make up the choice profiles. Therefore, survey respondents for this Lake Champlain study should have some prior knowledge of the ecosystem in order to understand the differences between the attribute levels so that they could make meaningful choices. The population of interest was the stakeholders of the Lake Champlain ecosystem, residents of the basin that have knowledge of and interest in Lake Champlain. However, this is not a readily accessible population for survey purposes. The LCBP is the central authority responsible for coordinating management, education and public outreach for Lake Champlain. A random sample of the mailing list of the LCBP was used as the study population. The LCBP appeals to a broad spectrum of Lake Champlain stakeholders, making it a better surrogate population than a random sample of basin residents, or more specialized subpopulations such as basin residents with fishing licenses, members of paddling clubs or marina members.

The problem of identifying an appropriate study population has been encountered in other choice experiments concerning environmental or recreational amenities. Boxall and Macnab (2000) conducted a choice experiment to examine preferences of hunters and wildlife viewers for forest management scenarios. The hunter population was identified through hunting licenses and the members of a provincial natural history organization, Nature Saskatchewan, were used as a substitute for wildlife viewers. Bullock, et al. (1998) used a choice experiment to determine the value of characteristics of the hunting experience in Scotland. Because there is no licensing of hunters in the United Kingdom, questionnaires were mailed to associates of 38 different organizations including sporting estates, associations and agencies (Bullock, et al. 1998).

The questionnaire design and multiple contact survey implementation followed procedures recommended by Dillman (2000). Fifty questionnaires, 10 of each version, were mailed out to a random sample of the survey population in a pilot mail survey. A final questionnaire revision followed the pilot mailing. The questionnaires were sent out in August of 2002. Postcard reminders were sent to non-respondents three weeks after the questionnaire mailing and replacement questionnaires were sent to approximately 80% of the non-respondents four weeks after the postcard reminder.

The choice experiment questions were analyzed using binary logistic regression in the Statistical Analysis System, version 8.2 (SAS Institute Inc. 2000). The choice of profile A or B was the response variable. The explanatory variables were derived from the choice profiles and coded using effects coding. The effects coded variables for profile B were subtracted from the effects coded variables for profile A to yield ten difference variables. These were the explanatory variables in the main effects model. The model was estimated with the first level excluded and then with the third level excluded in order to determine a utility coefficient, Wald Chi-Square statistic and p-value for each attribute level. The main effects model was also estimated using reference coding to determine if the differences between the utility coefficients for the attribute levels were statistically significant.

Additional models were estimated to examine differences in preferences across subpopulations. Evidence from focus group meetings suggested that preferences differ across the subregions of the Lake Champlain Basin. Choice observations from New York respondents and from Vermont respondents were used to estimate separate main effects models for the two states. There was not a sufficient sample size to independently estimate preferences for respondents from Quebec. Following the approach used by Opaluch, et al. (1993) and Johnston, et al. (2002) to examine variability of preferences across socioeconomic groups, a third model was estimated by interacting a dichotomous variable representing state of residence with each of the explanatory attribute difference variables. The separate New York and Vermont models show the statistical significance of the attribute levels for each subpopulation and the model with state interaction shows whether the differences in the coefficients for New York and Vermont are statistically significant. Similarly, variations on the main effects choice model were constructed to test the hypothesis that the water chestnut attribute would be a stronger predictor of profile choice for respondents living near the infestation in the south lake. Choice observations were categorized as north or south by zip code. Separate main effects models were estimated for northern respondents and southern respondents and a model was estimated with interactions between a dichotomous north/south variable and the water chestnut difference variables.

During focus groups, some participants expressed concern that they were unable to clearly and directly convey what ecosystem characteristics were most important to them because they had to make tradeoffs. In the final questionnaire, the choice questions were followed by questions that asked the respondent to rate how important (very important, important, somewhat important or not important) the five characteristics in the choice questions and four additional ecosystem characteristics were to them. Relative frequencies of the categorical judgments for the nine ecosystem characteristics were tabulated. To test the hypothesis that preferences were not discernible when characteristics were evaluated individually, the cumulative frequencies of the categorical judgments were normally transformed (Torgerson 1958; Powers and Xie 2000) and scaled along successive intervals representing the four categories (Torgerson 1958).

## RESULTS AND DISCUSSION

### Characteristics of the Sample Population

The final response rate was 41.6% (766 returned questionnaires), yielding 6548 usable responses for the choice experiment. Although the sample was drawn from a mailing list, the sample is diverse with respect to the demographic information obtained from survey respondents (Table 23). Compared to 2000 U.S. census data, however, the survey respondents were older and more educated than the population of Vermont on average.

**Table 23: Summary of sample demographic characteristics (percent of the populations showing various characteristics).**

| | | | | | | |
|---|---|---|---|---|---|---|
| **Age** | | | | | | |
| < 25 | 25 - 35 | 36 - 50 | > 50 | | No Response | Total |
| 1.2% | 5.7% | 32.4% | 56.3% | | 4.4% | 100% |
| **Education** | | | | | | |
| < High School | High School | Some College | College | Graduate School | No Response | Total |
| 0.9% | 5.1% | 9.5% | 30.9% | 49.0% | 4.6% | 100% |
| **Occupation** | | | | | | |
| Business/ Technical | Retired | Educator | Gov't/ Nonprofit | Other | No Response | Total |
| 32.6% | 19.5% | 17.6% | 11% | 11.6% | 7.7% | 100% |

The average number of visits to Lake Champlain during summer by survey respondents was nine per month. When asked how the lake has changed over the last five years, 41.4% reported that the condition of the lake has declined over the last 5 years, 29.2% claimed it has stayed the same, 19.7% reported improved lake condition and 9.7% did not answer the question. It is plausible that individuals that participate in different lake-oriented recreational activities have different preferences and expectations regarding lake condition. The sample population included participants in all major lake recreational activities (Figure 55). Therefore, there is no reason to suspect a systematic bias in the results due to exclusion of a particular recreational user group.

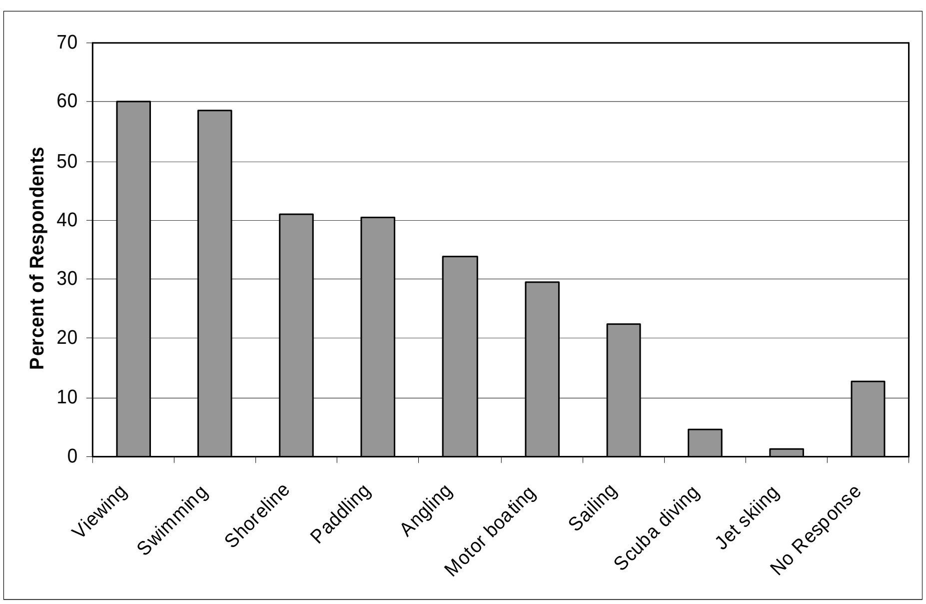



**Figure 55: Percent of respondents that participate in lake-oriented recreational activities.**

## Choice Experiment Analysis

The main effects model was highly significant (likelihood ratio = 1905.25, $p<0.001$). Because the model was estimated with effects coded variables, the coefficients are estimates of the utility associated with the ecosystem attribute levels (Table 24). All attribute levels except two were significant ($p < 0.05$). All the coefficients estimated with reference coded explanatory variables were also highly significant ($p<0.001$). This means that the differences in utility between the levels of all the attributes are statistically significant.

For the beach closure, algae bloom, fish consumption and water chestnut attributes, the degree of ecosystem impact or degradation increases from the first to the third level. Given that relationship, the signs of the coefficients for the attribute levels are as expected. The utility coefficients for the first levels are positive indicating that these levels add to satisfaction with the condition of the lake. The negative utility coefficients for the third levels indicate that these levels detract from satisfaction with the condition of the lake. For the second levels, some coefficients are positive and others are negative but all are close to zero because these middle levels do not provide as much utility or disutility as the first and third levels.

**Table 24: Results of the main effects choice model analysis.**

| Attribute | Level | Utility | Attribute | Level |
|---|---|---|---|---|
| Beaches not closed | 1 | 0.589 | 295.03 | <0.0001 |
| Beaches closed 7 days on average | 2 | -0.075 | 6.33 | 0.0119 |
| Beaches closed 14 days on average | 3 | -0.514 | 174.36 | <0.0001 |
| No algae blooms that produce surface scum | 1 | 0.573 | 267.02 | <0.0001 |
| 10 days of algae bloom with some surface scum | 2 | 0.006 | 0.04 | 0.8377 |
| 10 days of algae bloom with thick surface scum | 3 | -0.579 | 333.89 | <0.0001 |
| Current land use distribution | 1 | 0.288 | 76.72 | <0.0001 |
| Increase urban/suburban decrease agricultural | 2 | -0.265 | 85.08 | <0.0001 |
| Increase urban/suburban, decrease natural | 3 | -0.023 | 0.40 | 0.5249 |
| Safe to eat unlimited fish | 1 | 0.907 | 559.87 | <0.0001 |
| Safe to eat 1 fish/month | 2 | 0.074 | 5.38 | 0.0203 |
| Not safe to eat fish | 3 | -0.981 | 700.64 | <0.0001 |
| Extent of water chestnut reduced by 10 miles | 1 | 0.538 | 261.61 | <0.0001 |
| Extent of water chestnut not changed | 2 | 0.071 | 0.03 | <0.0001 |
| Extent of water chestnut increases by 10 miles | 3 | -0.609 | 286.31 | <0.0001 |

The Wald Chi-Square statistic is a measure of the strength of the individual predictors. In general, the first and third levels of the attributes are highly significant and strong predictors of choice. The middle levels, with smaller Wald statistics and larger p-values, are weaker predictors. As expected, these results indicate that respondents' choices between profiles tended to be driven by the attributes at their extreme levels. The third and first levels of the safe fish consumption attribute are the two strongest predictors of choice. This implies that safe fish consumption is the most important of the five attributes in the choice experiment. Based on these results, management actions to reduce toxic substances in the water and consequently in the tissue of fish should be prioritized above management actions addressing the other ecosystem characteristics included in this choice experiment.

Algae blooms, beach closures and the spread of water chestnut attributes are of similar significance and prediction strength. The land use attribute is a weaker predictor of choice. Although land use is an important pressure on the lake ecosystem, it is not a characteristic directly associated with lake use and enjoyment. Therefore, it may be the least important predictor because it is the most abstract and least directly relevant to respondents. Furthermore, the coefficient signs and Wald statistics for the land use distribution attribute vary from the pattern of the other attributes because the relationship between the levels of the land use attribute is not ordinal. Level one, the current land use distribution, has a positive and statistically significant coefficient and the second level, converting agricultural land into urban or suburban land, has a negative and statistically significant coefficient of approximately the same magnitude. The third level of the land use attribute, converting natural land cover to urban/suburban land cover, has a negative coefficient but it is not a statistically significant predictor of profile choice. Utility coefficients for the levels of this attribute suggest that respondents prefer less land development and that preserving the agricultural landscape is more important than preserving the natural landscape.

The Vermont choice model and New York choice model yielded separate utility estimates for respondents from the two states and the interaction model tested for statistically significant differences in utility estimates between the two states (Table 25). All three models are highly significant ($p<0.0001$). There are fewer choice observations available to estimate the New York model ($n=758$) compared to the Vermont model ($n=5710$). Therefore, the Wald statistics are smaller and more of the attribute levels are insignificant for the New York model.

**Table 25: Vermont and New York choice model results.**

| **Attribute** | Vermont | | New York | |
|---|---|---|---|---|
| | Utility Coefficient | Wald Statistic | Utility Coefficient | Wald Statistic |
| Beaches not closed | 0.622*† | 279.48† | 0.428*† | 19.18 |
| Beaches closed 7 days on average | -0.082* | 6.50 | -0.054 | 0.39 |
| Beaches closed 14 days on average | -0.540* | 162.67 | -0.374* | 11.87 |
| No algae blooms that produce surface scum | 0.582* | 235.64 | 0.633* | 38.20 |
| 10 days of algae bloom with some surface scum | 0.008 | 0.07 | 0.014 | 0.03 |
| 10 days of algae bloom with thick surface scum | -0.589* | 296.70 | -0.646* | 47.76 |
| Current land use distribution | 0.309* | 75.39 | 0.271* | 8.11 |
| Increase urban/suburban decrease agricultural | -0.261* | 71.68 | -0.302* | 13.00 |

| | | | | |
|---|---|---|---|---|
| Increase urban/suburban, decrease natural | -0.048 | 1.42 | 0.031 | 0.09 |
| Safe to eat unlimited fish | 0.932* | 499.62 | 0.837* | 60.75 |
| Safe to eat 1 fish/month | 0.077* | 4.96 | 0.020 | 0.05 |
| Not safe to eat fish | -1.009* | 631.17 | -0.856* | 66.28 |
| Extent of water chestnut reduced by 10 miles | 0.543* | 230.99 | 0.571* | 32.90 |
| Extent of water chestnut not changed | 0.072* | 4.42 | 0.026 | 0.08 |
| Extent of water chestnut increases by 10 miles | -0.616* | 252.09 | -0.596* | 32.38 |

* Statistically significant at 0.05 level or better
† Difference in VT and NY coefficients significant at 0.1 level

The relative predictor strength of the attributes differs somewhat between New York respondents and Vermont respondents. Because Vermont respondents are a majority of the total observations, the results of the Vermont model are similar to the combined model. In both subpopulations, as with the combined main effects model, safe fish consumption is the strongest predictor. For New York, the relative importance of the other attributes is more straightforward than for the combined or Vermont model. After safe fish consumption, the first and third algae bloom levels are the next strongest predictors followed by water chestnut levels. Compared to the combined and Vermont model, the prediction strength of the beach closure attribute in the New York model is weaker. It is more similar in strength to the land use attribute than to algae blooms and water chestnut.

Although attribute prediction strength varies between the separate main effects models, most of the differences in preferences between New York respondents and Vermont respondents are not significant. From the interaction model, only the coefficient representing the difference in utility between New York and Vermont for zero days of public beach closure is marginally significant (p=0.0631). This difference suggests that this attribute is somewhat more important in Vermont than in New York.

Another difference between the New York and Vermont models emerged with the land use attribute. In Vermont, maintaining the current land use distribution is a stronger predictor than loss of agricultural land to urban land. In the New York model, losing agricultural land is the stronger predictor. The third level of land use distribution, losing natural land cover to urban land cover is negative in the Vermont and combined models but positive in the New York model. This is the only attribute level for which the signs of the coefficients are different between the two state models. Although the third level of the land use attribute is not significant in any of the choice models, this potential difference in preference for land use between Vermonters and New Yorkers warrants further investigation.

The results of the models comparing residents that live near water chestnut (south) to those that do not (north) show that the first and third levels of the water chestnut attribute are relatively stronger predictors in the southern model compared to the northern model (based on Wald statistics). However, the interaction variables are insignificant ($p>0.45$), Therefore, these differences in the utilities estimated with the state models are not statistically significant.

As discussed by Louviere, et al. (2000), choice experiments place greater cognitive burden on respondents than more conventional question formats. Forty-nine percent of the respondents to the choice questions claimed that the questions were "difficult" or "very difficult" and thirty-three percent indicated that additional information would have helped them to make more meaningful choices. While choice experiment questions are more challenging for respondents, it is difficult to estimate preferences, and consequently prioritize ecosystem management activities, with data from simple, single characteristic evaluations. The relative frequencies of responses to the questions individually evaluating ecosystem characteristics indicate that respondents tended to report that all characteristics were "important" or "very important" to them (Figure 56). There were very few "not important" responses. Examination of the relative frequencies and the results of the successive interval scaling suggest that zebra mussels and public access are the most important and beach closure and safe fish consumption are the least important of the nine ecosystem characteristics considered. However, normal score transformation places all nine ecosystem characteristics within the "important" range of the categorical scale. This suggests that the ability to scale the relative importance of ecosystem characteristics with this type of questioning is limited. Although simple categorical judgment questions are easier for respondents to answer, the information gained is of little practical value compared to choice experiment questions.

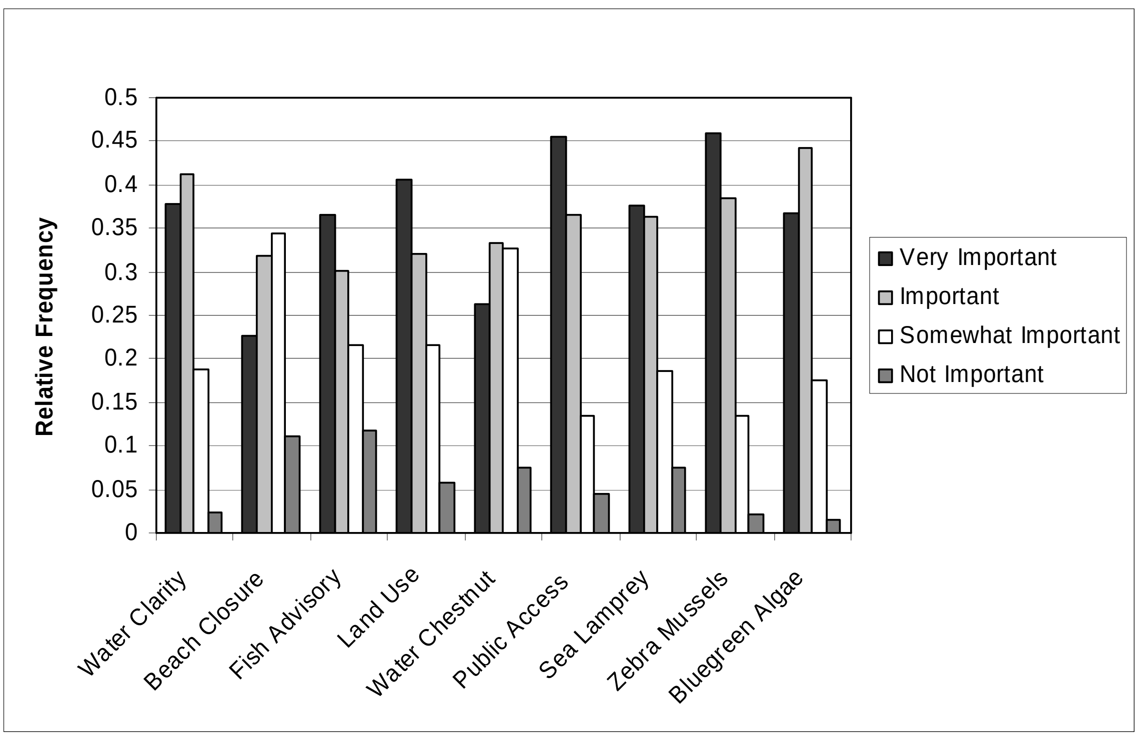



**Figure 56: Importance of individual ecosystem characteristics.**

## Social Norm Curves

The social norm curves trace the average acceptability ratings of the sample population across the range of levels of the eight ecosystem characteristics examined (Figure 57 a-h). All levels for which the y value is greater than zero are considered acceptable with the highest point of the curve along the y-axis representing the optimal condition. The point at which the curve crosses the zero point of the acceptability scale represents the minimum acceptable condition (Manning 1999, Manning and Lawson 2002). The spread of the curve about the y-axis is a measure of norm intensity or salience of the characteristic and the variability of responses about the mean, called crystallization, is a measure of consensus about the norm amongst the group sampled (Manning and Lawson 2002).

Overall, the direction of the slopes of the norm curves is as expected. The variability of responses about the mean acceptability ratings tends to be largest around the minimum acceptable condition and smallest for levels furthest away from the minimum acceptable condition. This signifies less consensus about where the condition shifts from minimally acceptable to minimally unacceptable than there is regarding the acceptability of the extreme values in the range of the conditions presented.

Public beach closure is the norm curve with the greatest salience, as indicated by widest range of y values (Figure 57a). This curve shows that 7 or more days of public beach closure on average during the summer is unacceptable. During the summer of 2002, Blanchard Beach in Burlington, VT was closed for the entire swimming season, clearly in violation of this measure of acceptability. The average acceptability ratings on this curve correspond well with utility estimates from the choice experiment. Algae blooms, which also impact recreational use and enjoyment along the shore of Lake Champlain, show a similar norm curve pattern (Figure 57b). Although the questions appeared in different questionnaire versions, the shift from acceptable to unacceptable occurs at approximately seven days for both public beach closure and algae blooms.

Safe fish consumption, the most important attribute in the choice experiment, shows less salience than some of the other characteristics examined with social norm curves (Figure 57c). However, the mean acceptability ratings correspond relatively well with the utility estimates. Choice experiment attribute levels were modeled after existing consumption advisories. Therefore, the number of fish safely consumed was varied only between none, one and unlimited fish. The current advisory suggests that adults limit their consumption of lake trout and walleye to one meal per month. The advisories are less restrictive with regards to other fish species. Nevertheless, there is little tolerance for limitations in safe fish consumption based on the social norm curve.

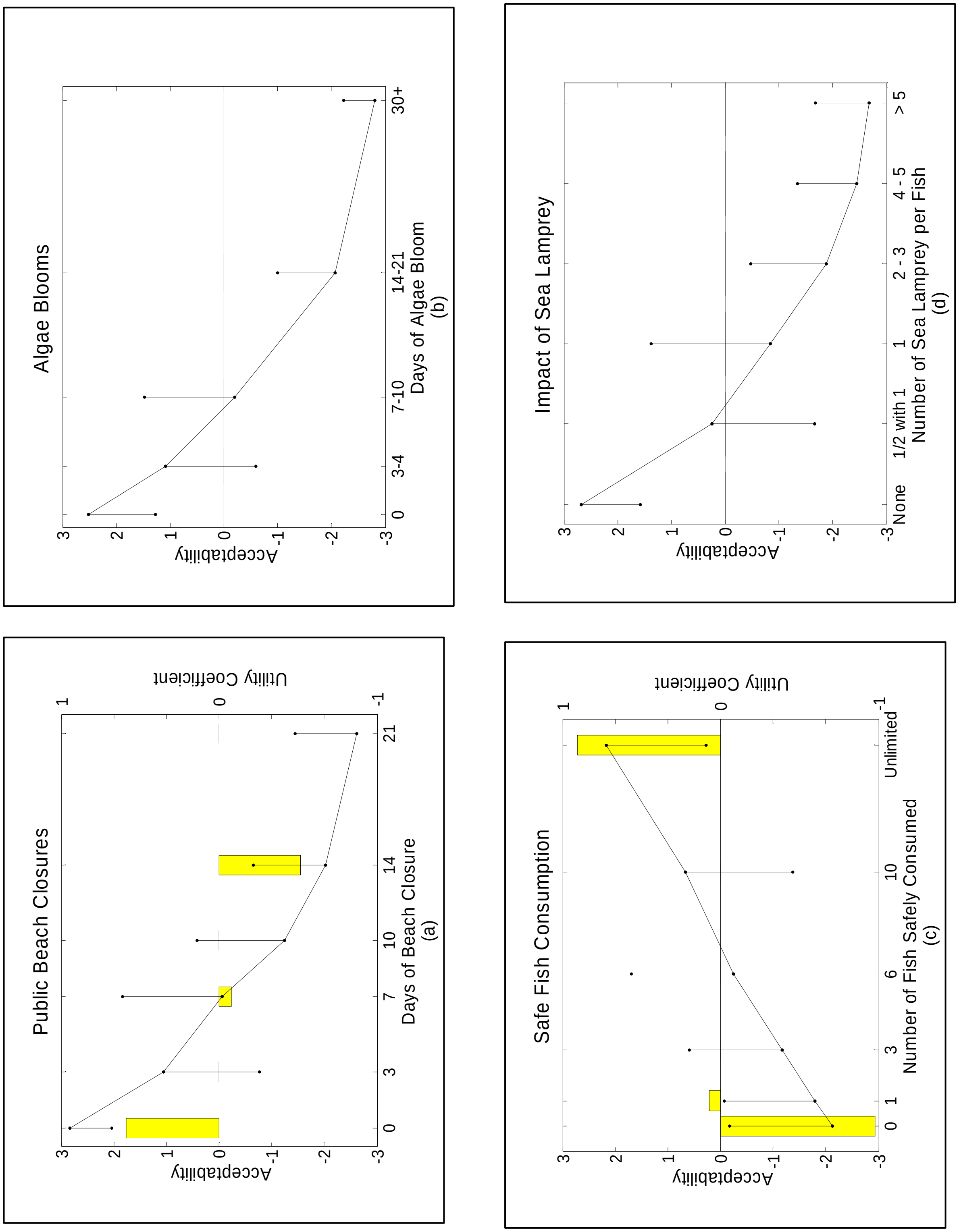



**Figure 57 a-d: Social norm curves for characteristics of Lake Champlain. Social norm curves are shown with one-sided standard deviations of the mean acceptability ratings. Bars represent standard deviation. Where applicable, utility coefficients estimated in the choice experiment are shown as bars on a secondary y-axis.**

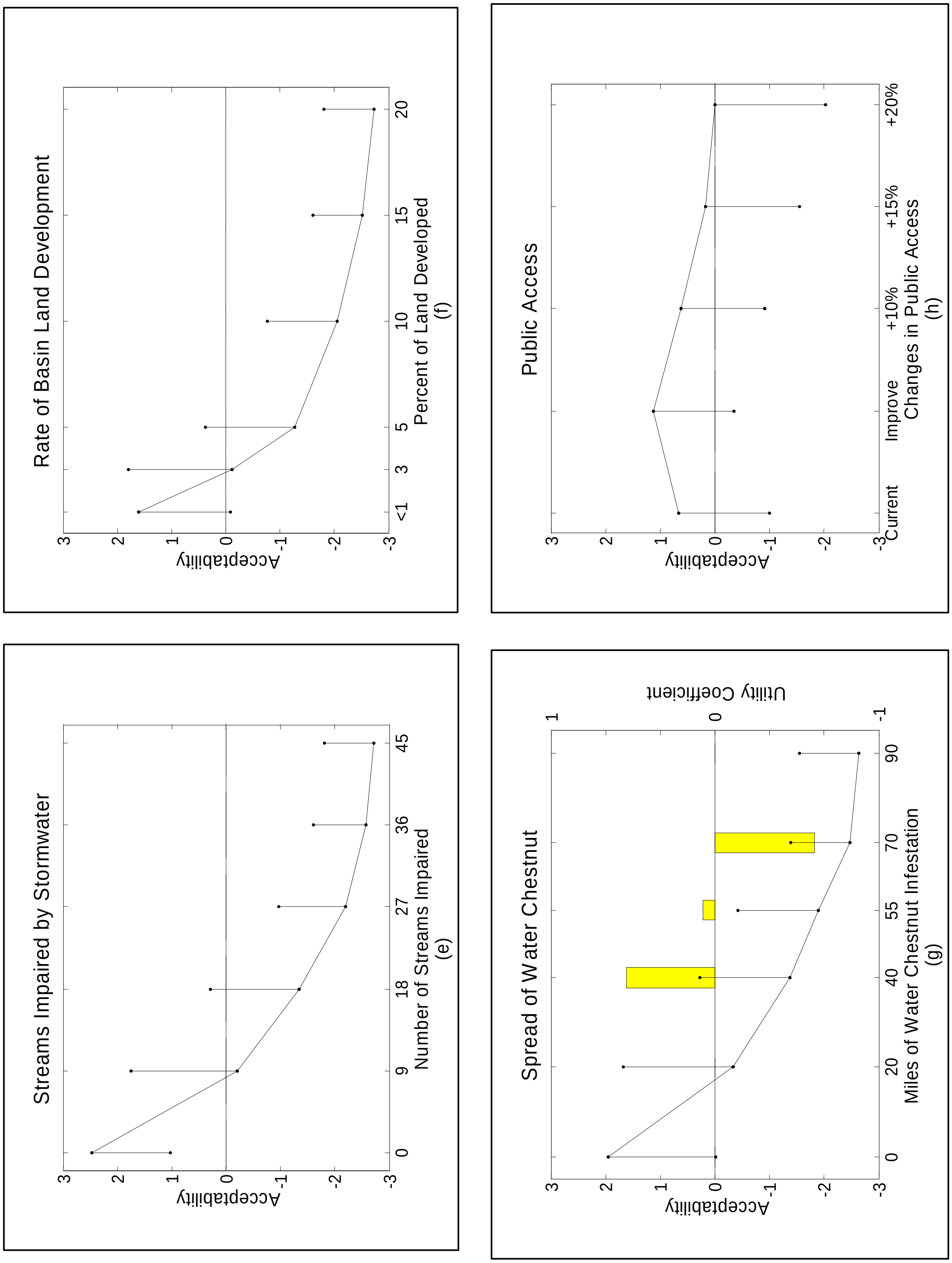



**Figure 57 e-h: Social norm curves for characteristics of Lake Champlain.**
**Social norm curves are shown with one-sided standard deviations of the mean acceptability ratings. Bars represent standard deviation. Where applicable, utility coefficients estimated in the choice experiment are shown as bars on a secondary y-axis.**

The impact of sea lamprey on sport fish, a characteristic not included in the choice experiment, is also highly salient (Figure 57d). Having no sea lamprey wounds on caught fish is a much preferred condition, with the curve dropping below zero between the second and third levels, one lamprey on half to one lamprey on most of the fish caught from Lake Champlain. Because the number of sea lamprey wounds varies considerably by fish species, fish size, and lake segment (Fisheries Technical Committee 1999), evaluating the current condition relative to this estimated range of acceptability will require further analysis. Overall, the current impact of sea lamprey on salmonid species is not acceptable based on this measured norm curve.

The acceptability ratings for the number of streams impaired by stormwater is another highly salient characteristic (Figure 57e). The questionnaire stated that there were at least 27 streams impaired by stormwater in the Champlain Basin. This is well beyond the minimum acceptable condition based on this measured norm curve. For this curve, the mean ratings for all levels except the first are in the unacceptable range. Crystallization around the means increases steadily for each level beyond nine impaired streams. At 27 impaired streams, the current condition, the standard deviation is well below the minimum acceptable condition. This suggests that stormwater impairment of streams is an area requiring more management attention.

The social norm curves for land development (Figure 57f) and stream impairment (Figure 57e) have a similar pattern with decreasing variability about the means in the unacceptable region. The levels of land development used to estimate the social norm curve varied from the land use distribution choice experiment attribute levels. In the choice experiment, the first attribute level was based on the most current land use distribution estimates, 6% developed, 16% agricultural and 78% natural. The second level has 12% developed and 10% agricultural and the third has 12% developed with 72% natural. The current land use distribution has positive utility and the other two levels where developed land increases have negative utilities. To measure the social norm curve, respondents were told that there are at least 300,000 acres of developed land in the Champlain Basin and then asked to rate the acceptability of a range of increases in developed land per year. In both the choice experiment and the social norm curve, increasing developed land is not favorable. Rates of land development vary across the basin. While growth in Chittenden County, Vermont may be outside the range of acceptability, rates of land development are lower in other areas of the basin, particularly in New York.

The mean acceptability ratings for the spread of water chestnut also show increasing crystallization for increasingly unacceptable conditions (Figure 57g). As with impaired streams, the current condition, 55 miles of northward spread, is highly unacceptable. However, the utility coefficients for the water chestnut attribute do not correspond with this curve as they did with the other attributes examined by both types of questions. The water chestnut attribute covered only the middle part of the range used in the acceptability rating question and the levels were described differently. In the choice experiment, the water chestnut attribute varied from reduce the extent by 10 miles (equivalent to 45 miles north), remain unchanged (equivalent to 55 miles north) and increase by 10 miles (equivalent to 65 miles north) and the levels used in the rating question varied from 0-90 miles. The positive utility associated with water chestnut extending 45 miles from Whitehall is more likely capturing satisfaction with management to reduce water chestnut rather than satisfaction with actual existence of water chestnut 45 miles from the southern end of the lake. This finding illustrates how preferences or acceptability can be influenced by the range of conditions presented to respondents in these types of questions.

The final norm curve is for public access (Figure 57h). There is minimal spread of the mean acceptability ratings about the y-axis for the levels of public access used in this question indicating that public access is less salient than the other characteristics. This curve shows that the sample population preferred management actions to improve existing public access over the creation of additional access areas. The LCBP has been active in both improving existing access and developing additional access sites.

**Management Implications**

Utility coefficients estimated with the choice experiment and acceptability ratings from the social norm curves provide a basis for defining acceptable levels for ecosystem indicators from a social perspective. This information will be used in conjunction with ecological data analysis and our best scientific understanding to set the initial acceptable levels for the Lake Champlain ecosystem indicators.

The survey results also provide some insights regarding social preference and Lake Champlain ecosystem management priorities. Reducing phosphorus inputs to Lake Champlain is a major focus of the LCBP. Although survey results show that water clarity and algae blooms are important to Lake Champlain stakeholders, there are other characteristics of the ecosystem of equal or potentially greater importance.

Survey results indicate that safe fish consumption, an attribute added to the choice experiment in response to focus group discussions, warrants more management attention. Safe fish consumption is not just a concern of the angling community. Focus group discussions, written survey comments and the strength of the safe fish consumption attribute in the choice experiment suggest that people are making a connection between the toxicity of fish and the overall health of the lake. A choice experiment examining preferences for alternative natural resource enhancement programs in Green Bay, Wisconsin yielded a similar result. Reduction in the number of years until fish and wildlife in the bay was safe from PCBs was preferred over the other resource enhancements included in the study (Breffle and Rowe 2002).

Breffle and Rowe (2002) also included public access in their Green Bay choice experiment. The attribute levels included were similar to the levels used to measure the public access social norm in this study. The attribute levels ranged from improving existing facilities to adding additional park acreage. Breffle and Rowe (2002) found limited interest in enhancing existing parks and almost no interest in adding park acreage. This corresponds well to the pattern of the social norm curve measured in this study. Because public access is generally considered an important issue for Lake Champlain, this is a surprising result. Furthermore, the individual ecosystem characteristic questions show that public access is an important concern of the sample population (Figure 56). Eighty-two percent of the respondents indicated that public access was "very important" or "important." Public access was also commonly mentioned in the comment section of the questionnaire. The reasons for this inconsistency are not clear. Because the potential conditions listed along the x-axis are not ordinal, as in the other social norm questions, this question may have been more difficult to answer. Another potential explanation is that the ability to access the lake is highly valued (hence the high importance ranking), but there is general satisfaction with the current level of public access (as shown by the social norm curve). Further examination is necessary to clarify social preferences for public access to Lake Champlain.

Stream impairment due to stormwater and land development is related issues that are not receiving much attention from the LCBP. However, they are ecologically important and are being considered in the ecosystem indicators project. Results from the survey indicate that these issues are also socially relevant. Land use is a complicated problem that is difficult to manage. There is evidence here, from the choice experiment and the social norm curves, suggesting LCBP stakeholders are interested in efforts to minimize land development and maintain the agricultural landscape in the Champlain Basin.

While public beach closure is a significant predictor of choice in all of the estimated models, it is a relatively more important predictor in Vermont. In part, this may reflect the differences in the bacteria standards and beach closure frequency in Vermont compared to New York rather than an actual difference in preferences. In the interest of managing Lake Champlain as one ecosystem, the LCBP should encourage the implementation of consistent standards, monitoring and closure policy for all Lake Champlain beaches. The current inconsistency in standards and beach closures sends a confusing message to the users of Lake Champlain beaches.

With respect to nuisance species, survey results support continued management of water chestnut and sea lamprey. However, written comments suggest a lack of awareness regarding water chestnut and a concern over the use of chemicals in the treatment of nuisance species. Several people commented that they were unaware of the water chestnut problem and others stated that they would not support management to reduce water chestnut if it involved the application of chemicals. This result is consistent with the response to safe fish consumption. In general, there is concern regarding the release and presence of toxic chemicals in Lake Champlain in addition to a concern for nuisance species.

**Additional Research**

This research explored the use of choice experiments and social norm curves in the context of ecosystem management. Although many previous studies utilizing choice experiments and social norm curves have included attributes associated with the natural environment, there are very few choice experiment applications (e.g., Breffle and Rowe 2002 and Johnston, et al. 2002) directly addressing ecosystem management issues and no known applications of social norm curves in the context of ecosystem management. Consequently, more research is necessary to develop these techniques to their full potential as tools for ecosystem management.

One challenge in using choice experiments and social norm curves is identifying the appropriate sample population. Because ecosystems are shared resources, it would be ideal to draw the sample from the general population. However, it is probably not reasonable to expect that all citizens are interested in or even familiar with the ecosystem under consideration. From the perspective of the investigator, there might be a tradeoff between obtaining a lower response rate and potentially lower quality data from a sample of the population at large or collecting data from some alternative, specialized population that cannot be used to draw more general conclusions. Alternatively, it may be possible, albeit more costly, to screen suitable respondents from the general population by telephone or with an initial mailing. The advantages and disadvantages of alternative sampling strategies is an area for further research.

The use of the choice experiments in conjunction with the social norm curves has not been fully explored. From a management standpoint, it may be beneficial to use the same or corresponding levels in choice experiments and social norm curves. This is a check for internal consistency of responses. It also allows for the resulting data to be utilized in a complimentary fashion. Choice experiments yield information regarding tradeoffs while social norm curves allow exploration of the acceptability of a wider range of conditions. Therefore, each of these techniques may be more powerful when used in conjunction with the other.

More research on the cognitive burden of choice experiment questions on respondents is warranted. Choice experiments were originally designed to analyze preferences for consumer goods, such as cars or soap, items for which people have experience making decisions. Regardless of the object of choice, choice experiment tasks are presumed to be difficult for respondents (Louviere, et al. 2000). Because people are generally not accustomed to trading-off ecosystem components or making ecosystem management decisions, choice experiment involving ecosystem characteristics are potentially even more challenging than traditional choice experiment applications. To obtain the highest quality data at the lowest cost to respondents, investigators should attempt to reduce choice task difficulty as much as possible. This is done by performing extensive pretesting on the survey instrument through interviews, focus groups and pilot survey implementation tests. It may also be possible to reduce task complexity (and increase data quality) by providing respondents with background information on the ecosystem attributes. In this study, respondent comments suggest a need for information regarding the significance, current condition and management approach for each ecosystem characteristic. Providing people with such information may help them to make choices that do, in fact, reflect their preferences.

In order to maximize the value of the resulting data, the levels used to measure social norm curves and in the choice experiment should be carefully planned. The range of levels and the wording of the levels should be understandable to respondents and useful to decision-makers. In other words, the investigator needs to be able to translate ecosystem variables of interest into an aspect of the ecosystem that is relevant to respondents. For example, managers are concerned with the level of phosphorus in Lake Champlain, but people experience the effect of phosphorus in terms of water clarity or extent of algae blooms. In a similar way, aspects of the ecosystem of concern to the public must be translated into measurable and manageable components of the ecosystem. In addition, the influence of level descriptions on the results and consequently, the management implications is another area requiring systematic analysis. Based on the results of the choice experiment and the social norm curve for water chestnut in this study, the management recommendations resulting from choice experiment and social norm curve analysis may be a function of the level descriptions.

**Summary of Social Analyses**

The results of the choice experiment and the social norm curve construction provide complementary information regarding social preferences for the characteristics of the Lake Champlain ecosystem. Both of these approaches yield information that is richer and more valuable than what can be learned from simple categorical judgment survey questions. Because the choice experiment forced respondents to make tradeoffs between attributes, respondents had to decide which ecosystem characteristics were more important to them. As discussed by Breffle and Rowe (2002), these types of decisions are a real and necessary component of resource management. Managers must decide where and how to allocate scarce staff and financial resources. A choice experiment provides a structured context in which members of the public can make ecosystem management tradeoff decisions. The results of choice experiments provide managers with information about how these tradeoffs can be made in a way that maximizes public support for ecosystem management.

Social norm curves are used to assess ecosystem characteristics one at a time. They do not provide any information regarding preferences amongst multiple ecosystem characteristics. However, social norm curves can be used to examine the acceptability of a wider range of potential ecosystem conditions than can be included in a choice experiment. The number of levels that can be included in a choice experiment is limited by the complex statistical design. While the coefficients of the utility parameters show whether a particular attribute level is acceptable (positive) or unacceptable (negative), it may not be clear where the shift from an acceptable to an unacceptable condition occurs if the attribute levels span a larger range of conditions. Social norm curves can more precisely identify the range of acceptable conditions, making this technique a useful complement to the choice experiment approach.

The importance of integrating social values and enhancing public support for ecosystem management is discussed at length in the literature. However, there is little consensus about how to accomplish this. The techniques presented here have considerable potential for quantitatively assessing social preferences and integrating that information within an adaptive ecosystem management framework.

## C. Moving Forward to Define Acceptable Levels for Lake Champlain's Indicators

To define acceptable levels for the Lake Champlain Ecosystem Indicators program, both ecological and social perspectives must be balanced. The acceptable levels must not only protect the ecological integrity of the ecosystem, but also provide a condition that is in keeping with the preferences of the stakeholder community. In some cases, the preferred state might dictate a higher standard for a state indicator than the level necessary to protect ecological integrity. In this case, the social perspective might be the deciding factor. In other cases, the level necessary to protect essential ecological functions might be more restrictive than the level that drives public response. In this case, the ecological perspective might be the deciding factor. Ideally, the socially desired condition and the range of conditions necessary for ecological integrity will overlap. If the indicators recommended in this report are adopted, we suggest that an appropriate next step might be to convene a group to consider acceptable levels.

**Phosphorus**

For the phosphorus issue area, the acceptable levels for phosphorus concentration in the lake have been agreed upon by the states of Vermont and New York and province of Quebec in the in a Water Quality Agreement that was adopted pursuant to Memorandum of Understanding for Lake Champlain in 1993. These concentrations have been codified in the phosphorus TMDL (VT DEC and NYSDEC 2002), which has now been approved by both states and the US EPA. The TMDL also defines the loadings that are acceptable for each lake segment. Although it may be prudent to revisit these loadings to make sure that they represent the most efficient path to achieving the in-lake standards, they are the current mandates.

There currently is no acceptable level for the other recommended state indicator, chlorophyll *a* concentration. Our social analyses suggest that 7-10 days of algae bloom conditions in a summer is the transition point to unacceptable conditions. Although we currently do not collect data in this manner, we suspect that this level is exceeded in many lake segments. The original analyses conducted by Vermont and New York DECs to establish the in-lake phosphorus standards (Smeltzer and Quinn 1996) suggest that phosphorus concentrations above about 14 μg/L can produce socially undesirable conditions, and these phosphorus concentrations are common in most lake segments.

From an ecological perspective, the acceptable level of chlorophyll *a* would be the level that provides sufficient biomass to support the desired biological community in the lake. Because the desired biological community has not been articulated, it is difficult to determine what level of productivity is necessary to support it. This should be addressed in the future.

The acceptable levels of the pressure indicators should relate directly to the acceptable levels of the state indicators. While the links between the phosphorus loading targets and the in-lake concentrations in the lake are clearly articulated in the TMDL, the acceptable levels of the pressure indicators that drive the phosphorus load have never been addressed. To address these levels, a variety of social and ecological factors must be considered, including the realities of the phosphorus generation associated with people and animals, the options for land use in the watershed, and a host of socio-economic variables.

Likewise, the acceptable levels of response indicators must be consistent with the goals for the state indicators, but will also require the consideration of a variety of other factors. The realities of state and federal management budgets and the social choices made by the citizens of the basin will drive these levels, but so long as the level of response is less than that needed to achieve the acceptable level of the state indicators, Lake Champlain will not reached the desired state outlined in *Opportunities for Action* and the phosphorus TMDL.

We included two indicators in our social norm curve analyses that relate to the phosphorus pressure indicators and to the urban BMP response indicator. In our study, basin residents indicated that the current level of land use conversion to urban land is marginally unacceptable.

They also clearly indicated that the number of streams impaired by stormwater was unacceptable, suggesting that the level of urban BMP implementation is below the acceptable level.

**Bacteria**

In the bacteria issue area, the primary state indicator is bacteria levels in lake water. The acceptable level for this indicator is established in the state and provincial sanitary codes. These standards currently differ significantly. The first level management response to bacteria levels that exceed the acceptable level is to close the beaches to swimming. Our social analyses suggest that an acceptable level of public beach closures as a result of bacterial contamination is about 7 days per summer season. If this were used as an acceptable level, it would appear that most public swimming areas have achieved an acceptable level of performance. For those areas where bacteria levels are too high and beach closures are too frequent, acceptable levels of the pressure indicators and other response indicators should be considered.

**Mercury Toxicity**

Establishing acceptable levels for the state indicators in this issue area will be difficult because the data that are currently available are so sparse. The mercury concentration in fish tissue is the state indicator that the public relates to most directly. Our social analyses clearly show that from the social perspective, this level is too high. To the residents of the Lake Champlain basin, fish with low levels of mercury in their tissue are desired not just for safe consumption but also because that would symbolize a healthy lake.

The FDA and EPA have established consumption guidelines for protection of human health that suggest that the concentration in fish should be less than 1 μg/g (U.S. FDA 2004). This level could be considered and acceptable level from both an ecological and social perspective because people are part of the Lake Champlain ecosystem. However, it is not clear whether this level would protect the remainder of the Lake Champlain ecosystem, including the fish themselves.

If additional data are gathered in this issue area, eventually bioaccumulation factors for mercury could be determined and used to help guide acceptable levels of mercury in the water, sediment and food web, but this would be a considerable undertaking. However, only after acceptable levels are established for these state indicators could acceptable levels be considered for the pressure indicators and the response indicators.

**Sport Fish Community**

Three of the state indicators in this issue area are population measures. From an ecological perspective, the acceptable levels of these measures should represent ranges that would be expected in a stable, self-maintaining population in Lake Champlain. From a social perspective, the acceptable levels should relate to the desired fish community in Lake Champlain. Although the fish and wildlife management community has described this desired community from their perspective in a very general way in the salmonid restoration plan (Fish and Wildlife Management Cooperative Policy Committee and Technical Committee 1977), numerical estimates would be difficult based on the current data base. No estimate of the productive capacity of the Lake Champlain ecosystem, either in its current trophic state or after the phosphorus standards are achieved, has been made. Such as estimate is prerequisite to predicting the overall carrying capacity in major lake segments. A broader discussion of the desired community among the Lake Champlain stakeholders may also be appropriate, to consider species other than the salmonids, and lake segments beyond the Main Lake, Inland Sea and Malletts Bay.

An acceptable level for the remaining two proposed state indicators will most appropriately be established from the social perspective. The percent abundance of exotic species recognizes the changing composition of the Lake Champlain fish community. There is clearly no consensus in the ecological literature about how many invasions an ecosystem can withstand (e.g., Mills et al. 1994, Paine et al. 1998, Gunderson and Holling 2002). Likewise, the extent of tolerable level of impact to nontarget species with the use of lampricides is a social choice.

The pressure indicators in this issue area represent the four most likely contributors to changes in the fish community in the lake. The only pressure indicator for which acceptable levels might be established in the near term is the sea lamprey wounding rate. Both tolerances of the basin stakeholder community and the level of parasitism the population can withstand at the desired community composition should be considered in determining this level. Our social norm curve analysis suggests just a single lamprey wound on a caught fish may be undesirable from a social perspective.

The response indicators relate to stocking and lamprey control; the acceptable level of these indictors should be linked to the management intensity necessary to achieve the desired sport fish community in Lake Champlain.

**The Pelagic Food Web**

A relatively large number of state indicators have been proposed in this issue area, but we clearly do not have enough information to establish acceptable levels from the ecological perspective. From a social perspective, the recent public concern about toxic blue-green algae blooms and the public health threat they pose suggest that the current percentage of toxin-producing cyanobacteria is above the acceptable level. The World Health Organization (1998, 2003) has recommended guidelines for toxins in drinking water and recreational waters used for that might guide acceptable levels for this indicator from a social perspective.

The acceptable levels of the pressure indicators in this area must be consistent with the levels established for phosphorus. Acceptable levels for the immediate response indicators, number of beach closure and drinking water advisories to protect human health, will like be predominantly a social choice.

## Water Chestnut

For this issue area, we currently have more data for the pressure indicator, extent of the water chestnut infestation, and the response indicators than we do for the state indicators. Our social norm curve analysis suggests that the current extent of the water chestnut infestation is above the acceptable level. While the acceptable levels for the pressure and response indicators might logically be established initially based on social choices, as data are gathered on the state indicators, these levels might be adjusted. Ecological information will be especially important in determining a long-term goal for native aquatic plants.

## Recreation

The indicators in the recreation issue area are limited by available data, and should be considered preliminary. As additional data are gathered, it will become easier to consider acceptable levels. In our social norm curve analysis, we explored whether changes in public access were needed around Lake Champlain. That analysis suggests that the current level of public access to the lake may be acceptable.

## Adaptive Management

Because the existing data sets for many indicators are so limited, acceptable levels should logically be approached within the context of adaptive management. Preliminary judgments can then be modified as additional data become available. A regular cycle of indicator evaluation will help narrow in on the most appropriate levels in the most efficient and effective manner.

*Opportunities for Action* presents eight goals for managing the Lake and the Basin. Although some of the goals convey the desired state of certain components of the ecosystem, and therefore, can help guide the development of acceptable levels, there is no overall desired ecosystem state articulated in the document. Furthermore, all of the goals in the plan may not be fully compatible. For example, the phosphorus reduction goal establishes target phosphorus concentrations for the lake segments. The objectives of the recreation goal include increasing public access to the lake and promoting sustainable tourism. Increases in tourism and recreational use of the lake can contribute to increased phosphorus loading unless more aggressive management actions are taken to offset this load.

In a similar vein, both phosphorus concentration and recreational use will affect the living resources in the lake, but these linkages are not considered in the current management strategies. For example, the in-lake phosphorus criteria were developed based on consideration of the links between phosphorus and recreational use and enjoyment of the lake, but these criteria have not been examined from the perspective of biological community composition desired in the lake. The living resource goal is to restore and maintain a healthy and diverse community of fish and wildlife; this goal cannot be achieved without consideration of the phosphorus, toxic substances, and habitat management actions presented elsewhere in the plan.

For more than a decade, we have been conducting a large-scale experiment on Lake Champlain without collecting the data we need to interpret it. At the top of the food web, we have been changing fish abundances by stocking sport fish and reducing sea lamprey. At the bottom of the food web, we have been reducing phosphorus, which controls the growth of the algae. Somewhere in the middle, through the introduction of zebra mussels and other exotic species, we have been altering a host of connections both up and down. Very clearly, adaptive management could provide a context for addressing these unknowns, if a set of indicators are adopted and the monitoring data to quantify them are collected over time.

As *Opportunities for Action* continues to evolve, the linkages between sections of the plan should be explicitly addressed so an overall desired ecosystem state can be more fully articulated. As this is done, it will also be possible to more fully examine the social and ecological factors that must be examine and balanced in establishing acceptable levels for the all the indicators we have recommended.

## IV. A FRAMEWORK FOR AN ENVIRONMENTAL SCORECARD

The goal of the environmental scorecard is to provide an overall assessment of the state of the Lake Champlain Basin ecosystem and the effectiveness of the management actions undertaken as part of the LCBP. It should be geared to the level of understanding in the general public and be presented in a simple and graphically appealing manner. Figure 2, the underlying pressure-state-response framework for the Lake Champlain Ecosystem Indicators Program, should be part of every scorecard that is published.

While there are too many indicators in the overall suite to be presented in every scorecard, and many of the indicators are still in need of additional refinement, the scorecard should still use the PSR framework to increase understanding of the cause and effects relationships in the ecosystem. The scorecard should reference the management goals and objectives outlined in *Opportunities for Action* and reinforce what the LCBP hopes to accomplish with its various investments and actions. Each indicator used in the scorecard should be presented with its acceptable level so that its current value is interpretable.

The information presented in the scorecard might be layered, with simple concepts and a general overview presented first for those who only want a brief report. More detailed information can then be presented in follow-up sections for those who wish to learn more. The so-called “tabloid” style of presentation might be most appropriate for this, with general graphics depicting the overall state of the lake in the middle, and explanatory text and trends data in the sidebars and other supporting locations. Most of the other indicators programs we reviewed use an “issue” format for presenting information in their data reports. That format fits nicely with our PSR diagrams and the chapters in *Opportunities for Action* so might be appropriate for Lake Champlain as well.

The LCBP Atlas (LCBP 2004) contains figures that might be used to report on individual indicators in such a tabloid format, focused on the issues. For example, the phosphorus concentrations in Lake Champlain lake segments are already presented in an excellent graphic that contains the acceptable level (Figure 58).

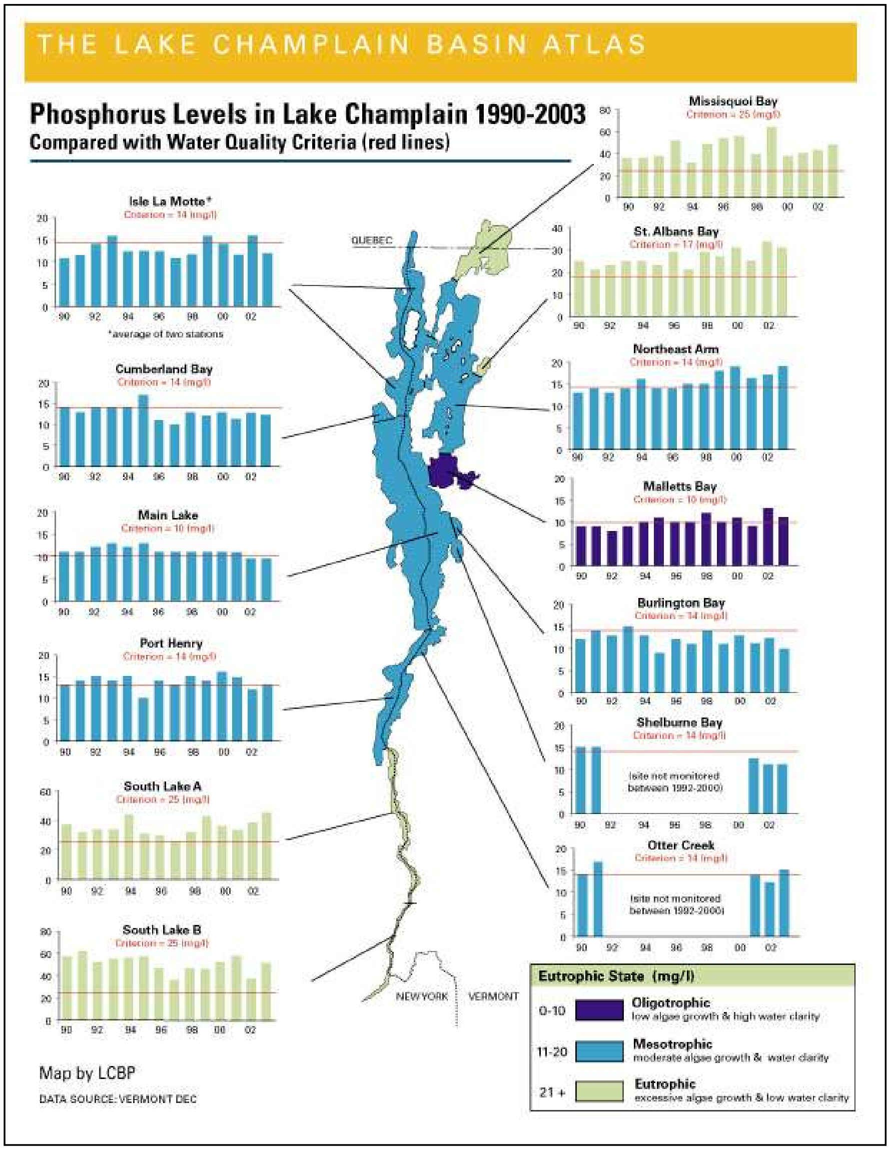



**Figure 58: Phosphorus concentrations in Lake Champlain, 1990-2003. Acceptable levels in each lake segment are indicated in red (LCBP 2004).**

In our search of the literature, we did not find an example of a grading system we thought credible. In fact, almost every project we reviewed rejected the notion of a grade because it would oversimplify the complexities of the ecosystem. We have come to agree with that perspective, but we still believe it would be valuable to present an overall assessment in a quickly registered format. Therefore, we propose that the scorecard be prepared as a series of scoresheets that communicate basic trends in the indicators using the common traffic light as a metaphor for indicator condition. Following this format, a green light or green color would mean an improving condition in an indicator, a yellow light or color would mean no change in condition (and thus caution), and a red light, would mean a deteriorating condition in an indicator. The traffic light colors might be used to present each indicator in an issue area, or to present the level of concern about each issue area in a lake segment. The latter presentation might use a base map from the LCBP atlas as the anchor graphic on the scoresheet.

As an example of what this might look like we present a prototype scoresheet for the phosphorus issue area for two lake segments, the Main Lake (Figure 59) and Missisquoi Bay (Figure 60). All pressure, state, and response indicators for which data exit should be presented on the scoresheet. The indicators are arrayed in similar placement to that in Figure 2, the framework diagram for the indicators program.

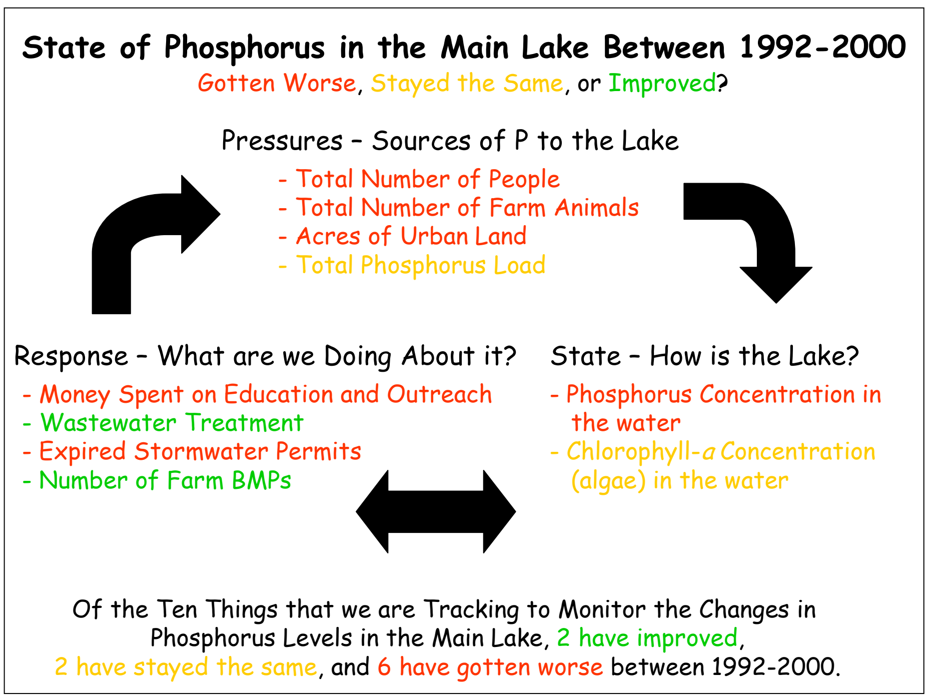



**Figure 59: An example scoresheet for the phosphorus issue in the Main Lake.**

This style of scoresheet easily conveys information about the sources of phosphorus to the lake. It also clearly shows that both phosphorus concentration and the density of algae in the lake reflect these pressures. The predominance of red tells the reader that the pressures are not declining and the state of the lake is not improving. In the response category, it is clear that management efforts have concentrated on wastewater treatment and implementation of BMPs on farms.

What this style of scoresheet cannot do, however, is give an indication of the magnitude the remaining problem, or convey how far away from the acceptable level any one indicator might be. This information would need to be communicated in the accompanying material in the publication. It is critically important the scorecard convey which components of the Lake Champlain ecosystem are furthest from the management targets, as expressed in both the acceptable levels and the management goals in *Opportunities for Action*.

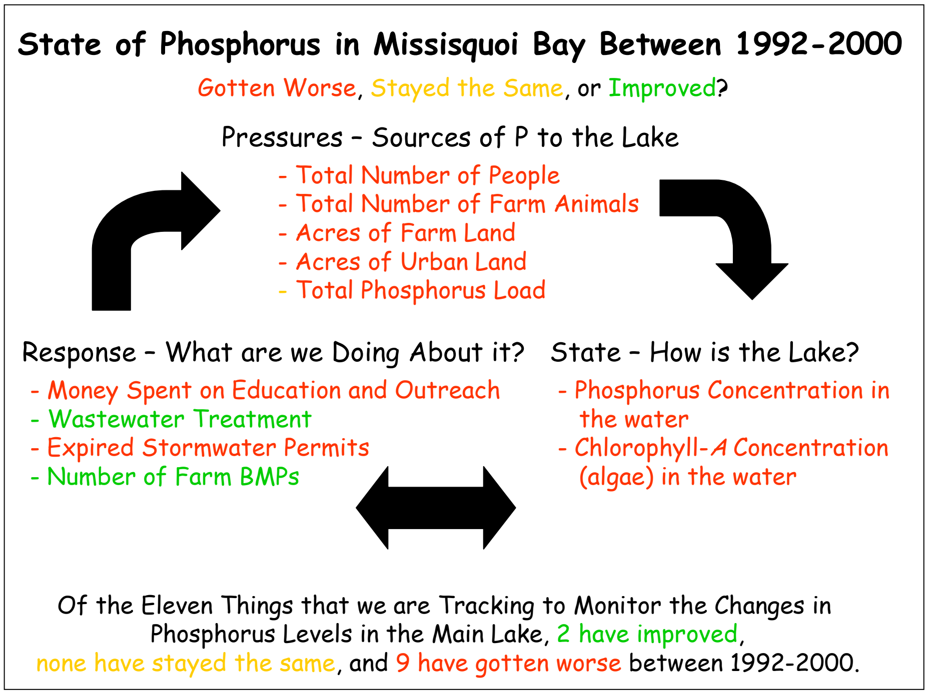



**Figure 60: An example scoresheet for the phosphorus issue in Missisquoi Bay.**

The same traffic signal format could be used with a map of the lake segments, highlighting the issues of greatest concern in each lake segment (Figure 61). Color assignments could be made on the basis of distance from the desired state in each issue area, or another protocol endorsed by the Lake Champlain Steering Committee.

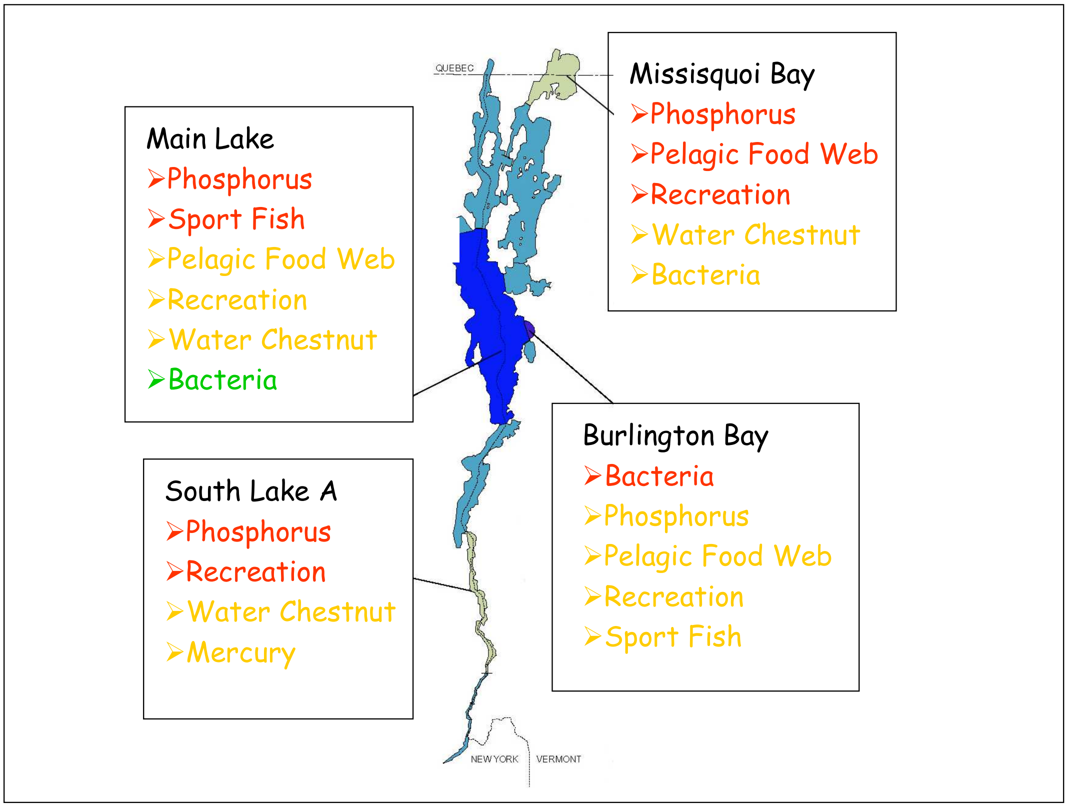



**Figure 61: Example scoresheet based on the relative importance of the issue areas in lake segments.**

As indicator data are assembled, the details of the specific elements of each scoresheet will become easier to design. The scorecard should become a regular vehicle for communication and public education. While it will be a long time before the LCBP has enough data to present a full "State of the Lake" report based on these indicators, we recommend moving towards a regular update of a core set of indicators as soon as possible. The Chesapeake Bay Program (CBP 2002) reports on its indicators in an approximately biannual basis, adding years to a trends diagram for key indicators each time the "State of the Bay" report is published. Because this program has established targets, or acceptable levels for many of their state indicators, they also report how far away from these targets they are for many key indicators.

The EPA's National Coastal Assessment (NCA) Program also uses a stoplight-type format to present their indicators. They have created five indexes that summarize the condition of estuaries and coastal waters on both regional and national scales. Each of these five indexes is rated on a scale of one to five. Scores of one are defined as "poor" and shown as red, mid-range scores are called "fair" and shown in yellow, and higher values are defined as "good," and shown as green. These traffic light colors, along with overall condition labels of poor, fair and good, are displayed horizontally with icons for each indicator placed below the colored bar where it falls along the continuum from 1-5, or red to green (U.S. EPA, 2004).

If the stoplight approach is used in a Lake Champlain “State of the Lake” report, a larger group would need to determine how the colors would be applied and whether they would only depict condition, or would be used to indicate trend, as suggested above. Simple trends information like that presented in this report could accompany the scoresheets, and explanations of the cause and effects relationships that are guiding the management actions could be included in the text. In the absence of trends data, an explanation of the hypothesized relationships and the current state of the indicator might be presented. In a first ever state of the environment report for Lake Ohrid (Watzin et al. 2002), these explanations were used by both the public and the Lake Ohrid Management Board to prioritize activities in their action plan. As subsequent scorecards are produced, the text can focus on emerging issues and the increased understanding coming from additional data accumulation.

## V. DISCUSSION

We have proposed a suite of indicators for the Lake Champlain Basin Program that can begin to provide information on both the state of the Lake Champlain ecosystem and the effectiveness of some of the principal management actions in the comprehensive management plan, *Opportunities for Action*. However, moving forward with an Ecological Indicators Program is not without its challenges and tradeoffs, as well as its opportunities.

### Monitoring and Data Management

The current monitoring efforts on Lake Champlain are fragmented, and sometimes without clear objectives. We believe that there would be substantial benefits in examining these programs in light of what they are accomplishing, both individually and in combination. The result of such an examination might be a streamlined monitoring program with significantly greater impact. Just as management of Lake Champlain must be approached on a cooperative basis, monitoring must also be a joint undertaking, with each partner contributing to an overall assessment of the state of the Lake. Currently, only the Long-term Biomonitoring project is providing lakewide data, and this sampling effort is focused on phosphorus; it does not include the other issue areas. Although plankton samples have been collected for a decade, all of these samples have not been analyzed and much of the data is not yet interpreted or easily available.

One of the biggest challenges we faced in Phase 2 of this project was gathering the data we needed to examine potential indicators. Data were often hard to track down, difficult to get, and in a form that required enormous effort in order to extract, summarize and use the information. If there were monitoring reports that accompanied the data, they tended to present the information with minimal analysis and interpretation, making it of minimal value for management.

These are universal problems, not ones unique to the Lake Champlain Basin. The same type of data management challenges are increasing receiving national attention in fora sponsored by the National Science Foundation (NSF) and others. Under the umbrella of standardizing the Cyberinfrastructure, the problems associated with maintaining, networking and providing access to heterogeneous environmental data and standardized formats for metadata are being discussed in the NSF's Advisory Committee for Environmental Research and Education. The National Academy of Science is also forming a Long-Lived Data Collection Task Force that will examine how to store and make monitoring data sets available. There is an opportunity for the LCBP to be in the forefront by placing focused attention on this issue in the future. Clearly, better data management and data availability are essential to both increasing our understanding of the Lake Champlain ecosystem and to informed decision-making.

The question of how to make the data themselves better is not a trivial one. Clearly, the more effort that goes into designing a useful monitoring program initially, the better the product will be. If a goal of the program is to detect trends over time, then a careful analysis of the power of the sampling effort, for both temporal and spatial resolution, is needed. The data set must meet the assumptions of the statistical tests that are envisioned, and must be appropriate for the area of extrapolation. Our phosphorus model and analysis of the phosphorus data, the LCBP's most extensive data set, shows that even these data provide relatively coarse resolution for detecting subtle trends over time. Because we only have one sampling station for each lake segment, we are also on tenuous ground extrapolating to the entire lake segment based on this data point.

**Other Indicators Programs**

Our struggles to define our Ecological Indicators Program are also not unique, and it is instructive to look at a few other selected examples. Each of these other programs, one national in scope and the others regional, has strengths and weaknesses. As we move forward with our program we can learn from their strengths and try to avoid their weaknesses.

The Heinz Center (2002) recently completed a large scale effort to define indicators that can provide information on all the major ecosystems in the United States. Their report on the state of the nation's ecosystems was designed to be (1) policy relevant, relating directly to questions of concern, (2) technically credible, or consistent with current scientific norms, and (3) politically legitimate, or rigorously nonpartisan. In recommending a national program of 103 indicators, the project team focused on measures of state, ignoring the pressures and responses because these would become areas of public and scientific debate. For each of the six major ecosystem types, indicators were chosen in four categories: system dimensions, chemical and physical conditions, biological components, and human use.

Only about a third of the indicators are currently supported by all the data they need for a national evaluation. The Heinz Center chose to select those indicators they thought most appropriate first, and then evaluate the data bases available to support them. The report clearly lays out the data gaps in the hope that this might encourage action to fill the gaps. The Heinz Center also did not to do an overall assessment of the state of the nation's ecosystems, and makes no judgments about whether any condition is "good" or "bad." In doing this, the Heinz Center hoped to avoid politicizing the results or promoting the perspective of any particular interest. Policy-makers will have to interpret the results and decide whether management is adequate or not. The team acknowledged that combining the state indicators with hypotheses of cause and potential management solution must be part of that decision-making process.

The freshwater ecosystem indictors recommended by the Heinz Center (2002) for lakes include total phosphorus concentration in the growing season, Secchi disk transparency in the growing season, a categorical ranking of the number of established non-native species in the watershed, the number of at-risk native freshwater species, the number of unusual animal deaths or deformities in a year, and a measure of fish and bottom-dwelling animal community integrity. The report also recommends indictors of the number of water-borne human disease outbreaks, and the number of freshwater recreation days.

The report goes on to discuss the limitations in some of the indicators. The team notes that at the national level, phosphorus data are not always readily available and the spatial and temporal coverage is inadequate for both general characterizations (because so many samples are focused on point source discharge sites) and trends detection. No measure of fish and macroinvertebrate community integrity was recommended. The team noted that the most commonly used measures of biotic integrity, such as the IBI, were developed for streams, and thus are not wholly appropriate for lakes. Any index that is developed must also be tailored to the species composition in the specific area where it is used. The report noted that defining a nationally consistent scoring system and approach to reference condition will also take considerable additional development (The Heinz Center 2002)

The Heinz Center project was unique in trying to provide a national database that could serve as a point of reference for considering many other questions. Around the nation, there are also a variety of indicators projects that are designed to provide information that can guide management decision-making in a more direct way. The approaches taken and lessons learned from these programs can also help guide the development of the Lake Champlain Ecosystem Indicators Program.

The EPA's National Coastal Assessment Program (U.S. EPA 2004) has synthesized a variety of measures of ecological condition into five broad indices of coastal ecological health. These include a water quality index, sediment quality index, a benthic index, a coastal habitat index, and a fish tissue index. These indices were selected based on the availability and consistency of monitoring data for estuaries and coastal waters across the country, and they are created by combining data collected from different programs administered by multiple federal agencies (EPA, NOAA, and US FWS) and coastal and Great Lake states.

The water quality index is based on a combination of monitoring data and ranges of acceptable values for dissolved oxygen, chlorophyll *a*, nitrogen, phosphorus, and water clarity at each site. It is designed to identify where water quality conditions are "acutely degraded" and, therefore, does not adequately capture episodes of intermittent hypoxia, eutrophication or low water clarity. Definitions of poor, fair and good condition were established using values taken from the scientific literature, as well as surveys of resource managers and the knowledgeable public. For example, a dissolved oxygen concentration of less than 2 mg/l was assigned a rating of "poor" because it is widely accepted as the threshold for hypoxia.

Once an individual site was assessed for each indicator, the NCA then assigned a regional rating based on the percentage of individual sites meeting certain criteria boundaries. These were based on a median of the survey responses provided by environmental managers, the scientific community and the knowledgeable public for these indicators. These regional indicators of estuarine condition were then used to compile a national assessment of aquatic life use and human use attainment. A site was judged impaired for aquatic life use if water quality condition, sediment quality, benthic condition or habitat loss were assigned a poor rating. A site was defined as threatened for aquatic life use if two or more indicators were determined to be fair. If three or four of these indicators were rated good and none poor, then the site was deemed unimpaired for aquatic life use. Human use attainment was based on the fish tissue contaminants indicator.

Although the concept of broad indexes is attractive, the lack of data for many estuaries around the country, the multiple levels of compositing and the heavy reliance on professional judgment leave this indicator approach open to question. As the report openly acknowledges, there are a number of shortcomings in the available data, and the ratings cannot represent all individual estuarine systems. However, if a consistent monitoring program were put into place to systematically collect the data that is used to create the indices over the coming decade, it might be used in the future to provide a reasonable regional and national assessment of coastal waters.

An effort to develop a comprehensive set of indicators for the Great Lakes has been underway for about 10 years. A long list of potential indicators for the Great Lakes basin was developed under the auspices of the Great Lakes Water Quality Agreement (GLWQA) in the 1990s. The purpose of the indicator suite is to "objectively represent the state of major ecosystem components across all Great Lakes basins," to report the status and trends of the basins every two years and to assess progress toward the achievement of the GLWQA objectives (Bertram and Stadler-Salt 2000). The Great Lakes indicator development process has been facilitated by a series of conferences, the State of the Lakes Ecosystem Conferences (SOLEC) beginning in 1994 and continuing on an approximately biannual schedule. Experts are working in seven core groups, including nearshore and open waters, coastal wetlands, nearshore terrestrial, land use, human health, societal and unbounded concerns. The list of indictors is dynamic, but the focus has been on using existing monitoring programs as the data source for the indicators.

The SOLEC indicators are organized using the Pressure-State-Response framework, but to date, the overwhelming emphasis has been on pressure and state indicators (Bertram and Stadler-Salt 2000, Shear et al. 2003, Neilson et al. 2003). Pressure indicators describe both natural processes and human activities that impact, stress or threaten environmental quality. State indicators examine the environment, the quality and quantity of natural resources, the state of human and ecological health and sometimes reflect environmental policy implementation. SOLEC indicators were developed to be applicable at a basin-wide scale and therefore, are relevant to all the Great Lakes. The program includes specific "targets" or acceptable levels for some indicators, especially the physical-chemical ones, but for many others, trends are reported over time (Bertram and Stadler-Salt 2000, Neilson et al. 2003), with the target being a more general restoration or maintenance of a healthy population.

The various investigators in the SOLEC program have explored a variety of metrics for the biological community in the Great Lakes. For phytoplankton, biomass ($g/m^3$) had been tracked in each of the Great Lakes except Lake Superior since 1979 (Barbiero and Tuchman 2001) as a primary state indicator. Several metrics for the zooplankton community have been suggested, including zooplankton size (mean length) and the ratio of calanoid copepods to the sum of the cyclopoids and cladocerans (Neilson et al. 2003). While acceptable levels and the interpretation of these metrics remain a matter of debate, it is expected that the ratio of calanoid copepods to the other groups should increase with decreasing nutrient enrichment. For several other taxonomic groups, such as the benthic amphipod *Diporia* and several species of fish, both the abundance of the population and its contaminant body burden are monitored.

In 2002 and 2003, the US General Accounting Office undertook an analysis of the SOLEC indicators program and progress towards restoration of the Great Lakes (US GAO 2003). The GAO found problems with the coordination of the many federal and state programs on the Great Lakes and came to the conclusion that the available information was not adequate to comprehensively assess restoration progress. The report specifically noted that many indicators were interpreted using "subjective judgment to determine whether conditions are improving" and that despite nearly 10 years work on the SOLEC Indicators program, no date for completing a final list of indicators had been established. Because of the focus on ecological state, the report also noted that the indicators program cannot yet be used to measure overall restoration progress or to "evaluate, prioritize, and make funding decisions on the merits of alternative restoration projects." The GAO recommended that "the Administrator, EPA, in coordination with Canadian officials and as part of an overarching Great Lakes strategy, (1) develop environmental indicators and a monitoring system for the Great Lakes Basin that can be used to measure overall restoration progress and (2) require that these indicators be used to evaluate, prioritize, and make funding decisions on the merits of alternative restoration projects" (US GAO 2003).

The Chesapeake Bay Program (CBP) is a regional partnership among Maryland, Pennsylvania, Virginia, the District of Columbia, the Chesapeake Bay Commission and the EPA that was established in 1983. It has an environmental indicators project with goals to (1) evaluate the progress of the Chesapeake Bay restoration effort, (2) monitor environmental condition and response to restoration, (3) provide information necessary to establish restoration goals, (4) inform and involve the public in achieving the goals and (5) make indicator data available upon request (Sylvester 2001). The CBP developed its own, unique framework that organizes indicators by "track" and "hierarchy." The four "tracks" are (1) nutrients, primarily nitrogen and phosphorus, (2) living resources, which addresses the organisms and habitat of the bay for the livelihood and enjoyment of the public, (3) toxics and (4) cross-cutting, which includes indicators for activities that cause multiple impacts (Sylvester 2001). These four tracks correspond to the major issues being addressed by the CBP. The indicators, in most cases, are also classified as one of the following six hierarchy levels (1) actions by regulatory agencies, (2) response of the regulated and nonregulated community, (3) changes in discharge/emission quantities, (4) changes in ambient conditions, (5) changes in uptake/assimilation and (6) changes in health, ecology or other effects (Sylvester 2001). The first two hierarchy levels are designated administrative and the other four are environmental.

The CBP presents regular "State of the Chesapeake Bay" reports for policy-makers and the interested public (Chesapeake Bay Program 2002), as mentioned in the last section. Each report presents information about the bay along with graphics that track the indicators over time to examine progress towards the goals of the program. Specific acceptable levels have not been established for every indicator, but all the indicators are presented with reference to the goals of the CBP program. The reports are comprehensive (the 2002 report is 60 pages in length) and include an overall analysis.

The CBP indicator program is a strong model; the program has a consistent reporting style that presents indicator data over time and in the context of the overall management goals. However, the indicators themselves are not evenly distributed among the issue tracks and across the indicator hierarchy, and no conceptual model seems to underlie the indicators or link the condition of the bay to particular pressures or individual management actions. The monitoring program has recently been criticized because it does not adequately account for the increases in nitrogen and phosphorus that are coming from a growing population (Whoriskey 2004). The model that has been used to track progress towards the nutrient goals over-credits for nutrient control efforts, and predicts lower nutrient concentrations in bay waters than what is measured through the monitoring program. There is currently a debate underway about how to more accurately present the information – and the state of scientific understanding – in the biannual "State of the Bay" report.

In the Long Island Sound Study (LISS), ecosystem indicators were developed to evaluate the health of the Long Island Sound and answer questions about water and sediment contamination, fish and wildlife populations and habitat and changes due to human activities (U.S. EPA 2001). The indicators were developed from federal, state and other available data and were summarized into a "State of the Sound" report in 2000 (Tedesco 2000) that has recently been updated (Burg 2004). By presenting the status and trends of different components of the Long Island Sound ecosystem together, the LISS hopes to show the public the complexity of the Sound, the links among the problems that are occurring in the Sound, and the human activities that cause those problems (U.S. EPA 2001). Other indicator project goals included providing insight into the effectiveness of the hundreds of millions of dollars invested into Long Island Sound restoration and identifying gaps in the existing monitoring (Tedesco 2000). Although a variety of indicators are included in the program, and trends in indicator measures are tracked over time, acceptable levels have not been explicitly addressed in the Long Island Sound Study for most indicators.

The indicators that we have proposed for Lake Champlain are consistent with many of the indicators used in these other indicators programs. The PSR framework is acknowledged by both The Heinz Center (2002) project and the Great Lakes Indicators project as a useful framework. By recommending indicators in all three areas, we hope to increase the ability of the LCBP to make judgments about the effectiveness of management as the data base grows. The program that we suggest has a conceptual framework, a set of indicators that is tied to this framework, and an application strategy that includes defining acceptable levels over time so that the data gathered can be interpreted and used in an adaptive management approach.

The Heinz Center (2002) recommended developing a biotic index for freshwater ecosystems, although it acknowledged that this would be a difficult undertaking. Alternatively, the Great Lakes and Chesapeake Bay Programs have relied on an indicator species approach. We acknowledge that additional biotic community indicators are desirable over time, but we didn't feel that we could recommend either a specific index or an appropriate set of indicator species at this time.

**Biotic Indexes and Indicator Species**

There is an extensive literature on biotic indexes. As mentioned previously, the most widely used indices are probably the Rapid Bioassessment Protocol (RBP) score (Plafkin et al. 1989, Barbour et al. 1999) developed by the US EPA and the Index of Biotic Integrity (IBI) developed by Karr (1981). The IBI consists of 12 measures, including fish species richness and composition, status of indicator species, trophic organization and function, fish abundance, reproductive status and condition of individual fish. The index value for a particular stream is calculated and then compared to a reference value to judge whether the condition of the stream is acceptable or not. Some authors have argued that multimetric indices obscure, rather than highlight, important patterns. They also suggest that many of the individual metrics that go into these indices are correlated, and, therefore, it is inappropriate to sum them (Green 1979, Norris and Georges 1993, Suter 1993). Despite these limitations, the IBI and a number of similar indexes are now in widespread use.

Over the last several years, the US EPA has been working to further develop its bioassessment approaches and adapt them for use in lakes (US EPA 1998, 2003). These methods are essential if biocriteria are to be used for judging compliance with water quality standards. Most of the measures being considered as indicators relate to taxonomic composition of the algae, macroinvertebrates, or fish. Multivariate statistical approaches are being used to classify sample sites either as reference locations or as sites that are in a healthy or impaired condition.

Multivariate statistics are attractive because no information is lost in the analysis. Each species is treated as a separate variable; therefore, more subtle changes in community structure can sometimes be detected, potentially allowing better differentiation among moderately impaired or degraded sites. Various techniques such as canonical correspondence analysis, principal components analysis, and multiple regression can be used to relate two sets of data to each other, for example, land use or water quality data and taxa abundance data.

However, multivariate approaches also have limitations. They cannot be used to show trends (improving or deteriorating conditions in a time series analysis) and the tests themselves are not generally amenable to significance testing. All multivariate approaches measure differences on a relative scale, so most are highly sensitive to outliers in the data set. If outliers are removed, very real biological information can be lost. Similarly, transformation and removal of rare taxa (to eliminate zeros, which make statistical solutions more difficult) may not be biologically appropriate; rare species may indicate much about biological condition (Watzin and McIntosh 1999).

In a recent essay, Suter (2001) argued that indicator monitoring would be more useful if the indicators were selected using the concepts of ecological risk assessment. In risk assessment, assessment endpoints are valued properties of the ecosystem that are susceptible to stressors of concern because the focus is on determining causal relationships. Suter suggests that many ecological indicators programs are only peripherally concerned about causal relationships and track measures that cannot be interpreted. If indicators were instead selected as if they were assessment endpoints, the monitoring program would have much greater significance for management decision-making.

The idea of selecting an indicator species to represent the response of many others is attractive, but it means that the indicator taxon must be correlated with many other less well-known taxa (Prendergast et al. 1993, Carignan and Villard 2002). Although there appears to be considerable congruence in patterns of biodiversity at global scales (Prendergast et al. 1993, Gaston 2000), there is much greater variation at finer scales, such as the landscape and habitat scale (Prendergast and Eversham 1997, Allen et al. 1999, Heino et al.2003). At these scales, it does not appear that single taxonomic groups are necessarily a good surrogate for overall biodiversity, although they still may indicate something about environmental quality.

Even if there was good co-occurrence among taxonomic groups, each species has a unique set of habitat requirements and responds to a variety of factors at very different scales. Finally, the sensitivity and response time of populations will vary. All of these things make selection of appropriate indicator species challenging, and the selection will require considerable knowledge about candidates (Niemi et al. 1997, Eiswerth and Haney 2001, Carignan and Villard 2002). While "keystone species," "umbrella species," and other concepts about species that have unusual importance in a community might suggest these would make good indicators, their selection is often likewise difficult and based on incomplete information about community interactions and ecosystem function (Paine 1995, Simberloff 1998). There are very few studies of potential keystones in lake ecosystems.

In a review of potential indicators for agri-ecosystems, Buchs (2003) argues that many managers and policy-makers have unrealistic expectations for biotic indicators based on their experiences with abiotic water quality and soils indicators. They expect that biotic indicators will be easy to assess and understand, that numerical measures, baseline reference values and thresholds can be clearly defined and thus made legally valid, and that there is one best biotic indicator that can represent an entire ecosystem and all the processes going on in it. Such unrealistic expectations can only lead to frustrations and monitoring programs that are both inadequate and scientifically indefensible.

Indicator species clearly can be selected based on human interest and management attention, and this, in large part, has guided the selection of biotic indictors in both the Great Lakes and the Chesapeake Bay programs. Several of the indicators that we have recommended in the sport fish and mercury toxicity issue areas also fall into this category. However, it is important to recognize that species selected by social criteria may, in fact, indicate very little about the overall integrity of the ecosystem (Simberloff 1998, Suter 2001, Carignan and Villard 2002).

For Lake Champlain, where there are significant differences between the biotic communities in the various lake segments, it seems unlikely that we would ever be able to develop a biotic index that would be appropriate for the whole lake, or come up with indicator species that are appropriate in all locations. In the same way that streams and lakes are first sorted into "types," before biocriteria and reference conditions are defined, Lake Champlain would likewise first need to be divided into appropriate segments, and then appropriate indicators be considered.

Obviously, this will be a long-term effort and might more appropriately be considered after additional data collection has increased knowledge about ecosystem function.

**Additional Indicator Areas**

Our proposed indicator suite does not generally include economic indicators. Clearly economic considerations are essential in the decision-making process, and capturing the links between ecological goods and services and their economic values through a series of economic indicators would add significantly to the overall indicators program for Lake Champlain. In recent years, new techniques have been developed that can associate market and non-market values with ecological goods and services (e.g., Costanza et al. 1997, Dailey and Ellison 2002). A number of investigators have also proposed alternative measures of economic activity that separate the positive contribution to the ubiquitous economic indicator, Gross Domestic Product (GDP) from the negative, thus accounting for a variety of social welfare and quality of life considerations. One of these is the Genuine Progress Indicator (GPI) developed by the group Redefining Progress (www.rprogress.org/projects/gpi/). In the spring of 2003, Costanza and Erikson (2003) and their ecological economics class calculated a suite of GPI indicators for Chittenden County. We recommend further exploration of this approach or one like it in the future.

There is particular merit in linking economic measures to environmental measures in ecological modeling and futures forecasting (Carpenter et al. 1999, Eiswerth and Haney 2001, Bennett et al. 2003, Osinski et al. 2003). The Lake Champlain ecosystem, like all ecosystems across the globe, is changing, often in ways that we cannot anticipate. Our ability to adapt to the changes depends on our ability to envision future scenarios and understand the ecosystem dynamics that drive them. Practical solutions to complex problems depend on sound socio-economic, as well as ecological data. The examination of scenarios has been part of good practice and decision-making in the business community for years (e.g., Wack 1985, Davis 1998). In ecosystem management, scenario planning could help minimize surprises and cope with uncertainty by developing management approaches that are robust under a range of potential future conditions (Bennett et al. 2003). Even in cases where quantitative estimates are not possible, qualitative expressions of the relative trade-offs that come with various activities can often be outlined and factored into decision-making (DeFries et al. 2004).

We know that natural factors and stochastic events like severe storms and drought can also influence ecosystem condition and the outcome of management. We have not included a full set of indicators to track natural factors in all our proposed indicator suites, but these probably should be included in the future. The influence of these factors can also be considered using scenarios and modeling. The Great Lakes indicators project has placed enormous attention on these factors (Bertram and Stadler-Salt 2000, Nielson et al. 2003) and we can definitely benefit from that experience.

Under the guise of natural factors, we might also consider indicators that capture important natural processes that we know are important for maintaining ecological integrity. For example, we know that the characteristics of the Lake Champlain basin also influence the water quality and ecological integrity of the lake. Therefore, we might consider adding a set of landscape indicators that can help capture these characteristics beyond the simple land use measures we have in the phosphorus issue area. For example, we know that the amount of wetland in Lake Champlain watersheds influences the load of phosphorus to the lake (Weller et al. 1996), and that in general, the proportion of wetland area to watershed area is a general indicator of the ability of the watershed to hold water, and thus potentially treat it (Wetzel 1990). We could consider a landscape state indicator related to these facts. We also know that higher proportions of vegetated riparian zones are associated with the reduced phosphorus loading to surface waters in watersheds (Correll et al. 1992, Osbourne and Kovacic 1993, Lowrance et al. 1997, Perry et al. 1999). Therefore, another potential indicator could be developed that tracks the proportion of the riparian zone that is vegetated in watersheds. This could be either a state indicator, or a response indicator as streambank stabilization and revegetation projects continue to be implemented throughout the basin.

**Research Needs**

Finally, we believe it would be both short-sighted and naive to move forward with an indicators program without also investing in additional process-oriented research, to provide a context for understanding and interpreting the monitoring data. Additional research is needed in all the issue areas and priorities for research funds should be established at the same time that an indicators program is endorsed and implemented.

We believe there are particularly acute needs in understanding the ecology of the lake itself, and in linking the issue areas together. Even though we have a lot of phosphorus data, there has only been limited process-oriented research on the phosphorus in the lake. Not only do we have very little data on sediment phosphorus concentration in most areas of the lake, we also have only a general understanding of the rates of exchange with the overlying water (internal loading). Therefore, we cannot accurately predict response times to phosphorus reductions in lake segments like Missisquoi Bay.

Managing the living resources in Lake Champlain requires both social information (what do the citizens of the basin want?) and biological information about the food web linkages and the biological capacity of the lake. Currently we have insufficient information in both categories.
The LCBP has not explicitly considered the desired condition the lake beyond “a healthy and diverse ecosystem.” Although a worthy goal, this goal is too vague to guide management or decision-making.

The current phosphorus criteria were developed based on data that show impairments in human use and enjoyment of the lake with increasing algal densities. However, there is little information about what other ecosystem characteristics are valued. We currently do not know whether the phosphorus criteria will support the aquatic community that is desired because that community has not been articulated. Although some experiments were conducted on the linkages between nitrogen and phosphorus and algal growth in the Main Lake in the mid-1990s (Levine et al. 1997/9a, 1999b), no work has been done in critical lake segments like Missisquoi Bay and St. Albans Bay. We do not understand plankton dynamics well enough to predict how the food web might respond to nutrient reductions and we have no idea how top-down pressures from the piscivorous fish are affecting the plankton. In other areas, links between the pelagic and benthic communities are increasingly recognized as important (Vadeboncoeur et al. 2002), yet except for acknowledging the importance of mysid shrimp (LaBar and Parrish 1995), little attention has been paid to the importance of the these linkages in Lake Champlain.

This lack of basic understanding makes it extremely difficult to even interpret the data we do have. For example, analyses of trends in the zooplankton monitoring data show dramatic shifts in taxonomic composition in several lake segments (Carling et al. 2004), but we can only speculate about why this might be. We cannot explain why the algae in Missisquoi Bay is now so strongly dominated by cyanobacteria. There have likely been significant changes in the fish communities in a number of lake segments, but we only have a fragmented data base on this assemblage. Some of these changes may be related to management, especially sea lamprey control, but the invasion of exotic species could be even more significant. Until we develop an understanding of the trophic linkages and transfer rates among the aquatic biota in the lake, we will not be able to explain the patterns that we see at any trophic level or make scientifically defensible management decisions.

As the monitoring activities in the Lake Champlain basin are modified to implement an ecological indicators program, testable hypotheses about ecosystem function should be developed in tandem. Those hypotheses might be partially answered by the collection of additional monitoring data if they are collected using the PSR framework we recommend, but process-oriented research will be needed as well. These data can also be used in models and futures scenarios that can inform an adaptive management approach. Models and scenarios offer significant opportunities to help optimize our actions and move as quickly as possible in the directions we want to go.

## VI. NEXT STEPS AND CONCLUSIONS

Obviously, additional effort will be required in order to establish a fully operational set of ecosystem indicators for Lake Champlain. As is evident in the case studies of other indicators efforts, development and implementation of an indicators program is a long-term endeavor and should be undertaken within an overall commitment to adaptive management.

We recommend the following specific steps to follow up on this work and continue the development of an ecological indicators program for the Lake Champlain Basin Program:

1. Convene both technical and policy-level workshops to consider the information in this report and select an initial set of indicators for implementation.

2. Revise the current monitoring programs among the partner institutions in the LCBP in order to collect the data necessary to implement the indicators program. We suggest particular attention be paid to improving current data collection to maximize its utility, and adding indicators in the phosphorus, sport fish, and pelagic food web issue areas.

3. Establish acceptable levels for the state indicators in the Lake Champlain ecosystem indicators program as soon as practical. Use these levels as a basis for defining acceptable levels of the pressure and response indicators over time.

4. Continue to explore the linkages between issue areas in *Opportunities for Action* and explicitly consider a set of indicators that can capture those linkages.

5. Add socio-economic indicators in core issue areas.

6. As part of a regular "state of the lake" report, publish a first scorecard for a core set of indicators as soon as practical, and commit to biannual updates of the scorecard and state of the lake report.

The evidence of human domination of earth's ecosystems is everywhere (Vitousek et al. 1997). Without explicit consideration of our role and our impacts, continued environmental degradation will be the most likely, if unintended, result. We believe it is imperative that people be considered an explicit part of the Lake Champlain ecosystem; it is not possible to manage ecosystems as if we are external to them. In a recent book entitled "Win-Win Ecology," Rosenweig (2003) argues that we can slow the loss of biodiversity and improve environmental conditions for all species in the future, but only if we acknowledge our role and responsibility and act aggressively and deliberately to share our habitats with other species. There are trade-offs that will have to be considered. Land use choices in particular will require balancing current human needs and wants with maintaining future ecosystem functions (DeFries et al. (2004). A thoughtfully implemented ecological indicators program can assist by providing the data and frameworks necessary to inform these choices as the LCBP embraces the challenges of the new millennium.

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# IX. APPENDIX A

## Comprehensive List of Proposed Indicators

### Table 3: Summary of recommended phosphorus indicators.

| Indicator | PSR | Available Measure | Year | Source | Recommended Measure | Frequency (minimum) |
|---|---|---|---|---|---|---|
| Population | P | Human population by state/province | US: 1950 - 2000; Canada: 2001 | US census data at Holmes and Associates; Statistics Canada | Human population by lake segment subwatershed | Update every 10 years |
| Developed land | P | Percent developed land by subwatershed | 1993 | LandSat imagery; data at VCGI | Percent developed land by subwatershed | Update every 10 years |
| Agricultural land | P | Percent agricultural land by subwatershed | 1993 | LandSat imagery; data at VCGI | Percent agricultural land by subwatershed | Update every 10 years |
| Animal units | P | Stocking density (animal units/ha) by subwatershed | VT:2001-2002; NY:1993-2002; QC:1998-2003 | VT AFM; NYS SWCC; QC ME | Stocking density (animal units/ha) by subwatershed | Update every 2 years |
| Phosphorus load | P | Annual mean tributary P load by lake segment | 1991-2002 | LCBP long-term bio-monitoring | Annual mean tributary P load by lake segment | Annually |
| P in water column | S | Annual mean P concentration by lake segment | 1991-2002 | LCBP long-term bio-monitoring | Annual mean P concentration by lake segment | Annually |
| Chlorophyll *a* | S | Lake segment annual average chl-*a* | 1991-2002 | LCBP long-term bio-monitoring | Lake segment annual average chl-*a* | Annually |
| P in sediment | S | Concentration of P in top 10 cm of lake sediment | 1994 | Hydroqual | Concentration of P in top 10 cm of lake sediment | Update every 5 years |
| Aquatic plants | S | Electronic data unavailable | 1998-2003 | VT DEC | Biomass/$m^2$ for areas less than 10 m deep | Update every 5 years |
| Zebra mussels | S | Data on adults unavailable | | | Biomass/$m^2$ for areas less than 30 m deep | Update every 5 years |
| Farm BMPs | R | Percent of farms and animal units treated | VT:1996-2003; NY:2002 | VT AFM; NYS SWCC | Phosphorus load reduction from implementation of agricultural BMPs by subwatershed | Update every 2 years |

| | | | | | | |
|---|---|---|---|---|---|---|
| Wastewater treatment | R | Lake segment P load from WWTF | 1991, 1995-2002 | VT DEC; NY DEC | Difference between lake segment P load from WWTF an the TMDL allocated load | Update every 2 years |
| Urban BMPs | R | Percent of stormwater permits that have expired | 2003 | VT DEC | Phosphorus load reduction from implementation of urban BMPs by subwatershed | Update every 2 years |
| Education | R | Phosphorus specific data unavailable | | LCBP | Dollars spent on phosphorus-oriented outreach and education | Update every 2 years |

**Table 12: Summary of recommended bacteria indicators.**

| Indicator | PSR | Available Measure | Year | Source | Recommended Measure | Frequency (minimum) |
|---|---|---|---|---|---|---|
| Population | P | Human population by state/province | US: 1950 - 2000; Canada: 2001 | Holmes and Associates; Statistics Canada | Human population by lake segment subwatershed | Update every 10 years |
| Stormwater | P | Percent samples that exceed state standard | 2002 | UVM (Burlington Bay only) | Percent samples that exceed standard | Annually |
| Animal units | P | Stocking density (animal units/ha) by subwatershed | VT:2002;NY:2002; QC:1998-2003 | VT AFM; NYS SWCC | Stocking density (animal units/ha) by subwatershed | Update every 2 years |
| Wildlife | P | Data unavailable | | | Measure when necessary in problem areas | As needed |
| Bacteria levels | S | Number of beach water samples that exceed state standards | 1997-2002 | | Percent of beach water samples that exceed state standards at priority locations | Annually |
| Beach closure | R | Days of beach closure at Burlington beaches | 1990-2002 | | Days of beach closure at all Champlain beaches | Annually |
| Farm BMPs | R | Percent of farms and animal units treated | VT:1996-2003; NY:2002 | VT AFM; NYS SWCC | Bacteria load reduction from implementation of agricultural BMPs by subwatershed | Update every 2 years |
| Urban BMPs | R | Percent of stormwater permits that have expired | 2003 | VT DEC | Bacteria load reduction from implementation of urban BMPs by subwatershed | Update every 2 years |

**Table 13: Summary of recommended mercury contamination indicators.**

| Indicator | PSR | Available Measure | Year | Source | Recommended Measure | Frequency (minimum) |
|---|---|---|---|---|---|---|
| Atmospheric load | P | Mean Hg load from atmospheric deposition | 1994-1996 | NOAA | Annual mean Hg load from atmospheric deposition | Updated every 5 years |
| Tributary load | P | Discharge and Hg concentrations | 2000-2002 | USGS | Annual mean Hg load by lake segment | Updated every 5 years |
| Point Discharge | P | Estimated Hg load from point sources | NA | SLU | Measured Hg load from point sources by lake segment | Updated every 5 years |
| Hg in Water | S | Total Hg concentration in water column by lake segment | 2001 | USGS | Total and methyl Hg concentration in water column by lake segment | Updated every 5 years |
| Hg in Sediment | S | Total Hg concentration in sediment by lake segment | 1991 | UVM | Total and methyl Hg concentration in sediment by lake segment | Updated every 10 years |
| | | | | | Number of lake segments above threshold effects level | Updated every 10 years |
| Food web | S | Mean Hg concentration in plankton | 1997 | UVM | Mean Hg concentration in key lower trophic level species in selected lake segments | Updated every 5 years |
| Hg in Fish | S | Mean Hg concentration in walleye and yellow perch | 1988-2000 | VT DEC | Body burden in key species by weight class | Updated every 5 years |
| Hg in Piscivorous Wildlife | S | Mean Hg concentration in selected wildlife species | | | Body burden in selected wildlife species | |
| Consumption advisories | R | NY and VT fish consumption advisories | | VT DOH, NY DOH | Number of species for which advisories exist | Updated every 5 years |
| Basin source reduction programs | R | Dollars spent on source reduction | | LCBP | Hg load reduction achieved | Updated every 2 years |
| Emission reductions | R | | | | Hg load reduction achieved | Updated every 5 years |

**Table 16: Summary of recommended indicators for a healthy sport fish community.**

| Indicator | P S R | Available Measure | Year | Source | Recommended Measure | Frequency (minimum) |
|---|---|---|---|---|---|---|
| Sea Lamprey | P | Mean number of wounds per 100 lake trout | 1982-2002 | LCFWMC; VT FW | Mean number of wounds per 100 lake trout | Annually |
| Habitat Alterations | P | Data unavailable | | | Develop index of habitat quality for spawning areas | Update every 10 years |
| Angler harvest | P | Data unavailable | | | Creel surveys by selected lake segments | Annually |
| Hg and PCBs | P | Mean Hg concentration in walleye and yellow perch | 1988-2000 | VT DEC | Body burden in key species by weight class | Updated every 5 years |
| | | Gonadosomatic index in juvenile walleye | 1995 | UVM | Gonadosomatic index for selected species | Updated every 5 years |
| Sport Fish | S | Mean number of lake trout per gill net lift | 1982-1997 | LCFWMC | Annual lake trout population abundance by lake segment | Annually |
| Forage Fish | S | Mean rainbow smelt catch per trawl | 1987-2002 | LCFWMC | Mean rainbow smelt catch per trawl | Annually |
| | | Rainbow smelt mean length | 1984-2002 | LCFWMC | Rainbow smelt mean length | Annually |
| Plankton and biodiversity | S | Phytoplankton and zooplankton taxonomic composition and relative abundance | 1991-2002, with some missing dates | LCBP long-term bio-monitoring; SUNY-Plattsburgh | Biomass and size distribution of zooplankton | Annually |
| | | Number of exotic species in the lake (fish and plankton) | 2000 | LCBP – ANS plan | Percent abundance of exotic species by taxa | Update every 2 years |
| Nontarget species | S | Abundance of selected taxa before and after TFM application | 1990-1995 | LCFWMC | Abundance of selected taxa before and after TFM application | Annually |
| Stocking | R | Hatchery released smolt equivalents by lake segment | 1972-2002 | LCFWMC | Hatchery released smolt equivalents by lake segment | Annually |
| TFM | R | Miles of stream exposed to TFM | 1990-2000 | LCFWMC | Miles of stream exposed to TFM | Annually |
| Nontarget species | S | Abundance of selected taxa before and after TFM application | 1990-1995 | LCFWMC | Abundance of selected taxa before and after TFM application | Annually |

| | | | | | | |
|---|---|---|---|---|---|---|
| Nonchemical alternatives | R | Data unavailable | | LCFWMC | Stream miles treated by nonchemical alternatives | Annually |

**Table 17: Summary of pelagic food web indicators.**

| Indicator | P S R | Available Measure | Year | Source | Recommended Measure | Frequency (minimum) |
|---|---|---|---|---|---|---|
| P and N in water | P | Number of samples with N:P $\geq$ 50 | 1992-2002 | LCBP | Number of samples with an N:P $\geq$ 50 by lake segment | Annually |
| Phytoplankton community | S | Taxonomic composition and relative abundance | 1992-2001 (some missing years) | LCBP and SUNY-Plattsburgh | Taxonomic composition and relative abundance | Annually |
| | | Percent toxin producing cyanobacteria in selected locations | 2001-2003 | UVM | Percent toxin producing cyanobacteria by lake segment | Annually |
| Blue green algae toxins | S | Toxin concentrations by selected lake segment | 2001-2002 | UVM | Toxin concentrations by lake segment | Annually |
| Zooplankton community | S | Taxonomic composition and relative abundance | 1992-2002 | LCBP and SUNY-Plattsburgh | Taxonomic composition and relative abundance; average size of the zooplankton; | Annually |
| | | | 1992-2002 | LCBP and SUNY-Plattsburgh | Ratio of phytoplankton biomass to zooplankton biomass | Annually |
| Zebra mussels | S | Data for adults unavailable | | | Biomass/$m^2$ for areas less than 30 m deep | Update every 5 years |
| Forage Fish | S | Mean rainbow smelt catch per trawl | 1987-2002 | LCFWMC | Mean rainbow smelt catch per trawl | Annually |
| | | Number of exotic species in the lake | 2000 | LCBP – ANS plan | Percent abundance of exotic species | Update every 2 years |
| Beach closure | R | | | | Days of beach closure by lake segment | Annually |

| | | | | | | |
|---|---|---|---|---|---|---|
| Drinking water advisories | R | | | | Number of drinking water advisories by lake segment | Annually |
| Phosphorus load reduction | R | Annual mean tributary P load by lake segment | 1991-2002 | LCBP long-term biomonitoring | Total phosphorus load reduction by lake segment | Every 2 years |

**Table 20. Summary of water chestnut indicators.**

| Indicator | PSR | Measure | Year | Source | Recommended | Frequency (minimum) |
|---|---|---|---|---|---|---|
| Area Infested | P | Miles north of Whitehall, NY | 1982-2002 | VT DEC and LCBP | Miles north of Whitehall, NY | Annually |
| Water-based Recreation | S | Data unavailable | | | Develop measure of recreational use | Update every 5 years |
| Native Aquatic Plants | S | Data unavailable | | | Native species present and percent cover in shallow water by affected lake segment | Update every 5 years |
| Harvesting | R | Dollars spent on harvesting | 1991-2002 | LCBP | Dollars spent on harvesting | Annually |
| | | Number of mechanical harvester loads | 1982-2002 | LCBP | Number of mechanical harvester loads | Annually |
| | | Biomass removed by hand-pulling | ??? | LCBP | Biomass removed by hand-pulling | Annually |

**Table 21. Summary of recreation and cultural heritage indicators.**

| Indicator | PSR | Measure | Year | Source | Recommended | Frequency |
|---|---|---|---|---|---|---|
| Demand | P | | | | Population and tourism growth rates | Update every 5 years |
| | | | | | Lake-related recreation interests, both residents and tourists | Update every 10 years |
| Ecosystem Condition | P | State measures in other issue areas | | | | Update every 5 years |
| Recreational Infrastructure | S | Miles of bikeway around Lake Champlain | 1995-2003 | LCBP | Miles of bikeway around Lake Champlain | Annually |
| | | Number of public access sites or improvements around Lake Champlain funded by LCBP | 1995-2003 | LCBP | Number of public access sites around Lake Champlain per capita (residents and tourists) by lake segment | Update every 5 years |
| | | | | | Congestion and adequacy of harbor facilities by lake segment | Update every 5 years |

| | | | | | | |
|---|---|---|---|---|---|---|
| Cultural Heritage Sites | S | Number of interpreted wayside exhibits in the Lake Champlain basin | 2003 | LCBP | Number of interpreted wayside exhibits in the Lake Champlain basin | Update every 5 years |
| | | | | | Number of interpreted cultural heritage sites in the Lake Champlain basin | Update every 5 years |
| LCBP Infrastructure Investments | R | Dollars granted by LCBP for recreation and cultural heritage projects | 1993-2001 | LCBP | Dollars granted by LCBP for recreation and cultural heritage projects | Annually |
| Ecosystem Management | R | Response measures in other issue area | | | Response measures in other issue area | As appropriate |
| | | Dollars spent on environmental education and outreach | 1994-2003 | LCBP | Dollars spent on environmental education and outreach | Annually |

# X. APPENDIX B

## Phosphorus Ecological Indicator (PEI) Model Description, Algorithms, Function, Calibration, and Sensitivity

The PEI models were developed for this project to illustrate how semi-quantitative models can be used by managers to help guide their (a) understanding of phosphorus dynamics in the Lake Champlain ecosystem, (b) thinking about the selection of ecological indicators and (c) understanding of the potential long-term impacts of management programs. As currently configured the model is designed to simulate long-term trends in how selected ecological indicators might change in future decades. For this project, PEI models were developed for the Main Lake, Shelburne Bay and Missisquoi Bay segments of Lake Champlain. The PEI models are not designed for formal quantitative forecasting but, rather, are meant to allow semi-quantitative assessment of "what if this," "what if that" questions within a planning/management environment.

### A. General Conceptual Model Framework

The general conceptual framework in which the PEI model functions is shown in Figure B-1. The framework has three parts; (1) the lake segment of interest in which phosphorus is stored in the water column and in the bottom sediments, (2) the lake segment watershed that directly contributes TP to the lake segment water column via non-point and point source discharges and (3) the adjacent, connecting lake segment(s) that exchange TP with the lake segment of interest via exchange and advective flows, in and out.

The PEI model assesses the change over time in the amount of phosphorus stored in the lake segment water column by continuously accounting for all inputs and outputs of phosphorus, a notion analogous to managing a checking account. In the model this is simply described as:

$$\Delta S = \Sigma \text{ TP Inputs} - \Sigma \text{ TP Outputs} \qquad \text{Eq. 1}$$

and

$$S2 = S1 + \Delta S \qquad \text{Eq. 2}$$

where: S1 = amount of TP stored in water column at time t = 1 (kg TP)

S2 = amount of TP stored in water column at time t = 2 (one month later since the PEI model makes iterative computations on a monthly basis) (kg TP)

$\Delta S$ = change in phosphorus storage over one month (kg TP/month)

$\Sigma$ TP Inputs = sum of all the TP inputs to the lake segment water column (Figure A-1). These input fluxes may include: TP in point source discharges, TP in non point runoff, TP resuspended from bottom sediments, TP in exchange and advective flows from adjacent lake segments (kg TP/month)

$\Sigma$ TP Outputs = sum of all the TP outputs to the lake segment water column (Figure B-1). These output fluxes may include: TP settling to the bottom sediments, TP in exchange and advective flows to adjacent lake segments (kg TP/month)

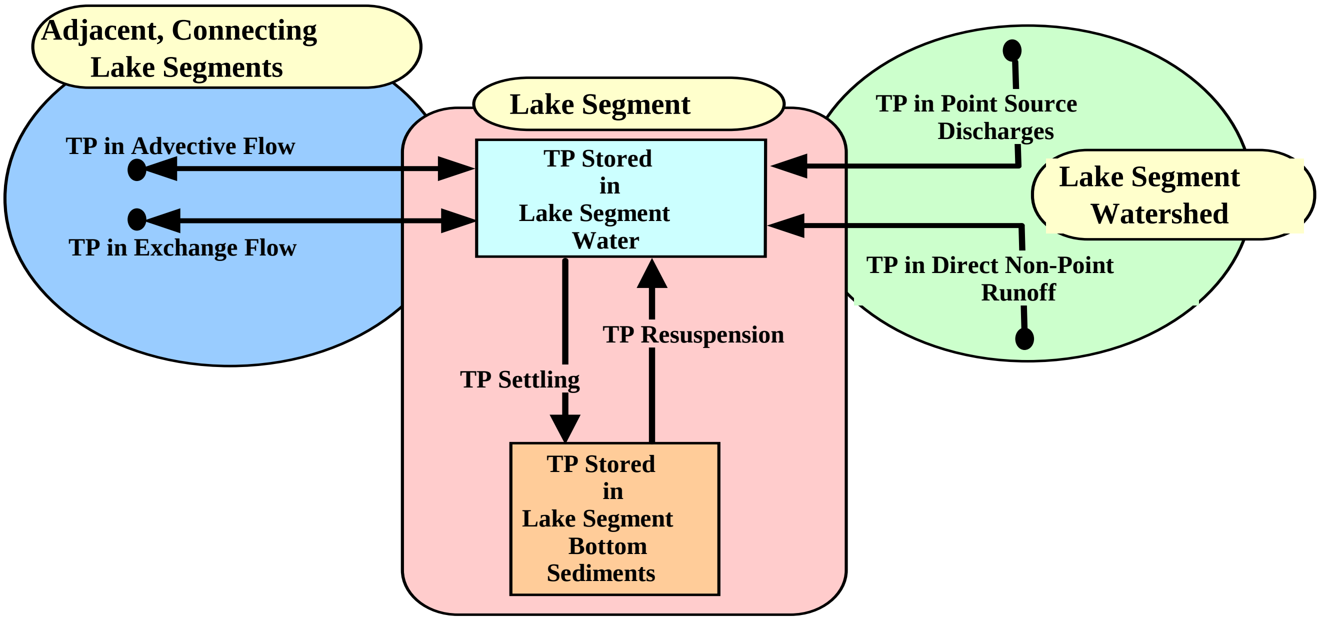


**Figure B-1: A schematic compartment-flux diagram of the framework of the PEI model. The yellow objects are the components of this framework, the lake segment of interest, adjacent connecting lake segments and the lake segment watershed. The arrows are phosphorus (TP) fluxes (mass TP/time) moving from one component to another. The rectangular boxes are locations where TP is stored (in the water column and bottom sediments). This framework allows phosphorus to move from the watershed to the lake segment while, at the same time, being exchanged with adjacent connected lake segments and with the sediments on the bottom of the lake segment of interest.**

Examination of equations 1 and 2 suggest that only three conditions in the trend of phosphorus storage can occur in the lake segment water column. First, the amount of phosphorus stored remains constant over time (e.g. $\Delta S = 0$). This can occur only when the $\Sigma$ TP Inputs = $\Sigma$ TP Outputs, a transitory condition rarely found in natural ecosystems. Second, the amount of phosphorus stored increases over time (e.g. $\Delta S$ is positive). This can occur only when the $\Sigma$ TP Inputs > $\Sigma$ TP Outputs, the condition that currently exists in most Lake Champlain lake segments. Third, the amount of phosphorus stored decreases over time (e.g. $\Delta S$ is negative). This can occur only when the $\Sigma$ TP Inputs < $\Sigma$ TP Outputs, the condition typically sought by the implementation of phosphorus management programs.

By iteratively accounting for the inputs and outputs of phosphorus at each monthly time step, the PEI model generates trends over time in the amount of phosphorus stored in the water column. The model then computes the phosphorus concentration (TP Conc.) in the water column for each monthly time step with equation 3.

$$\text{TP Conc.} = k * (S/V) \qquad \text{Eq. 3}$$

where:

TP Conc. = TP concentration in lake segment water column at any time ($\mu$g TP/L)
k = a constant that allows for adjustment of units
S = amount of TP stored in water column at any time (kg TP)
V = total volume of water in lake segment (cu. meters)

Because the PEI model incorporates these very simple accounting notions it can be a very powerful tool in examining phosphorus dynamics in lake segments. Consider the following general example of how managers might use the model. For any set of TP input and output flux values the PEI calculates a long-term trend in the TP stored and the TP concentration in the water column. When an input flux is reduced, as may happen when a phosphorus management program is implemented in the watershed, a new and different trend is calculated. Maybe the water column TP concentrations increases over time more slowly than it did originally, maybe it trends downward. With this simple example it is seen that the model provides an opportunity to qualitatively assess the sensitivity of long-term trends in water column TP concentration to change in management strategy.

The PEI model, like all other computer models, depends on the algorithms that are used in the model calculations. These algorithms need to appropriately describe what is happening in the lake segment ecosystem. The model also requires the input of data which defines the initial conditions that exist at the beginning of the simulation run and rates at which things happen within the lake segment. Considerable attention was given to incorporating competent algorithms and input data into the model. Whenever a computer model is first run its output is typically compared to a known (true?) value(s). Even under the best of circumstances model generated values rarely coincide exactly with the known values and "calibration" is undertaken to match the model values to the known values; this was the case for the PEI model. The calibration process for the PEI model involves adjusting parameter values that describe TP cycling between the water column and bottom sediments until model output values of water column TP concentration matched the values of TP concentration derived from Lake Champlain water quality monitoring activities (Diagnostic Feasibility Study 1992). This was done for each of the three modeled lake segments and is discussed in more detail below.

**B. Watershed population and land use**

The PEI model tracks, over time, population and land use change for the watersheds that drain directly into each lake segment. To do this initial values are entered of population and the areas of agricultural, urban and forest lands for each watershed. Additionally, a rate of population growth must be specified. The values of these parameters that were input to the lake segment PEI models are summarized in Table B-1.

Currently, in the Lake Champlain basin, population is growing so that urban areas are increasing at the expense of agricultural and forest lands. As population increases over time, the model increases the urban land area by 0.18 ha/capita (calculated from Hegman et al. 1999, Table 4.8 and Lake Champlain Atlas 1990 population estimate). For each hectare increase in urban land area agricultural and forest lands are each decreased by 0.5 hectares so that the total land area remains constant.

**Table B-1: Population and land areas for the watersheds that drain directly into the Main Lake, Shelburne Bay and Missisquoi Bay lake segments. The total land area in each of these watershed is assumed to consist of agricultural, forest and urban lands. Populations for USA are estimated for 2000 from 1990 census data (US Census 1950-2000 at Holmes and Assoc.). Canadian populations for 2001 are estimated from *Statistics Canada* data. Population growth rates are the last 40 year basin average (Lake Champlain Atlas) except a higher rate was applied to Shelburne Bay. All land use areas were calculated from Hegman et al. (1999) Table 4.8. All values in this table are considered in the PEI model to represent conditions that existed around the year 2000, i.e., current conditions or time 0 conditions.**

| Parameter | Units | Main Lake | Shelburne Bay | Missisquoi Bay |
|---|---|---|---|---|
| Population | no. people | 204,860 | 20,000 | 10,000 (USA)<br>24,000 (CAN) |
| Population growth rate | %/year | 1.2 | 2.4 | 1.2 (USA)<br>1.2 (CAN) |
| Agricultural land | ha | 49,180 | 6,600 | 37,540 (USA)<br>39,230 (CAN) |
| Forest land | ha | 419,680 | 5,580 | 113,530 (USA)<br>79,350 (CAN) |
| Urban land | ha | 35, 390 | 4,130 | 8,490 (USA)<br>6,150 (CAN) |

### C. Phosphorus inputs directly from the watershed

In the PEI model, the sources of phosphorus that enter any given lake segment is limited to surface runoff from the contiguous watershed (non point sources) and discharges from wastewater treatment plants in the watershed (point sources).

There are three fluxes of non point source TP input to a modeled lake segment (e.g. separate inputs from the forested, urban and agricultural land areas in the watershed). The model computes each individually according to equation 4.

$$\text{NPS TP flux} = \text{TP Export Coeff.} * \text{Land Area} \qquad \text{Eq. 4}$$

where:

NPS TP flux = TP flux into lake segment for a given land use (kg TP/year)
TP Export Coeff = TP export coefficient calculated from or taken directly from Hegman et al. (1999) (kg TP/ha/year) (there may be different values for the various land uses for different watersheds)

Land Area = the initial area of land within a given land use type, either forest, agriculture or urban (hectares). Because of population change over time land areas within the three use categories change.

The initial values of the TP Export coefficient entered into the PEI models are summarized in Table B-2. These values represent, in the PEI model, current conditions of TP export for the different land uses in the watersheds. The values of these TP Export coefficients can be changed at any time to simulate the implementation of BMPs that reduce erosion and/or enhance nutrient management.

**Table B-2: The TP export coefficients for the different land uses for the Main Lake, Shelburne Bay and Missisquoi Bay lake segment watersheds. These values were taken directly from Hegman et al. (1999) or are average area weighted values calculated from the data of Hegman et al. (1999). In the PEI model these values define the current or initial conditions in the watersheds.**

| Parameter | Units | Main Lake | Shelburne Bay | Missisquoi Bay |
|---|---|---|---|---|
| Agricultural TP Export Coefficient | kg TP/ha/yr | 0.42 | 0.42 | 1.73 (USA)<br>1.63 (CAN) |
| Urban TP Export Coefficient | kg TP/ha/yr | 1.5 | 1.5 | 1.5 (USA)<br>1.5 (CAN) |
| Forest TP Export Coefficient | kg TP/ha/yr | 0.04 | 0.04 | 0.04 (USA)<br>0.04 (CAN) |

Phosphorus is also input to a modeled lake segment in the point source discharges originating from within the watershed. These are wastewater treatment plant discharges and septic system discharges. The PEI model computes the input TP flux from treatment plants as per equation 5.

WWTP TP flux = K*Population*% Sewered WWTP Flow/capita* Effl. TP Std. Eq. 5

where:

WWTP TP flux = point source TP flux entering lake segment from wastewater treatment plant discharges (kg TP/year)
K = a constant to allow for adjustment of units
Population = population of watershed at any point in time (no. people)
Frac. Sewered = fraction of watershed population that is sewered thus contributing to wastewater flow (dimensionless)
Effl. TP Std. = the legally mandated standard for TP concentration of wastewater treatment plant effluents (μg TP/L)
WWTP Flow/capita = per capita wastewater average flow (gallons/day) model uses 120 gpd)

The PEI computes the input TP flux from septic system discharges as per equation 6.

$$\text{SepSys TP flux} = K * \text{Population} * (1 - \%\text{Sewered}) * \text{WWTP Flow/capita} * \text{SSysDrain TP} \qquad \text{Eq. 6}$$

where:

SepSys TP flux = point source TP flux entering lake segment from septic system discharges (kg TP/year) This flux is typically quite small in comparison to treatment plant discharges.

K = a constant to allow for adjustment of units

Population = population of watershed at any point in time (no. people)

1 -Frac. Sewered = fraction of watershed population that is not sewered thus contributing to septic system flow (dimensionless)

SSysDrain TP. = an estimate of the TP concentration septic system drainage discharges (μg TP/L)

WWTP Flow/capita = per capita wastewater average flow (gallons/day) (model uses 120 gpd)

Additionally, phosphorus enters a lake segment due to its hydraulic connectivity with adjacent lake segments. Some lake segments may be connected to many other segments (the Main Lake segment), some to only one other segment (the Missisquoi Bay segment). Regardless the input TP fluxes are associated with exchange flows and or advective flows. In both cases, the TP fluxes is computed by the model as per equations 7 and 8. One such calculation is made for each exchange and advective flow input.

$$\text{Adv TP Input}_i = a * \text{Adv Flow}_i * \text{TP Conc.}_i \qquad \text{Eq. 7}$$

where:

$\text{Adv TP Input}_i$ = advective input flux of TP to the lake segment of interest from segment i (kg TP/year)

a = a constant to allow for adjustment of units

$\text{Adv Flow}_i$ = constant advective flow into the lake segment from connected lake segment i (cu. hm/year)(from Smeltzer 1999)

$\text{TP Conc.}_i$ = the concentration of TP in the ith lake segment at time 0. (μg TP/L)

and

$$\text{Exch TP Input}_i = a * \text{Exch Flow}_i * \text{TP Conc.}_i \qquad \text{Eq. 8}$$

where:

$\text{Exch TP Input}_i$ = exchange input flux of TP to the lake segment of interest from segment i(kg TP/year)

a = a constant to allow for adjustment of units

$\text{Exch Flow}_i$ = constant exchange flow into the lake segment from connected lake segment i (cu. hm/year) (from Smeltzer 1999)

$\text{TP Conc.}_i$ = the concentration of TP in the ith lake segment at time 0. (μg TP/L)

The advective flow inflows and exchange flows used in the PEI models of the Main Lake, Shelburne Bay and Missispuoi Bay lake segments are summarized in Table B-3. Initial TP levels for each lake segment involved in the PEI model computations are summarized in Table B-4.

**Table B-3: Summary of average annual advective flow and exchange flow inputs for the Main Lake, Shelburne Bay and Missispuoi Bay lake segments. All values were taken from Smeltzer et al. (1999). These values remain constant throughout all PEI model simulation runs.**

| Parameter | Units | Main Lake | Shelburne Bay | Missisquoi Bay |
|---|---|---|---|---|
| Advective Inflows | | | | |
| 1. fr. Burl Bay | hm3/year | 9 | X | X |
| 2. fr. Cmbld Bay | hm3/year | 950 | X | X |
| 3. fr. Malletts Bay | hm3/year | 1315 | X | X |
| 4. fr. Shelb Bay | hm3/year | 79 | X | X |
| 5. fr. So. Lake | hm3/year | 3515 | X | X |
| Exchange Inflows | | | | |
| 1. Btwn Burl Bay & Main Lake | hm3/year | 2986 | X | X |
| 2. Btwn Cmbld Bay & Main Lake | hm3/year | 8672 | X | X |
| 3. Btwn Isle LMtte & Main Lake | hm3/year | 8861 | X | X |
| 4. Btwn Malletts Bay & Main Lake | hm3/year | 272 | X | X |
| 5. Btwn Shelb Bay & Main Lake | hm3/year | 4816 | 4816 | X |
| 6. Btwn So. Lake & Main Lake | hm3/year | 49427 | X | X |
| 7. Btwn Miss Bay & NE Arm | hm3/year | X | X | 297 |

Based on equations 4 to 8 using the values in Tables B-2 to B-4, the PEI models estimates the input loads of TP of non point sources, point sources and advective and exchange inflows for the Main Lake, Shelburne Bay and Missisquoi Bay lake segments. These estimates are summarized in Table B-5.

**Table B-4: The initial water column TP concentrations for those lake segments contributing advective and exchange flows to the Main Lake, Shelburne Bay and Missisquoi Bay lake segment PEI models. In the PEI models these values define current conditions and, except for the Main Lake, Shelburne Bay and Missisquoi Bay lake segments, remain constant throughout all PEI model runs. These values were derived from data in the 1992 Diagnostic Feasibility Study.**

| Lake Segment | Units | Value |
|---|---|---|
| Main Lake | µg TP/L | 11.79 |
| Burlington Bay | µg TP/L | 13.34 |
| Cumberland Bay | µg TP/L | 13.57 |
| Isle LaMotte | µg TP/L | 12.10 |
| Malletts Bay | µg TP/L | 9.35 |
| Shelburne Bay | µg TP/L | 15.09 |
| South Lake | µg TP/L | 14.58 |
| Missisquoi Bay | µg TP/L | 44.90 |
| North East Arm | µg TP/L | 14.23 |

**Table B-5: Input loadings to the Main Lake, Shelburne Bay and Missisquoi Bay lake segments as computed by the PEI models. These loadings are for time 0, the estimated current loading conditions. All units are kg TP/year.**

| Parameter | Main Lake | Shelburne Bay | Missisquoi Bay |
|---|---|---|---|
| **Non point TP inputs** | | | |
| 1. fr. Agricultural lands | 20,650 | 2,770 | 63,940 (USA)<br>64,950 (CAN) |
| 2. fr. Urban lands | 53,090 | 6,190 | 12,740 (USA)<br>9,230 (CAN) |
| 3. fr. Forest lands | 16,790 | 220 | 4,530 (USA)<br>3,170 (CAN) |
| *TOTALS* | *90,530* | *9,180* | *158,560* |
| **Point source TP inputs** | | | |
| 1. fr. WWTPs | 11,680 | 1140 | 570 (USA)<br>1,370 (CAN) |
| 2. fr. Septic systems | 380 | 40 | 20 (USA)<br>40 (CAN) |
| *TOTALS* | *12,060* | *1,180* | *2,000* |
| **TOTAL INPUT LOAD FROM WATERSHED** | **102,590** | **10,360** | **160,560** |
| **Advective flow inputs** | | | |
| 1. fr. Burl. Bay | 120 | X | X |
| 2. fr. Cumb. Bay | 13,390 | X | X |
| 3. fr. Malletts Bay | 2,540 | X | X |
| 4. fr. Shelb. Bay | 1,190 | X | X |
| 5. fr. So. Lake | 51,310 | X | X |
| *TOTALS* | *68,550* | *0* | *0* |

| **Exchange flow inputs** | | | |
|---|---|---|---|
| 1. fr. Burl. Bay | 39,830 | X | X |
| 2. fr. Cumb. Bay | 122,180 | X | X |
| 3. fr. Isle La Motte | 107,220 | X | X |
| 4. fr. Malletts Bay | 12,300 | X | X |
| 5. fr. NE Arm | X | X | 350 |
| 6. fr. Shelb. Bay | 72,680 | X | X |
| 7. fr. So. Lake | 720,650 | X | X |
| 8. fr. Main Lake | X | 56,780 | X |
| *TOTALS* | *1,075,040* | *56,780* | *350* |
| **TOTAL INPUT LOAD FOR SEGMENT** | **1,177,630** | **67,140** | **160,910** |

When model estimates of total loading from contiguous watersheds (nonpoint source + point source) are compared to loading estimates from Medelie and Smeltzer (2004) (Table B-4) differences are seen. The PEI model estimates vs. Medelie and Smeltzer are 100.7 mt/yr vs 230.9 mt/yr,10.4 mt/yr vs. 5.1 mt/yr and 160.6 mt/yr vs.169.4 mt/yr for the Main Lake, Shelburne Bay and Missisquoi Bay lake segments, respectively. These three lake segments are strikingly different with regard to their source of phosphorus input; in Missisquoi Bay essentially100% of the total input load comes directly from the watershed whereas only about 15% and 8% of the total input loads comes directly from the Shelburne Bay and Main Lake segment watersheds, respectively.

**D. Phosphorus outputs from the lake segment**

In the PEI model phosphorus is output from the lake segment in advective and exchange flows. The value of these TP output fluxes is calculated as per equations 7 and 8 except that the TP $Conc_i$ term is replaced by the lake segment water column TP concentration as calculated by equation 3 in the PEI model. Thus, the values of these output fluxes change over time in proportion to change in the TP concentration in the lake segment. The time 0 advective and exchange TP outflows from the Main Lake, Shelburne Bay and Missisquoi Bay lake segments as estimated by the PEI model are summarized in Table B-6.

**Table B-6: Summary of average annual advective flow and exchange flow outputs for the Main Lake, Shelburne Bay and Missispuoi Bay lake segments. The values in this table are for time 0 and represent currrent conditions as calculated by the PEI model. Calculations are based on flow rates from Smeltzer et al. (1999).**

| **Parameter** | **Units** | **Main Lake** | **Shelburne Bay** | **Missisquoi Bay** |
|---|---|---|---|---|
| | | <u>Advective Outflows</u> | | |
| 1. fr. Main Lake | hm3/year | 9402 | X | X |
| 2. fr. Miss Bay | hm3/year | 2039 | X | X |
| 3. fr. Shelb Bay | hm3/year | 79 | X | X |
| <u>Exchange Outflows</u> | | | | |
| 1. Btwn Burl Bay & Main Lake | hm3/year | 2986 | X | X |
| 2. Btwn Cmbld Bay & Main Lake | hm3/year | 8672 | X | X |
| 3. Btwn Isle Lmtte & Main Lake | hm3/year | 8861 | X | X |
| 4. Btwn Malletts Bay & Main Lake | hm3/year | 272 | X | X |
| 5. Btwn Shelb Bay & Main Lake | hm3/year | 4816 | 4816 | X |
| 6. Btwn So. Lake & Main Lake | hm3/year | 49427 | X | X |
| 7. Btwn Miss Bay & NE Arm | hm3/year | X | X | 297 |

**E. Phosphorus cycling or internal loadings**

Phosphorus is continually being exchanged between the water column and the lake bottom sediments. In some lake segments this exchange may be a very significant factor in how water column phosphorus levels change over time in response to management while in other lake segments it is not. Phosphorus is removed from the water column when phosphorus-laden particulates settle to the sediments on the bottom of the segment. The velocity of particle descent is controlled by particle size, shape and density, water viscosity and the level of water turbulence through which the particle settles. Large, dense particles (sand) settle rapidly; small dense particles (silt and clay) settle slowly as do particulates of organic debris. Internal water column turbulence from seiches, seasonal turnovers and wind action reduce this settling velocity. In shallow segments like Missisquoi Bay, wind induced turbulence and turbulence from boats usage is likely an important factor influencing particulate settling. HydroQual, Inc. (1999) lists numerous phosphorus sedimentation rates that were measured in the late 1990's for various locations in Lake Champlain. Table B-7 summarizes those values reported for the Main Lake, Shelburne Bay and Missisquoi Bay lake segments.

**Table B-7. Summary TP sedimentation rates and estimated TP sedimentation fluxes for the Missisquoi Bay, Shelburne Bay and Main Lake lake segments (HydroQual, Inc. 1999).**

| Lake Segment | TP Sedimentation Rates | | Est. Sedimentation Flux of TP**** | | |
|---|---|---|---|---|---|
| | Units | Value | Units | Value*** | % of Total Input Load from Watershed |
| Missisquoi Bay | g TP/sq. m/yr | 1.09* | kg TP/yr | 98,030 | 61 |
| Main Lake | g TP/sq. m/yr | 1.138** | kg TP/yr | 323,030 | 106 |
| Shelburne Bay | g TP/sq. m/yr | 0.78*** | kg TP/yr | 10,950 | 321 |
| * (avg. of Sta's. 48, 50, 51, 52), ** (from Sta.19), *** (from Sta. 21) | | | | | |
| **** calculated as the product of TP sedimentation rate and lake segment surface area | | | | | |

The phosphorus sedimentation rates in Table B-7 represent but a snapshot in time. In Missisquoi Bay, the estimated sedimentation flux of TP is 61% of the total input TP load from the watershed and because the total input TP load from the watershed is essentially 100% of the total TP input flux to the segment, this suggests that the physical characteristics of Missisquoi Bay allows around 40% of the entering phosphorus (the soluble phosphorus and smaller particulates) to pass through. Because this sedimentation is occuring unhindered in dilute suspension, changes in the input TP loading to the Bay would not be expected to change the proportion of the entering TP that passes through the Missisquoi Bay lake segment.

However, this logic appears to be inappropriate for the Shelburne Bay and Main Lake segments because of their high degree of hydraulic connectivity to adjacent lake segments. In both cases the estimated sedimentation flux of TP exceeds the total input TP load from their contiguous watersheds indicating that TP in suspended sediments in the advective and exchange inflows settle out along with some undetermined fraction of the TP loads from the watersheds. Under these conditions it appears likely that the physical characteristics of these two lake segments would allow continued settling of TP at rates that change little even when management implementations might alter the TP inputs from the watershed. Thus, in the PEI models for the Main Lake and Shelburne Bay the rate of input of phosphorus settling to the sediment remains constant at *about* the current levels in Table B-7.

At the same time phosphorus settles to the bottom sediments, the phosphorus trapped in these sediments moves back into the water column via at least three pathways. (a) Some macrophytes grow by taking up phosphorus through their root structures, with most being incorporated into plant biomass while some may be leached from the plant directly into the water column. When the plant dies and ultimately decays some of this phosphorus then may enter the water column while the remainder is reincorporated back as debris into the sediment. There a few data to quantify this pathway for Lake Champlain. (b) Some phosphorus in the sediment resides in the interstitial pore waters and may diffuse upward into the water column. Diffusion always moves phosphorus from areas of high concentration (sediment pore water) to lower concentrations (water column). Diffusion rate is directly proportional to the magnitude of the concentration gradient, however, these rates are substantially influenced by dissolved oxygen levels in the sediment, sediment chemistry, animal burrowing and by turbulence levels at the sediment-water interface that can modify the gradient. In lake ecosystems diffusion and diffusion-like processes are complex and not completely understood. (c) Lake bottom sediments are matrices of smaller particulates that form a mucky and somewhat fluffy mass with little internal structure. Water turbulence at the sediment-water interface can disrupt this matrix to resuspend particles in the water column, a phenomenon that is likely more important in shallow lake segments such as Missisquoi Bay than in deeper lake segments.

HydoQual, Inc. (1999) studied the rates of diffusion and resuspension of phosphorus from Lake Champlain sediments to the overlying water column. Their data make it difficult to assess differences among the various lake segments, however, their overall lake-wide average soluble P sediment to water exchange rate is 1.03 g TP/sq.m/yr (SD +/- ~50%) (HydoQual 1999, Table VII-4). Since this rate accounts for only soluble phosphorus, actual prevailing rates would likely be *somewhat higher* to account for resuspension of particulate phosphorus. This diffusion rate defines the conditions that existed only at time of measurement and can change as the character of the sediment changes. Generally, the rate of diffusion is directly related to amount of phosphorus in the sediment given similar levels of dissolved oxygen and sediment chemistry.

The algorithms in the PEI models describe phosphorus cycling from the lake bottom sediments to the water column require the input of several parameters including: active depth of sediment from which phosphorus moves into the water column, the bulk density of the bottom sediments, the initial TP assay of the sediments and a rate constant defining what fraction of the sediment phosphorus mass moves into the water column each year. In our models an active sediment depth of 150 cm (about 6 inches) was assumed as were values within reported ranges (HydoQual 1999) for bulk density and sediment TP assay. The model computed the mass of phosphorus in the sediment layer based on the values assigned to these parameters. The mass of TP in the sediments was then divided into the average rate at which soluble P was exchanged from the sediment to the water column yielding an estimate of an exponential rate constant (the fraction of TP in the sediment that moves into the water column per year). This calculation was carried out for each lake segment. These values of this exchange rate constant are summarized in Table B-8. The values used in the PEI models were higher than those calculated in order to achieve calibration, however, these higher values may also account for resuspension effects.

**Table B-8: Summary of parameters used in the PEI models to calculate the flux of phosphorus that moves from the lake segment bottom sediments to the overlying water column.**

| Parameter | Units | Main Lake | Shelburne Bay | Missisquoi Bay |
|---|---|---|---|---|
| Sediment Bulk Density* | $g/cm^3$ | 400 | 700 | 500 |
| TP Assay of Sediment | mg TP/g | 1.00 | 1.50 | 1.02 |
| Sediment Depth | cm | 150 | 150 | 150 |
| Yearly % of sediment TP exchanged to water column | % yr-1 | | | |

| | | | |
|---|---|---|---|
| a. calculated as per text above | 0.17 | 0.12 | 0.13 |
| b.used in PEI models | 2.16 | 2.16 | 1.44 |
| *reported range for the lake is 156 to 551 g/cm$^3$ (HydoQual1999) | | | |

**F. Model calibration and sensitivity assessments.**

Each PEI model was calibrated so that the time 0 value calculated by the model matched the average TP value for the lake segment derived from 1990 - 2000 monitoring data. These values are 11.79 μg TP/L, 15.09 μg TP/L and 44.9 μg TP/L for the Main Lake, Shelburne Bay and Missisquoi Bay, respectively, and serve as the starting point for all trend lines generated by the models. During calibration all TP inputs from the watershed and advective and exchange flows remained constant at levels determined by TP export coefficients, land areas, advective and exchange flow rates and average TP concentrations of the various lake segments. Only the parameter values of (a) sediment depth, (b) sediment TP assay, (c) sediment bulk density, (d) fraction of input TP that settles out and (e) yearly percent of sediment TP that enters the water column were changed. Values for these parameters were adjusted so that, within the 5 parameter set, each value fell as close as possible to values from the literature (as discussed in the above sections of this Appendix). Calibration was achieved when the time 0 water TP concentration computed by the model closely matched the average value from monitoring data. More than one set of values for these parameters can lead to such a calibration; we chose a set of values that appeared to fairly represent reported known current conditions in the lake segments.

The PEI model for the Missisquoi Bay lake segment is employed here to show the sensitivity of a primary model output, the TP concentration of the Missisquoi Bay waters, to change in selected input parameter values. For each sensitivity analysis four simulation runs were made, one for each of four different values of the input parameter. These values ranged generally from about -100% to +170% of the calibrated value. The time 0 water TP concentration computed in each simulation run was then compared to the average TP concentration of 44.9 μg TP/L derived from water quality monitoring data that defines the initial, or current, TP concentration of the Bay waters.

A total of five sensitivity analyses were conducted, one for each of the following parameters. They are:

(a) the depth of sediment through which active diffusion is assumed to occur,
(b) the average TP assay of the sediments on the bottom of the lake segment,
(c) the average bulk density of the sediments on the bottom of the lake segment,
(d) fraction of the TP inputs from the watershed that settle out upon entry into the lake segment
(e) the yearly percentage of the TP in the sediment that enters the water column via diffusion and resuspension.

The results of these sensitivity analyses are shown on Figure B-2, a plot of % change in TP concentration vs. the % change made in the input parameters. On this figure all plots run through coordinate 0,0, the point at which the model is calibrated. At this point the calibrated values of the input parameters are input and the model computes a TP concentration of 44.9 µg TP/L, the time 0 calibrated value for Missisquoi Bay. At all other points along each plot the value of a particular input parameter is changed, as indicated by 'per cent change in model parameter' and the model then computes a value of TP concentration as represented by 'per cent change in TP concentration'.

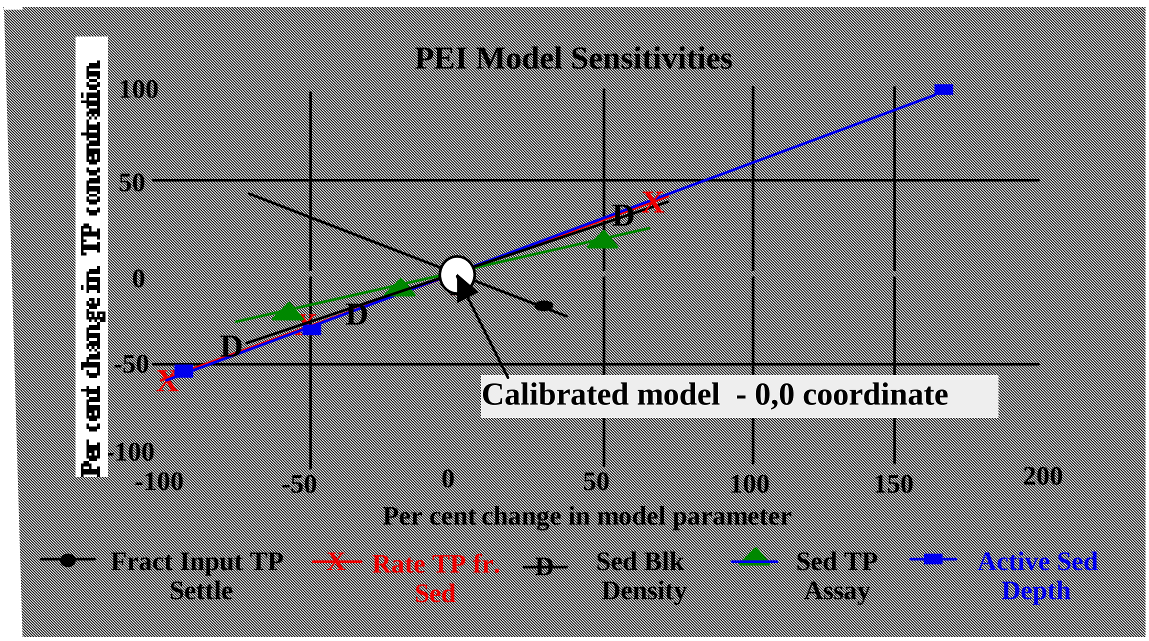


**Figure B-2: Summary of the sensitivity of the TP concentration in Missisquoi Bay waters as calculated by the PEI model over a range of model parameter values. At the 0,0 coordinate, the model is calibrated to current conditions.**

These analyses indicate that the calculated TP concentrations are directly related to the values of sediment depth, sediment TP assay, sediment bulk density and percentage of sediment TP exchange. For all these parameters a 10% increase results in about a 5% increase in TP concentration. The calculated TP concentration is, however, inversely related to the fraction of the TP from the watershed that settles out, that is, when more TP settles out less is in the water. A 10% increase in settling results in about a 5% decrease in TP concentration.

## G. Input Parameters for the PEI models

A complete listing of the parameters required by the PEI models for the Main Lake, Shelburne Bay and Missisquoi Bay lake segments is given in Table B-9. The table includes the units and the calibration values for each parameter as used to generate all results presented in the body of this report.

**Table B-9: A complete listing of all the parameters that must be defined in the PEI models for the Main Lake, Shelburne Bay and Missisquoi Bay segments.**

| **checked 2/9/05** | | **Main Lake** | **Shelburne Bay** | **Missisquoi Bay** |
|---|---|---|---|---|
| **Input Parameters** | **Units** | **Value** | **Value** | **Value** |
| **Advective Flows** | | | | |
| Burlington Bay to Main Lake | hm^3/yr | 9 | NA | NA |
| Cumberland Bay to Main Lake | hm^3/yr | 950 | NA | NA |
| Main Lake to Isle la Motte | hm^3/yr | 9402 | NA | NA |
| Malletts Bay to Main Lake | hm^3/yr | 1315 | NA | NA |
| Shelburne Bay to Main Lake | hm^3/yr | 79 | 79 | NA |
| South Lake to Main Lake | hm^3/yr | 3519 | NA | NA |
| Missisquoi Bay to Northeast Arm | hm^3/yr | NA | NA | 2039 |
| **Exchange Flows** | | | | |
| Between Burlington Bay and Main Lake | hm^3/yr | 2986 | NA | NA |
| Between Cumberland Bay and Main Lake | hm^3/yr | 8672 | NA | NA |
| Between Isle la Motte and Main Lake | hm^3/yr | 8861 | NA | NA |
| Between Malletts Bay and Main Lake | hm^3/yr | 272 | NA | NA |
| Between Shelburne Bay and Main Lake | hm^3/yr | 4816 | 4816 | NA |
| Between South Lake and Main Lake | hm^3/yr | 49427 | NA | NA |
| Between Missisquoi Bay and Northeast Arm | hm^3/yr | NA | NA | 297 |
| **Land Use and Land Use Change** | | | | |
| Initial Agricultural Land area - Main Lake | ha | 49180 | NA | NA |
| Initial Agricultural Land area - Shelburne Bay | ha | NA | 6600 | NA |
| Initial Agricultural Land area - Missisquoi Bay USA | ha | NA | NA | 37544 |
| Initial Agricultural Land area - Missisquoi Bay Can | ha | NA | NA | 39228 |
| Agricultural Land Change Fraction - Main Lake | dimensionless | 0.5 | NA | NA |
| Agricultural Land Change Fraction - Shelburne Bay | dimensionless | NA | 0.5 | NA |
| Agricultural Land Change Fraction - Missisquoi Bay USA | dimensionless | NA | NA | 0.5 |
| Agricultural Land Change Fraction - Missisquoi Bay Can | dimensionless | NA | NA | 0.5 |
| Agricultural TP export coefficient - Main Lake | kg TP/ha/yr | 0.42 | NA | NA |
| Agricultural TP export coefficient - Shelburne Bay | kg TP/ha/yr | NA | 0.42 | NA |
| Agricultural TP export coefficient - Missisquoi Bay USA | kg TP/ha/yr | NA | NA | 1.73 |

| | | | | |
|---|---|---|---|---|
| Agricultural TP export coefficient - Missisquoi Bay Can | kg TP/ha/yr | NA | NA | 1.63 |
| *Agricultural TP export coefficient adjustment - Main Lake | dimensionless | 1 | NA | NA |
| *Agricultural TP export coefficient adjust- Shelburne Bay | dimensionless | NA | 1 | NA |
| *Agricultural TP export coefficient adjust - Missisquoi Bay USA | dimensionless | NA | NA | 1 |
| *Agricultural TP export coefficient adjust - Missisquoi Bay Can | dimensionless | NA | NA | 1 |
| Initial Forested Land area - Main Lake | ha | 419680 | NA | NA |
| Initial Forested Land area - Shelburne Bay | ha | NA | 5578 | NA |
| Initial Forested Land area - Missisquoi Bay USA | ha | NA | NA | 113528 |
| Initial Forested Land area - Missisquoi Bay Can | ha | NA | NA | 79349 |
| Forested Land Change Fraction - Main Lake | dimensionless | 0.5 | NA | NA |
| Forested Land Change Fraction - Shelburne Bay | dimensionless | NA | 0.5 | NA |
| Forested Land Change Fraction - Missisquoi Bay USA | dimensionless | NA | NA | 0.5 |
| Forested Land Change Fraction - Missisquoi Bay Can | dimensionless | NA | NA | 0.5 |
| Forestry TP export coefficient - Main Lake | kg TP/ha/yr | 0.04 | NA | NA |
| Forestry TP export coefficient - Shelburne Bay | kg TP/ha/yr | NA | 0.04 | NA |
| Forestry TP export coefficient - Missisquoi Bay USA | kg TP/ha/yr | NA | NA | 0.04 |
| Forestry TP export coefficient - Missisquoi Bay Can | kg TP/ha/yr | NA | NA | 0.04 |
| *Forestry TP export coefficient adjust - Main Lake | dimensionless | 1 | NA | NA |
| *Forestry TP export coefficient adjust - Shelburne Bay | dimensionless | NA | 1 | NA |
| *Forestry TP export coefficient adjust - Missisquoi Bay USA | dimensionless | NA | NA | 1 |
| *Forestry TP export coefficient adjust - Missisquoi Bay Can | dimensionless | NA | NA | 1 |
| Initial Urban Land area - Main Lake | ha | 35390 | NA | NA |
| Initial Urban Land area - Shelburne Bay | ha | NA | 4125 | NA |
| Initial Urban Land area - Missisquoi Bay USA | ha | NA | NA | 8491 |
| Initial Urban Land area - Missisquoi Bay Can | ha | NA | NA | 6153 |
| Urban Land Area Consumption Ratio - Main Lake | ha/capita | 0.18 | NA | NA |
| Urban Land Area Consumption Ratio - Shelburne Bay | ha/capita | NA | 0.18 | NA |
| Urban Land Area Consumption Ratio - Missisquoi Bay USA | ha/capita | NA | NA | 0.18 |
| Urban Land Area Consumption Ratio - Missisquoi Bay Can | ha/capita | NA | NA | 0.18 |
| Urban TP export coefficient - Main Lake | kg TP/ha/yr | 1.5 | NA | NA |
| Urban TP export coefficient - Shelburne Bay | kg TP/ha/yr | NA | 1.5 | NA |
| Urban TP export coefficient - Missisquoi Bay USA | kg TP/ha/yr | NA | NA | 1.5 |
| Urban TP export coefficient - Missisquoi Bay Can | kg TP/ha/yr | NA | NA | 1.5 |
| *Urban TP export coefficient adjust - Main Lake | dimensionless | 1 | NA | NA |
| *Urban TP export coefficient adjust- Shelburne Bay | dimensionless | NA | 1 | NA |
| *Urban TP export coefficient adjust- Missisquoi Bay USA | dimensionless | NA | NA | 1 |
| *Urban TP export coefficient adjust- Missisquoi Bay Can | dimensionless | NA | NA | 1 |

**Demographic and Demographic Change**

| | | | | |
|---|---|---|---|---|
| Population - Main Lake | no. people | 204860 | NA | NA |
| Population - Shelburne Bay | no. people | NA | 20000 | NA |
| Population - Missisquoi Bay USA | no. people | NA | NA | 10000 |

| | | | | |
|---|---|---|---|---|
| Population - Missisquoi Bay Can | no. people | NA | NA | 24000 |
| Population Change Rate - Main Lake | %/yr | 1.2 | NA | NA |
| Population Change Rate - Shelburne Bay | %/yr | NA | 2.4 | NA |
| Population Change Rate - Missisquoi Bay USA | %/yr | NA | NA | 1.2 |
| Population Change Rate - Missisquoi Bay Can | %/yr | NA | NA | 1.2 |
| Sewered Fraction of Population - Main Lake | dimensionless | 0.43 | NA | NA |
| Sewered Fraction of Population - Shelburne Bay | dimensionless | NA | 0.43 | NA |
| Sewered Fraction of Population - Missisquoi Bay USA | dimensionless | NA | NA | 0.43 |
| Sewered Fraction of Population - Missisquoi Bay Can | dimensionless | NA | NA | 0.43 |
| **Point Source & Subsurface Parameters** | | | | |
| Per capita watstewater flow | gal/capita/day | 120 | 120 | 120 |
| Effluent TP Standard - USA | mg TP/L | 0.8 | 0.8 | 0.8 |
| Effluent TP Standard - Canada | mg TP/L | NA | NA | 1 |
| TP Conc. of Subsurface Drainage | mg TP/L | 0.02 | 0.02 | 0.02 |
| **Lake Segment Characteristics** | | | | |
| Surface area - Main Lake | ha | 41414 | NA | NA |
| Surface area - Shelburne Bay | ha | NA | 962 | NA |
| Surface area - Missisquoi Bay | ha | NA | NA | 8994 |
| Volume - Main Lake | cu. meters | 16787*10^6 | NA | NA |
| Volume - Shelburne Bay | cu. meters | NA | 140*10^6 | NA |
| Volume - Missisquoi Bay | cu. meters | NA | NA | 205 *10^6 |
| Sediment average depth - Main Lake | meters | 0.15 | NA | NA |
| Sediment average depth - Shelburne Bay | meters | NA | 0.15 | NA |
| Sediment average depth - Missisquoi Bay | meters | NA | NA | 0.15 |
| **Sediment Bulk density - Main Lake | kg/cu. meter | 400 | NA | NA |
| **Sediment Bulk density - Shelburne Bay | kg/cu. meter | NA | 700 | NA |
| **Sediment Bulk density - Missisquoi Bay | kg/cu. meter | NA | NA | 500 |
| **Initial TP Assay of Sediment - Main Lake | mg TP/kg dry weight | 1000 | NA | NA |
| **Initial TP Assay of Sediment - Shelburne Bay | mg TP/kg dry weight | NA | 1500 | NA |
| **Initial TP Assay of Sediment - Missisquoi Bay | mg TP/kg dry weight | NA | NA | 1020 |
| TP Sedimentation Rate - Main Lake | g TP/yr/m^2 water | 1.138 | NA | NA |
| TP Sedimentation Rate - Shelburne Bay | g TP/yr/m^2 water | NA | 0.8 | NA |
| TP Sedimentation Rate - Missisquoi Bay | fraction of Input TP | NA | NA | 0.65 |
| **TP Resuspension Rate constant- Main Lake | %/month Sediment P | 0.18 | NA | NA |
| **TP Resuspension Rate - Shelburne Bay | %/month Sediment P | NA | 0.18 | NA |
| **TP Resuspension Rate - Missisquoi Bay | %/month Sediment P | NA | NA | 0.12 |
| Water TP standard - Main Lake | micro-gms TP/L | 10 | NA | NA |
| Water TP standard - Shelburne Bay | micro-gms TP/L | NA | 14 | NA |
| Water TP standard - Missisquoi Bay | micro-gms TP/L | NA | NA | 25 |
| **Initial TP Concentrations** | | | | |
| Initial Main Lake TP Conc. | micro-gms TP/L | 11.79 | 11.79 | NA |

| | | | | |
|---|---|---|---|---|
| Initial Burlington Bay TP Conc. | micro-gms TP/L | 13.34 | NA | NA |
| Initial Cumberland Bay TP Conc. | micro-gms TP/L | 13.57 | NA | NA |
| Initial Isle la Motte TP Conc. | micro-gms TP/L | 12.1 | NA | NA |
| Initial Mallettes Bay TP Conc. | micro-gms TP/L | 9.35 | NA | NA |
| Initial Shelburne Bay TP Conc. | micro-gms TP/L | 15.09 | 15.09 | NA |
| Initial South Lake TP Conc. (Otter Creek) | micro-gms TP/L | 14.58 | NA | NA |
| Initial Missisquoi Bay TP Conc. | micro-gms TP/L | NA | NA | 44.9 |
| Initial Northeast Arm TP Conc. | micro-gms TP/L | NA | NA | 14.23 |

**Notes** * TP export coeff adjust to emulate changed Levels of BMPs. ** primary parameters used in model calibration