**APPENDIX A – KEY FINDINGS, LAKE CAPACITY SUMMARIES, AND RECOMMENDATIONS FROM 1993 CHANDOS TOWNSHIP LAKE CARRYING CAPACITIES**

In the following pages, key findings including lake carrying capacity summaries and recommendations from our 1993 report **Chandos Township Lake Carrying Capacities** are reproduced.

## Key Findings of the 1993 Report for Chandos Lake

- The Chandos Lake basin above its outlet is 8,696 hectares (ha). The basin was almost entirely within the then Township of Chandos, and included all of the municipality's significant water bodies. Chandos Lake, at 1,659 ha was by far the largest of these.

- There were an estimated 1,007 residential properties on Chandos Lake, with nine tenths of these (i.e., approximately 900) being seasonal. Shoreline tourist establishments accounted for 23 cabins, 10 campsites, and 80 youth camp places.

- Concentrations of chlorophyll *a* in Chandos Lake were predicted to have increased by less than 1.0 microgram per litre (µg/L) or part per billion over the last 200 years, primarily as a result of shoreline development. The exception was Gilmour Bay where concentrations increased from a pre-development level of 1.2 µg/L to a 1991 concentration of 4.1 µg/L.

- Optimal lake trout habitat (i.e., the percent of total lake volume with temperatures $\leq 10°$C and dissolved oxygen concentrations $\geq 6.0$ milligrams per litre [mg/L] on August 31 in any given year) was predicted to have declined by about two thirds in Gilmour Bay, and one quarter in the rest of Chandos Lake over the last 200 years, again primarily as a result of shoreline development.

- At most, assuming development to the minimum frontages permitted by the Township's zoning bylaw, the six significant remaining stretches of undeveloped privately owned shoreline on Chandos Lake could accommodate 33 shoreline lots.

- 2,318 boats were based at Chandos Lake shoreline households and commercial operations. The average Chandos Lake household owned 2.42 boats, and 93% of households owned at least one boat.

- Boats resident on Chandos Lake required an estimated 33% of the lake's usable surface at peak boating times. Of the four boating observational areas studied in detail, West Bay Narrows was by far the most crowded, and was frequently overcapacity.

**Lake Capacities Summary**

- Shoreland development significantly impaired **lake trophic state** (i.e., determined by concentrations of phosphorus and chlorophyll *a* and water clarity) and **lake trout habitat** only in Gilmour Bay. However, lake trophic state could be a significant constraint to any development that would further increase phosphorus loads to Chandos Lake, depending on the trophic state objective that shoreline residents wished to set, and the degree of change that the community would be prepared to permit.

- **Shoreland development capability** (i.e., physical terrain conditions) did not significantly limit development based on detailed field work. The number of additional detached shoreline residential lots that could be created was quite small, because Chandos Lake's shoreline was by and large already developed or subdivided. However, the amount of more intensive shoreline and backshore development or redevelopment that the shorelands could physically accommodate far exceeded not only demand, but also the level of development that would be environmentally, aesthetically, or socially acceptable to the lake's community.

- The **sports fisheries** of Chandos Lake were not being over-exploited. Any over-exploitation of fisheries should be controlled by regulating angling, rather than limiting development.

- **Boating** was not a significant constraint to development generally. However, local congestion problems might require both development control and recreational management solutions.

- Given the above, **lake trophic state** was determined to be the primary factor that would limit development, or dictate more stringent conditions for permitting shoreline development.

## Recommendations of the 1993 Report

Based on the above findings/analyses, municipal planning policy directions were subsequently recommended that would ensure that lake capacity considerations would be fully taken into account in planning and development decisions, thereby reflecting the environmental concerns of shoreline residents. The policy directions were expressed in general form. The Township of Chandos and its planning consultants were to be responsible for translating agreed policy directions into official plan and zoning bylaw provisions, and deciding on the specific mechanisms to be used (e.g., land use designations, zoning bylaw provisions, site plan control, property standards bylaw provisions, etc.).

### *Water Quality*

Based on questionnaire responses, and an emerging climate favoring protection of recreational lake water quality, it was obvious that the trophic status of Chandos Lake should not be further reduced from 1991/1992 conditions. In a letter of May 28, 1992, Jim O'Shea, President of the Chandos Lake Property Owners' Association (CLPOA) advised that the Association's executive committee had taken the position that, ". . . water quality should not be allowed to deteriorate . . . Every effort should be made to maintain the present standards and if possible enhance the present level of water quality for Chandos Lake." This position/direction resulted in the following recommendations.

1. The Township of Chandos Official Plan should state as a general objective that recreational water quality in the Township's lakes be maintained or enhanced. Recreational water quality would be defined as the long term average of mean summertime chlorophyll *a* concentrations.

2. All applicants within the Chandos Lake basin should be required to demonstrate that:

    - for all new development (i.e., the creation of new lots and the development of existing vacant lots), phosphorus loads would not exceed background loads previously existing; and

    - in case of re-development (extensions to and enlargements of existing buildings and the re-development of existing developed lots), phosphorus loads would be reduced from those previously existing.

3. Any applicant within the Chandos Lake basin should be required to commission independent professional studies of trophic state and lake trout habitat impacts where deemed appropriate by the Township. As well, based on information provided by shoreline residents through questionnaire returns, the Official Plan should no longer require that lake capacity studies assume that permanent residences would be developed on vacant lots.

4. New and redeveloped buildings, campsites, and sewage systems on Chandos Lake should be subject to the following.

    - Minimum horizontal setbacks from water of 30 metres (m) to 90 m should be applied to individual sites to reflect their phosphorus retention capabilities in accordance with guidelines proposed in Section 5.3 of the 1993 report. The setbacks should apply as follows:

        – buildings (except marina buildings), campsites, and sewage systems should be set back accordingly (at the time, the Official Plan required 22 m for buildings and 30 m for sewage systems);

        – vegetation within the setbacks should be disturbed as little as possible consistent with passage, safety, and provisions of views and ventilation (this proposal elaborated on a provision already existing in the Official Plan).

        – tile field fill should be installed with and regularly maintained to a phosphorus retention capability of at least 6.0 milligrams phosphorus per 100 grams of soil (mg P/100 g) (again, this proposal elaborated on a provision already existing in the Official Plan).

    It was proposed that these conditions would apply to the following.

        - To development on existing vacant lots and all types of redevelopment, wherever possible. On lots whose locations and dimensions made it impossible to meet the conditions, additional remedial measures (such as reestablishment of natural vegetation) to intercept and retain phosphorus should be required through site plan control.

- To existing developed lots not subject to planning and development approvals, by seeking the cooperation of landowners in achieving the objectives of these conditions by voluntarily relocating and upgrading their septic tank-tile field systems, renaturalizing their shorelines, etc.

5. Provided the first four recommendations above were implemented, all existing conditions specific to Gilmour Bay should be removed from the Official Plan.

6. Shoreland residential development alternatives to single tier detached residences should be encouraged, provided that these development forms are socially acceptable to the community. If the Township of Chandos wished to contemplate such developments, it should consider planning policies to encourage proponents to redevelop already developed areas, rather than locate in undeveloped natural environments. The specific policies are as follows.

   - In order to minimize per unit environmental and recreational impacts of shoreland residential development, alternatives to single tier detached residences should be encouraged, provided that these development forms are socially acceptable to the community. This would be done by:

     – providing reasonable and flexible planning standards for cluster residential resort development on properties with lake frontage, with considerably smaller lot sizes than currently required for single detached residences, provisions for common shoreline depth and for common area as a percentage of total area, and requirements for associated recreational facilities; and

     – contemplating the development of stand-alone municipal communal water and sewage systems to service these types of developments.

### *Resource Protection*

7. In addition to having resource protection benefits, it was recognized that the following recommendation contributed to water quality improvements.

   - If the Township wished to contemplate larger scale new shoreland developments as described above, it should also consider planning policies to encourage proponents to re-develop areas currently considered "undeveloped" by the market, such as commercial cabin operations, rather than locate them in undeveloped natural environments.

8. Any applicant within 300 m of a lake should be required to commission independent professional studies of impacts upon the following where deemed appropriate by the Township, and where it was reasonable to anticipate that undue impacts could occur:

   – wetlands;
   – fish and wildlife habitat; and
   – other significant natural and culture heritage features.

9. Although the Crown is not legally bound by the Official Plan, all Crown shoreland on Chandos Lake should be included in an open space designation (which did not exist in the Township's Official Plan) that prohibits development.

### *Boating*

10. Any applicant for shoreline commercial development or creation of a new shoreline lot or lots should be required to commission independent professional studies of boating impacts where deemed appropriate by the Township.

11. Because of the high degree of boating congestion in West Bay Narrows, there should be no further creation of shoreline lots, and it should be policy (requiring the cooperation of the Ministry of Natural Resources (MNR); see Recommendation 14) that there be no further water lot dispositions

for berthing in excess of basic riparian rights, on the shorelines of West Bay Narrows within Lots 11, 12 and 13, Concession VII and Lots 11, 12 and 13, Concession X Chandos.

### *Planning and Development Policy Implementation*

These recommendations deal with issues that would need to have been addressed in implementing the preceding planning policies, as well as matters where the cooperation of other agencies with the Township of Chandos would have been required to ensure that lake capacities for development and recreational use are not exceeded.

12. In the review of water quality undertaken as part of the 1993 report, considerable gaps in the information base were noted. For example, good data were available for the main basin of Chandos Lake, but not for Gilmour Bay. Better data would have permitted better calibration of the Lake Trophic State model, would have minimized the imprecision associated with year-to-year variation in biological conditions, and would have allowed for long term trends to be more clearly identified. The specific recommendation read as follows.

> "The chlorophyll *a* Secchi disc self-help program should be expanded to sample the main basin and Gilmour Bay of Chandos Lake, on a regular, frequent cycle. Each year's sampling of each water body should include a sufficient number of samplings throughout the summer, and on Chandos Lake, a sufficient number of sampling locations, to ensure that the data collected are representative of the water body's biological conditions. These efforts should be coordinated by the Township and the Chandos Lake Property Owners' Association, in consultation with the Crowe Valley Conservation Authority and/or the Ministry of Environment."

The report then went on to say that a commitment to maintaining lake trophic states at or below present levels would require more than controls on development, as recommended above. It would also require reductions in per unit phosphorus loads from existing and new development, to respond to the continuing demand for summer and year-round living on recreational lakes. With regard to reducing phosphorus loads from existing development, it should be kept in mind that efforts directed at the phosphorus contributors with the lowest standards would achieve the greatest reductions. The higher the standard of existing development, the harder it would be to achieve significant reductions.

The planning policy directions recommended above would have contributed to reducing phosphorus loads, but equal consideration would have had to be given to a number of additional initiatives involving not only the Township but also the Crowe Valley Conservation Authority, the Ministries of the Environment (MOE), MNR, and (then) Agriculture and Food (MAF), and the Peterborough County-City Health Unit.

- Retroactively imposing private services standards, requiring owners of existing septic tank-tile field systems to upgrade to current standards, if inspection suggested that those systems were not performing up to the levels achievable with new installed systems.

- Raising private services standards, including requiring that tile field fill be installed with and regularly maintained to a phosphorus retention capability of at least 6 mg/100 g of soil, and that to the greatest extent possible, tile fields be located in accordance with the setback guidelines recommended in the 1993 report.

- Promoting and encouraging the use of low impact alternatives to septic tank-tile field systems, such as composting toilets and grey water leaching pits.

- Strengthening and enriching existing programs to encourage and promote the re-naturalization of shorelines on developed properties.

- Restricting the use of fertilizers and other phosphorus-rich substances within specified distances of shorelines.

- Developing and implementing strategies to control and reduce agricultural contributions of phosphorus to the lakes and their tributaries.

As explained in the 1993 report, such initiatives would call on existing shoreline residents as well as the proponents of new shoreland development to make fundamental changes in their stewardship of Chandos Lake basin shorelands. Traditional land clearing practices and the transfer of conventional urban landscapes to a lakeshore setting would have to be foregone. Instead, the principles of landscape naturalization and of on-site retention and treatment of pollutants would be at the heart of a new stewardship ethic for shorelands. These changes would represent a whole new way of using and developing recreational lakeshores, and would not be easy to implement. They would not be achieved overnight, nor would their benefits be immediately apparent, especially to those shoreline residents who would have to invest many thousands of dollars to improving their shorelines and upgrading their sewage systems. As well, the complex and unwieldy character of many of the problems that our recreational lakes are now facing, encompassing a wide range of environmental, economic, and social issues, underscores the need for coordination and cooperation among resource management agencies, shore property owners, and all others concerned with the future of our lakes.

The last three recommendations focused on boating, and were as follows.

13. The Township should seek the cooperation of the MNR as owner of the bed of Chandos Lake in implementing Recommendation 11 regarding the limitation of dispositions on the narrowest portion of West Bay Narrows.

14. The Chandos Lake observation area and shoreline boat counts and boating capacity analyses undertaken for the study should be repeated on a comparable basis when warranted by significant changes in shoreline development, in patterns of boat ownership or boating activity, or in other factors that would influence boating capacity conditions on the lake.

15. The Township should initiate the following amendments to the Boating Restriction Regulations under the *Canada Shipping Act*.

    - Portion of West Bay Narrows less than 300 m wide (i.e., this is approximately the same area as proposed for lot creation and water lot disposition control in Recommendation 11): wake and 10 km/h speed controls.

    - Portion of Gilmour Bay Narrows less than 200 m wide: wake and 10 km/h speed controls.

    - Gilmour Bay south of above portion of Gilmour Bay Narrows: 10 km/h speed control.

    These would be in addition to the Township's 1992 application for a prohibition of all motorboats up to 150 m offshore of Sandy Beach.

# APPENDIX B – LAKE CARRYING CAPACITY STANDARDS METHODOLOGIES – HISTORICAL PERSPECTIVE

In Ontario, **Lakealert**, which was commissioned by the Ministry of Natural Resources (MNR) in 1972, described two approaches to determining the carrying capacity of Precambrian Shield lakes. These were: the shoreline capability calculation; and the theoretical boat density calculation.

The shoreline capability compared the existing number of cottages on a lake with the maximum number that are desirable for development. The maximum desirable number was based on the capability that the shoreline could sustain, and was dependent on factors such as lakeshore slope, drainage, soils and surficial material. Essentially, a shoreline having poor physical conditions would have fewer cottages than would a shoreline having high development capability. The Ontario Lake Inventory provided the basis for the shoreline capability analysis. For example, for high capability shorelines, 50 foot (15 metre) cottage widths were considered acceptable. In contrast, for lower capability shorelines, cottages were spaced further apart, so that on shorelines of the lowest capability rank, the desirable lot width was 500 feet (152 metres).

The boat-density calculation described in **Lakealert** compared the number of boats that theoretically could be active on a lake at a given time, with the number of boats introduced to the lake from existing shoreline development, and public access points. From interviews, questionnaires, and surveys, it was possible to estimate the average number of boats per cottage. By integrating a number of assumptions into a simple mathematical formula, a theoretical boat-density calculation could be made. The assumptions were that:

- there was a net-acreage of surface water that is suitable for recreational boating on a lake;
- the net acreage excluded a shoreline protection zone of a width of 200 feet (61 metres) for the lake in general, 100 feet (30 metres) around emergent aquatic vegetation and navigation hazards, and 400 feet (122 metres) around marinas, and public swimming beaches;
- 10% of the total boats may actually be expected on the water at a given time during peak boating; and
- an average of 10 acres (4 hectares) of water surface was needed per boat for safe boating.

Over the last thirty years, this approach has been improved by conducting actual observations of boating activity and comparing the amount of water surface required by boats and the number available on each lake. The calculations have been refined by varying the above-noted space standard of 10 acres (4 hectares) per boat to a suite of standards depending on boat type and speed. This latter approach has been applied to many lakes and large river systems in Ontario, with varying levels of success/acceptability. While shoreline residents typically recognize that most small to medium inland lakes have a limited capacity for recreational boating, not many are interested in restricting their own activities, even when matters of public safety are evident.

In the early 1970s, Dr. P. Dillon (MOE) and Dr. Frank Rigler (Professor, University of Toronto) published the first scientific approach to determining the carrying capacity of lakes. The protocol, commonly referred to as the Dillon Model dealt with the issue of water quality or lake trophic state in a series of equations that quantified how phosphorus concentrations in a lake can be estimated or predicted, by integrating atmospheric deposition, hydrology and land use (including shoreline development), as these factors influence not only water quality on the subject lake, but all upstream lakes that drain into it. Once the supply of phosphorus was estimated, concentrations of phosphorus, chlorophyll and Secchi disc transparencies can be predicted. These were then compared against measured values, if available, to determine the predictive power of the model; if necessary, the model was calibrated through adjustments and corrections to improve its effectiveness.

By setting a water quality objective, for either phosphorus or chlorophyll, the capacity of a lake can subsequently be determined in terms of the supply of additional or man-made phosphorus that could be loaded to a lake to achieve the objective. In turn, this supply can be expressed in numbers of shoreline lots (permanent and/or seasonal) or any other type of development having an annual supply of phosphorus. This approach, which does not distinguish between warmwater and coldwater lake trout lakes was refined in 1986 through a major initiative of the Province called the **Lakeshore Capacity Study**, and again in 2006, is now recognized and used by the MOE, other resource managers, and many municipalities.

The Dillon model has two primary benefits. First, it deals with lakes, in a watershed context. Second, it predicts. While measurements may be representative of current conditions, measurements alone cannot tell us how conditions will change, if for example, the use of existing development changes, new development occurs, or the phosphorus loads from development can be reduced. Only a model can tell us that. Even if there is a significant gap between the predicted and measured trophic state indicators for a particular lake, and some doubt surrounds the absolute value of the predicted indicators, the model will still indicate the relative change in trophic state that would result from a given change in conditions.

The above-mentioned **Lakeshore Capacity Study** which was completed by the Province in 1986 included six components, as follows:

- Microbiology;
- Fisheries;
- Wildlife;
- Trophic Status;
- Land Use; and
- Integration.

Despite this extensive effort, the only component that has been regularly applied by government agencies, applicants of shoreline development, or cottage associations, etc., is the Dillon Model or Trophic Status component.

The reason that the Microbiological component is not applied is that microbiologists concluded that while fecal coliform and streptococci bacteria tended to appear more frequently and in higher numbers in samples collected from single-tiered cottage shoreline than in samples secured from undeveloped waters, the bacterial densities found in developed areas did not produce significant public health effects.

As well, the Fisheries component, which was based on a supply/demand analysis, was never pursued at a practical or operational level. This is because the negative impacts associated with increased angling pressure (resulting from higher levels of development) can be dealt with by implementing more stringent control on the recreational angler. Of some interest is that when applied in the manner intended, this fisheries component is often the most limiting of all components. Since publication of the Fisheries Component, the Province has substantially firmed up its position with respect to coldwater lake trout lakes; as a result, this aspect of fish habitat is often the prevailing lake capacity limit (see **Section XXXX**).

The Wildlife component examined the affects of shoreline habitat removal on several different animal communities, which are used as indicators of the wildlife population. The primary achievement was the development of quantitative methods for relating wildlife habitat losses associated with cottage developments to actual impact on the wildlife population. This component has never been applied to a single lake on Ontario; rather, its results have been packaged as a series of guidelines that can be implemented as part of a biophysical shoreline capability evaluation.

The Lakeshore Capacity Study did not specifically set out approaches to determine carrying capacity based on shoreline/terrain capability or recreational boating. Both approaches are thoroughly described in our 1993 report.

One of the main reasons that municipalities have generally shied away from undertaking and showing lake specific capacities in their planning documents is the cost implications of doing so, although the District Municipality of Muskoka has been a notable exception. Instead, as a first cut, capacity limits are identified based on standard surface area per dwelling requirements, sometimes with a proviso that for lakes that are approaching their capacity, additional work, which typically relates to water quality, is recommended. Other municipalities have simply reproduced lists provided by the Province of at-capacity lakes.

The current state of the art with respect to lake capacity is that undertaken by the District Municipality of Muskoka.

# APPENDIX C – SHORELINE RESIDENTS' SURVEY – CHANDOS LAKE

# SHORELINE RESIDENTS SURVEY – CHANDOS LAKE

1. **Do you own a property with a built private residence <u>and</u> shoreline on Chandos Lake?** (check one)

   a. Yes, we own Chandos Lake shoreline residence _____

   b. No, we do not own Chandos Lake shoreline residence _____

   **NOTE:** If your property consists of more than one adjacent lot under the same ownership, please answer for the entire group of adjacent lots.

   **If you answered "a" to Question 1, please proceed to Question 3.**

2. **If you answered "b" to Question 1, which of the following applies to you?**

   a. We own vacant property with shoreline on Chandos Lake _____

   b. We own commercial property with shoreline on Chandos Lake _____

   c. We do not own any property with shoreline on Chandos Lake _____

   **If you answered "b" or "c" to Question 2, please stop here and return the survey.**

3. **If you answered "a" to Question 2, do you expect to build a private residence on this property, or to sell/leave it to someone who will, in the next 10 years?**

   Probably ___________ Probably not ___________ Don't know ______________

   **If you answered "a" to Question 2, please proceed to Question 14.**

4. **Is this property your family's principal residence?** (check one)

   a. Yes, principal residence _____

   b. No, not principal residence _____

   **If you answered "a" to Question 4, please proceed to Question 7.**

5. **If you answered "b" to Question 4, to what standard is your residence built?** (Please ignore access issues like snowploughing etc., zoning restrictions, etc.)

   a. Built to year-round occupancy standards and could be lived in as a year-round residence ______

   b. Built to seasonal occupancy standards only ______

**6. If you answered "b" to Question 4, do you expect to do either of the following to your residence over the next 10 years?**

a. Improve/rebuild to year-round occupancy standards so that it could be lived in as a year-round residence, or sell/leave it to someone who will

Probably __________ Probably not __________ Don't know _____________

b. Keep it to seasonal occupancy standards, but improve/rebuild so that it will accommodate significantly more people, or sell/leave it to someone who will

Probably __________ Probably not __________ Don't know _____________

**7. In what year did you or your family acquire the property?** _____

**8. What kind of a sewage treatment system do you have?**

- Septic tank tile field __________
  - What is the size of the tank? ____________ gallons
  - What year was your tile field renewed/upgraded? _________
  - If it hasn't been renewed/upgraded, in what year was it built? ________
  - What is the frequency of emptying/maintaining the septic tank? Every _____ years (approx).
  - How far from the shoreline of the lake is the tile field? __________ metres.
- Holding tank _________
- Pit privy __________

**9. Could you estimate about how many person-nights you, your family, your guests, and any tenants will spend at the property in 2004?**

**NOTE**: A person-night is one night spent by one person. _____ person nights

**EXAMPLE for principal residents**:

| | |
|---|---|
| • Your family of two lives here year round.<br>2 persons x 365 days | 730 |
| • The two of you took one major holiday, going south for four weeks in February and March 2004.<br>2 persons x 28 days | – 56 |
| • In June 2004, three relatives came to stay for two weeks.<br>3 persons x 14 days | + 42 |
| • Total person nights | 716 |

**EXAMPLE for seasonal residents:**

| | | | |
|---|---|---|---|
| • Your family of four spent four long weekends and six regular weekends here in the summer of 2004. | | | |
| 4 persons x 4 weekends x 3 nights | = | 48 | |
| 4 persons x 6 weekends x 2 nights | = | 48 | 96 |
| • Two members of your family spent two weeks (Friday to the following Sunday) here in addition to the weekends. | | | |
| 2 persons x 2 weeks x 9 nights | | | 36 |
| • You had six guests, each for one long weekend. | | | |
| 6 persons x 3 nights | | | 18 |
| • You rented to a group of three for a week (Saturday to Saturday) | | | |
| 3 persons x 7 nights | | | 21 |
| • Your family spent four nights here in February 2004. | | | |
| 4 persons x 4 nights | | | 16 |
| • Total person nights | | | 187 |

**10. Can you tell us something about the number and types of boats on your property?**

| Boat type | Number |
|---|---|
| skiff (outboard steered by throttle grip) | |
| runabout (outboard or inboard steered by wheel) | |
| jet ski boat/personal watercraft | |
| other motorboat | |
| windsurfer | |
| sailboat | |
| canoe/rowboat/paddleboat | |

**11. Do you have an automatic dishwasher?**

Yes _____ No _____

12. **Over the last 10 years, have you noticed any changes in the <u>nearshore</u> environment of Chandos Lake?** (Check one for each of a, b, c.) **If you did not begin to use the property until 1993 or later, please do not answer this question; stop here and return the survey.**

| | | More | No change | Less |
|---|---|---|---|---|
| a. | Water clarity | _____ | _____ | _____ |
| b. | Floating and bottom algae | _____ | _____ | _____ |
| c. | Aquatic plants, including weeds | _____ | _____ | _____ |

13. **Over the last 10 years, have you noticed any changes in the <u>deepwater</u> environment of Chandos Lake?** (Check one for each of a, b, c.) **If you did not begin to use the property until 1994 or later, please do not answer this question; stop here and return the survey.**

**Note:** The **deepwater environment** includes those parts of the lake more than 3 m/10 feet deep. For (a), water clarity, more = more clear; less = less clear.

| | | More | No change | Less |
|---|---|---|---|---|
| a. | Water clarity | _____ | _____ | _____ |
| b. | Floating and bottom algae | _____ | _____ | _____ |
| c. | Aquatic plants, including weeds | _____ | _____ | _____ |

14. **Various planning policies have been advocated or adopted elsewhere in cottage country. Please tell us how you feel about adopting these as part of North Kawartha's official plan and zoning bylaw, keeping in mind that (if relevant) they would apply to your own property should you require any development approval in future.**

| **Policy** | **For** | **Against** | **Undec-ided** |
|---|---|---|---|
| Require that no development take place unless it can demonstrate that its phosphorus (recreational water quality) impact on the lake will be the same or less than the existing impact from the property | | | |
| Require that no development approval or transfer of title take place without septic system reinspection and, if required, upgrading to current standards | | | |
| Increase minimum lot frontage for all new lots created | | | |
| Require site evaluation reports as part of development/redevelopment approval on sites that have development constraints, such as steep slopes, bare rock, wet conditions, or fish or wildlife habitat, across much of the property – larger lot areas and greater frontages and shoreline setbacks would likely be required as a result | | | |

| **Policy** | **For** | **Against** | **Undec-ided** |
|---|---|---|---|
| Encourage alternative residential development styles such as cluster development, where it can be shown that they will have less impacts per unit and per metre of frontage than traditional shoreline development | | | |
| Permit beds & breakfasts and other home businesses where they can demonstrate minimum impact on neighbours | | | |
| Control the amount of the lot that can be covered by development (lot coverage), based not on the entire lot area as in most zoning bylaws, but on the area nearer to shore (for example, maximum x% coverage of that part of the lot within 60 m/200 ft of shore) | | | |
| Control extent of outside decks, stairways, etc. by including them in lot coverage calculations | | | |
| Set a limit on the maximum size of the main dwelling (for example, a maximum gross floor area) | | | |
| Require that all buildings (except boathouses, pumphouses, etc.) and tile fields be set back at least 30 m/100 ft from shore | | | |
| Prohibit clearing of vegetation and other site alteration, and require planting in cleared areas, in much of the required shoreline setback (one example is no disturbance in 75% of the area up to 8 m/25 ft from shore) | | | |
| Require that docks etc. can only be developed and shorelines "hardened" along a portion of shoreline frontage (one example is up to 23 m/75 ft or 25% of frontage, whichever is less) | | | |
| Restrict nearshore lighting so that its light pollution impacts are minimized | | | |
| Prohibit development in valued natural areas beyond what is required by provincial policy (for example, smaller, less significant wetlands, and key wildlife habitats), and require development near those natural areas show no adverse impacts | | | |

**15. If you have any other comments about the environmental and recreational health of Chandos Lake, or about environmental or land use planning policies affecting the lake, please provide them here or attach additional pages.**

# APPENDIX D – SHORELINE DEVELOPMENT DATA COLLECTION AND ANALYSIS

# SHORELINE DEVELOPMENT DATA COLLECTION AND ANALYSIS

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September 17, 2004
revised May 8, 2009

***Why collect shoreline development data?***

The purpose of collecting and analyzing the data is to provide reasonably reliable counts of how many developed residential lots and developable vacant lots there are on or near a lake. This information is essential for trophic state (recreational water quality) capacity modelling, and is also often required for other planning purposes.

Unless the municipality has already done this work, there is usually no reliable way to obtain this information, except by going through the assessment roll. Because of the rate of development in cottage country, even if the municipality has done this work, its numbers should not be relied on unless the work was done within the last couple of years, or staff have kept the numbers up to date.

***What is a developed residential lot?***

A developed residential lot:
- can be conveyed (sold) separately from any other lot,
- fronts on, or is within a set distance from, the water body,
- has on it one or more seasonal or permanent dwellings,
- is not commercial in use, unless the size of the lot is "typically residential" for the area and the commercial use is inside or subordinate to the dwelling, that is, the lot could easily change to residential use.

***What is a developable vacant lot?***

A developable vacant lot:
- can be conveyed (sold) separately from any other lot,
- fronts on, or is within a set distance from, the water body,
- is vacant,
- is not owned by the municipality or other public authority, unless you know that the public authority plans to sell the lot for development,
- may have less area or frontage than is currently permitted by the local zoning bylaw, but is not so small that the lot is for all practical purposes impossible to build on and/or would be extremely unlikely to be granted a rezoning or minor variance to permit development.

***What is a conveyable lot?***

A property shown on the assessment roll cannot necessarily be sold separately from any other lot. Or, it may include two or more lots that can be sold separately. The rules for determining this are complex, but the simplified version in "How do I interpret the data to count conveyable lots?", below, will account for most situations.

***What do I need to record the data?***

You need:
- access to the assessment roll and maps for the municipality, normally obtainable during regular hours at the municipal office,
- plenty of paper (or electronic equivalent) for recording the data,
- a ruler.

***What is in the assessment roll and maps?***

Each separately taxed property is individually identified in the assessment roll prepared by the Municipal Property Assessment Corporation (MPAC) for each municipality. Usually each property is also individually mapped on the accompanying assessment maps. However, while MPAC updates the rolls every year, it does not update the maps. Sometimes municipal staff update the maps by hand; in other cases, you may have to do some guesswork as to exactly where newly created properties are located.

Each property has a unique identifier number that is shown on the assessment roll and map. This 15 digit number has the following components.
- digits 1-2: county/region code
- digits 3-4: local municipality code
- digits 5-7: map division within municipality
- digits 8-10: subdivision within map division
- digits 11-15: property number, unique to each property within subdivision.

Therefore, the key component of the identifier number is the property number (digits 11-15). In a rural municipality, a subdivision (digits 8-10) covers a large area, and within that area, each property has its own number.

The assessment map will show the property number (digits 11-15) for each property, plus whatever preceding digits are necessary to clearly identify the property. The roll, but not the map, will show an additional component (digits 16-19), called a subordinate, for each entry. Most often, there is only one entry per property, and the subordinate is "0000". Some properties have more than one entry, each with its own subordinate. However, subordinates do not represent legal divisions of land, and are of no significance to this exercise.

Often, there will be several maps covering a water body. Sometimes there will be a base map that shows the information for large properties, and serves as a key map referring you to other, more detailed maps for concentrations of smaller properties. Note that there may be many maps within one map division code (digits 5-7).

***How do I proceed?***

Using the map, you should pick a starting point on the shoreline where a new group of roll numbers begins (typically, where a map begins if there are several maps, or where properties are divided by a road allowance). You should then proceed by roll number through the assessment roll, looking each roll number up on the map and recording the information listed below.

As the same property number (digits 11-15) can recur within various map divisions and subdivisions within a municipality, it is important to regularly check location information, such as lot and concession, in the roll against that on the assessment maps, to ensure that you are in the right place and recording the right data.

When you get to the end of each map, check to make sure that every property on or near the shoreline (see "What properties should be included?", below) that is shown on the map has been covered in the roll information you have recorded. If not, go back to the roll to fill in the gaps. (Often there are oddities in how roll numbers are assigned, and some properties on the same map may be in quite different places in the roll.) Make sure you have covered islands, ends of peninsulas the bases of which are on other maps, etc., as these are especially likely to have out-of-sequence roll numbers.

***What properties should be included?***

You should record data for all properties on or near the shoreline of the water body.

A property on the shoreline:
- fronts on the water body, or
- is separated from the water body by a shoreline road allowance only (normally a 66 ft shoreline strip).

A property near the shoreline is not on the shoreline, but is partly or entirely within 300 m of the shoreline. However, you should not record data for properties that:
- based on your knowledge of local topography and drainage, drain into another water body,
- based on their shape within and beyond 300 m of shore and their existing or likely dwelling locations, likely have or will have their tile fields located beyond 300 m from shore.

In determining whether and how much of a property is within 300 m of shore, take care to note the scale on the map. Sometimes the maps surrounding one lake may have two or three different scales.

If you are compiling data on more than one water body (e.g., for a watershed), a property that is on or near more than one water body should be counted only once, on the water body farthest upstream.

***What information should be recorded?***

For each property on or near the shoreline, you should record the following.

1. Assessment map number (each map has its own number, shown on the map; the last three digits are unique to the map and are all that needs to be recorded).

2. 15 digit roll number.

3. Whether the property is on or near the shoreline. See "What properties should be included?", above.

4. Name(s) of owner(s). The owner is coded as O in the "OTV" column of the assessment roll; the roll also shows any tenants, coded T, but these should be ignored.

5. Unit class code. This is a two or three letter code for type of use (e.g., RU - residential unit, RDU - recreational dwelling unit, VL - vacant lands).

6. Whether or not the property is in a registered plan of subdivision. There are two kinds of registered plans (that is, plans registered in the local registry office as legal descriptions of land): reference plans and plans of subdivision. If a property is part of either type of registered plan, the plan and lot number will usually be included in the legal description provided in the roll for each property. Plan and lot numbers may also be shown on the maps.

   To distinguish between a reference plan and a plan of subdivision, look at the plan number. A reference plan is usually referred to as Plan xx R yyyy, where xx is a number or letter county code and yyyy is an individual plan number. Its components are usually referred to as "parts". A plan of subdivision is usually referred to as Plan yyyy only (the first y may be a letter or number, and the rest are numbers). Its components are usually referred to as "lots" and "blocks". Again, all that matters here is whether or not the property is part of a plan of subdivision.

7. If the property is not vacant, whether or not it is the owner's principal residence. This should be based on the owner's mailing address (provided in the roll), that is, where her tax bill is sent (this method is not 100% accurate but provides a reasonable estimate). You will need to determine which mailing addresses would be used as local addresses by permanent residents.

8. Occasionally, what the assessment map shows as two or more separate but abutting map units share the same property number on the roll (they will, of course, have the same owner). You should note where this is the case, and you will need to determine whether this property consists of two or more lots (see "How do I interpret the data to count conveyable lots", below).

***What other information might I need?***

Part of the shorelands of your lake may be served by a municipal or communal sewage system. A communal system serves six or more residential lots or units but is not municipally owned, e.g., in a condominium development. This information is not in assessment roll or maps. If you believe there are municipal or communal systems on your lake, find out from the municipality what areas they serve. Then, record this information for the affected properties.

***How do I interpret the data to count conveyable lots?***

This part of the exercise is easiest if you can photocopy the relevant portions of the assessment maps, although if the property fabric is very simple (e.g., a single tier of standard-shaped lots running around a small lake) there is less advantage in doing so. However, MPAC is attempting to enforce its ownership rights of the assessment maps and is asking municipalities not to allow copying. Municipal staff may refuse your request for that reason.

If you do not or cannot photocopy the maps, you will have to make notes on each entry, and/or draw lines linking entries, wherever the following rules indicate that an assessment roll property is not identical to a conveyable lot (either because it merges with one or more other properties to form one conveyable lot, or consists of two or more conveyable lots).

The following rules will help you count conveyable lots.

1. Two properties have the same owner if their owners are the exact same person(s) or corporation(s). (However, if property A is owned by John and Mary Smith, and property B is owned by John Smith, or by John, Mary, and Jane Smith, those two properties do not have the same owner.)

2. Each property that does not border any other property with the same owner is one conveyable lot.

3. Each property that is within a plan of subdivision is one conveyable lot, even if it borders another property that is still within the subdivision and has the same owner. (However, one property may consist of two or more abutting lots in a plan of subdivision. This does not mean that that property consists of two or more conveyable lots. It usually means that the subdivision lots have been legally been merged in title, so any such property should be counted as one conveyable lot only.) (Also, each lot created by consent (severance) is one conveyable lot, even if it borders another property that has the same owner. However, it is not generally possible to identify lots created by consent from the assessment roll, so this methodology does not take consents into account.)

4. Each group of abutting properties with the same owner but not in a plan of subdivision is one conveyable lot.

5. Despite rules 1 to 4, any single property, any group of abutting properties with the same owner, or any group of abutting assessment map units with the same roll number, is two (or more) conveyable lots if:
   - divided by a public road allowance,
   - divided into island(s) and mainland, or otherwise divided by a lake or river (a minor stream that doesn't, for example, connect one lake to another doesn't

count).

***How do I interpret the data to determine residential, vacant, etc. properties?***

This is done using the unit class code information, as follows.

| *Unit class code* | *Type of use* |
|---|---|
| RU, FRU, RDU | Developed residential |
| VL, FL, CL, MF, MFC | Vacant |
| COM | Commercial |
| Any other code | Other |

Properties with multiple assessment roll entries and subordinates may have more than one unit class code. The property should be classified as developed residential if any of its codes are RU, FRU, or RDU (for example, a farm with a dwelling will likely have one FRU and one FL entry). The property should be classified as vacant if it has no codes other than VL, FL, CL, MF, or MFC. Otherwise, the property should be classified as commercial or other.

Commercial properties are important and should be individually highlighted. They may be resorts, campgrounds, camps, etc. that are significant lake users, and will be separately incorporated into any lake capacity modelling.

***How do I count lots?***

Now, it should finally be possible to count the lots, and classify them into developed residential lots, vacant lots, and commercial and other lots, making a few more adjustments along the way.

When a lot consists of two or more properties, determine the use of the lot as follows:

| *Lot consists of the following properties* | *Use of the lot* |
|---|---|
| 2 or more developed residential | developed residential |
| 2 or more vacant | vacant |
| 1 or more vacant + 1 or more other use (developed residential, or commercial or other) | developed residential, or commercial or other |

When a property is divided into more than one lot, each lot has the use of the parent property. But if, for example, the parent property is developed residential and you believe from the assessment information or your own knowledge that as a result of dividing it, one lot would be vacant, then classify the lots accordingly.

You will also need to make the following adjustments:

- If you believe that a lot with a commercial use also has a dwelling, that the lot size is "typically residential" for the area, and that the commercial use is inside or subordinate to the dwelling, then reclassify the lot as developed residential.
- If a vacant lot is owned by the municipality or other public authority, unless you believe that the public authority plans to sell the lot for development, then note that the lot is undevelopable.
- If you believe that the lot is so small that the lot is for all practical purposes impossible to build on and/or would be extremely unlikely to be granted a rezoning or minor variance to permit development, then reclassify the lot as undevelopable.

Finally, you should do your analysis so that the following can be tabulated for each of the three use types:

- on the shoreline vs. near the shoreline
- not served vs. served by a municipal/communal sewage system
- owner's principal residence vs. not principal residence (developed residential lots only)
- developable vs. undevelopable (vacant lots only).

# APPENDIX E – MEAN VOLUME WEIGHTED HYPOLIMNETIC DISSOLVED OXYGEN

FAU UPDATE

Ontario Ministry of Natural Resources

# DETERMINATION OF VOLUME-WEIGHTED OXYGEN CONCENTRATION

There are several methods used to quantify cold-water fish species habitat based on oxygen concentrations. For lake trout, optimal habitat has been described as having greater than 6 mg $L^{-1}$ oxygen at less than 10°C (OMOE and OMNR 1986). Usable habitat has expanded boundaries at greater than 4 mg $L^{-1}$ and less than 15°C (OMOE and OMNR 1986). These guidelines can be used to generate habitat 'volumes' which may be difficult to interpret since they also show a great deal of variation. Similar 'volumes' may take on different meanings when expressed as percents of total volume. It is also difficult to assess the importance of small amounts of habitat loss that occur in the bottom few metres of many lakes.

The proposed use of volume-weighted hypolimnetic oxygen concentrations (D. Evans, Ontario Ministry of Natural Resources (OMNR), personal communication) to define lake trout habitat would eliminate many of these problems. Oxygen profiles from the first week in September are used in conjunction with individual stratum volumes to derive single volume-weighted oxygen 'numbers' for each lake. Volume-weighted oxygen is calculated as the measured dissolved oxygen at each stratum times the proportion of the hypolimnetic volume represented by that stratum. These values are summed for each stratum in the hypolimnion. It is suggested for lake trout that these values remain above 7 mg $L^{-1}$. Lakes with large volumes of oxygenated water would not have their average greatly affected by small volumes of depleted water near the bottom. Lakes with small and enriched hypolimnia that have generally lower oxygen concentrations would be affected to a greater degree by oxygen-depleted bottom waters. End-of-summer volume-weighted oxygen allows the use of a single number to compare conditions between lakes which may otherwise show seasonal and spatial variability.

Calculating volume-weighted hypolimnetic oxygen requires morphometry data and at least one end-of-summer oxygen and temperature profile (Aug 15 - Sept 15 ). Ideally, several oxygen and temperature profiles would be used to reflect long-term average conditions. Temperature profiles are used only to establish the upper limit of the hypolimnion which is defined as the lower depth of the first 1 m interval (in the hypolimnion) where temperature change is less than 1°C per m. Once these boundaries are established, the volumes for each stratum in the hypolimnion are determined. The depth of each stratum will depend on the intervals observed on the contour map and in the oxygen profiles themselves. OMNR staff record and calculate results based on 1 m strata and Ministry of the Environment and Energy (MOEE) staff currently use 2 m strata.

Area and depth information from morphometric maps should first be converted to ha and m if originally in acres and feet. This will yield contour areas in ha for uneven numbers of m but these can be converted to 1 or 2 m contour areas by one of two methods. 1. M and ha are graphed and the individual areas for each stratum are simply read from the axis of the graph. 2. Individual pairs of adjacent points in ha and m are used to interpolate areas for the intervals that fall within the depth range spanned by the pair of points. This can be done through simple linear interpolation or by doing a linear regression on two pairs of points. However, it is important to note that entire sets of hypolimnetic depth/area data cannot be regressed as a single group of numbers because the relationship is almost always curvilinear.

Individual contour areas are then converted to volumes by the formula:

$$V = \frac{m}{3}\left(A_t + A_b + \sqrt{A_t * A_b}\right)$$

where

V is volume in $m^3 \times 10^4$
$A_t$ is the area in ha of the top of the stratum
$A_b$ is the area in ha of the bottom of the stratum
m is the depth of the stratum in m

Volume data for each stratum of the hypolimnion are then expressed as a fraction of the total hypolimnetic volume and these are multiplied by the oxygen concentration observed for each stratum. These individual concentrations are summed to yield volume-weighted average oxygen as shown in Table I. Typical uppermost stratum depths will be 10-12, 12-14, or 14-16 m, depending generally on fetch.

In lake trout lakes, volume-weighted oxygen concentrations above 7 mg $L^{-1}$ are desirable and this number seems to be reasonable when compared to optimal and usable habitat calculations. Obviously, hypolimnia with 6.1 mg $L^{-1}$ oxygen from top to bottom would represent habitat that was above 'optimal', yet it would be below the guidelines for volume-weighted oxygen since the volume-weighted number would be 6.1 mg $L^{-1}$ as well. This type of discrepancy will be rare considering that concentrations are usually not homogeneous in lakes that show some oxygen depletion. Most lakes that have suffered habitat loss or have natural marginal habitats for lake trout will show volume-weighted hypolimnetic oxygen concentrations below 7 mg $L^{-1}$.

| Stratum | Volume | A-Volume as fraction of total volume | B-dissolved oxygen (mg $L^{-1}$) | A*B |
|---|---|---|---|---|
| 14-16m | 1500 | 0.49 | 10.0 | 4.9 |
| 16-18m | 1000 | 0.33 | 8.0 | 2.6 |
| 18-20m | 400 | 0.13 | 6.0 | 0.78 |
| 20-22m | 150 | 0.05 | 1.0 | 0.05 |
| Total | 3050 | | | 8.33 |

Table I: Example calculation of volume-weighted average oxygen

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ONTARIO MINISTRY OF THE ENVIRONMENT (OMOE) AND ONTARIO MINISTRY OF NATURAL RESOURCES (OMNR). 1986. Inland lake trout management in Eastern Ontario. Ont. Min. Environ. and Ont. Min. Nat. Resour. Ms rep. 56 p.

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## APPENDIX F – LIMNOLOGY, PLUMBING AND PLANNING: EVALUATION OF NUTRIENT – BASED LIMITS TO SHORELINE DEVELOPMENT IN PRECAMBRIAN SHIELD WATERSHEDS

# HANDBOOK OF WATER SENSITIVE PLANNING and DESIGN

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# chapter II.17

# Limnology, plumbing and planning: Evaluation of nutrient-based limits to shoreline development in Precambrian Shield watersheds

*Neil J. Hutchinson*

## Abstract

The concept of using water quality as a planning tool for recreational lakes has been in active practice in Ontario and parts of the United States for approximately 25 years. In practice, assumptions regarding anthropogenic loadings of phosphorus to a watershed (generally septic systems servicing shoreline development) are linked to estimates of natural phosphorus loading. The resultant model estimates total phosphorus concentration and the response of trophic status indicators such as water clarity and dissolved oxygen in specific lakes. Linking the model to a water quality objective allows planners to set capacities for anthropogenic phosphorus loads, and hence shoreline development such as cottages, resorts, or permanent homes. This chapter presents an example of how the concept can be applied in practice, based on the application of the author's experience to a test watershed in south-central Ontario. Practical examples are given to show the development and calibration of accurate trophic status models, the use of monitoring data to set ecologically valid water quality objectives and their translation into shoreline development capacities, and to show the strengths and weaknesses of the approach.

The availability of a scientifically based water quality model has overemphasized water quality as a planning tool and generated unrealistic expectations of a single-capacity determinant among the public. Recent advances in our understanding of the geochemistry of domestic septic systems indicates that less phosphorus may be mobile than was previously assumed. In addition, as alternative septic technologies for phosphorus abatement are developed a refocusing of capacity determinants will be required. A combination of land-use regulations and a scientifically based management program is recommended as an alternative to a single, phosphorus-focused approach. These could address stresses to the ecology of the riparian and littoral zones and acknowledge the importance of social determinants such as noise, crowding, powerboats, and the wilderness aesthetic. This would promote a diversity of planning approaches, shift the existing focus away from plumbing and septic systems, and provide a more holistic management program which protected more components of the lake system.

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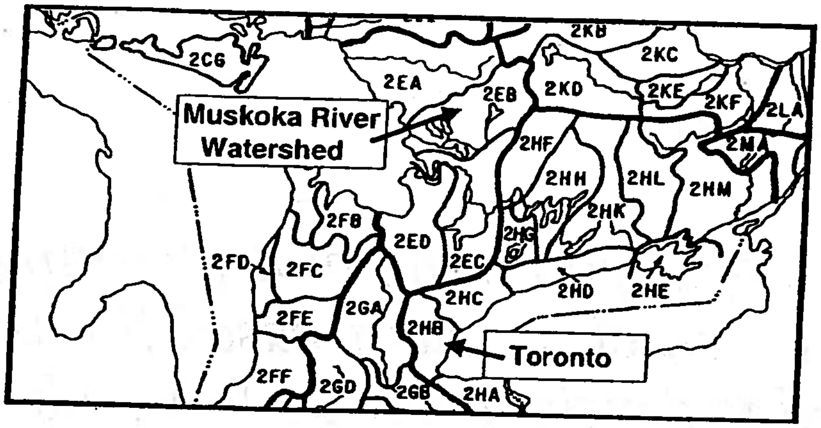


*Figure II.17.1* Southern Ontario tertiary watershed divisions showing Muskoka River watershed draining west to Georgian Bay.

## *Introduction*

Lake-based recreation and tourism is a fundamental cornerstone of the economy for many small rural communities situated on the Precambrian Shield in Ontario. The economic stimulus provided by tourism has diversified the economy of these communities, which were historically dependent on declining revenues from forestry or mining. Tourism on the southern portions of the Shield draws from the large and prosperous demographic of southern Ontario, in particular the "Golden Horseshoe" of 7 million people, which extends from east of Toronto to Niagara Falls, but also draws from the Great Lakes states of Michigan, Ohio, and New York.

The Precambrian (or "Canadian") Shield is made up of Precambrian gneiss and granite bedrock that lies near or on the surface throughout most of the northern areas of the province. Landscapes on the southern Shield are dominated by bedrock ridges and knolls, thin soil cover, and pine and hardwood forests. Lakes are also a dominant presence on the landscape. The Province of Ontario has over 260,000 lakes greater than 1 ha in size (Cox, 1978) and most of these are situated on the Shield. Thin soils and acidic bedrock mean that most of these lakes are nutrient-poor and possess high water clarity. Clear waters, rocky shorelines, and forest cover produce exceptional recreational attributes and, as a result, southern portions of the Shield have served as a focus for shoreline recreation in the form of cottages and resorts.

The Muskoka River Watershed was chosen as a test case to illustrate the development, validation, and application of a trophic status model to govern recreational shoreline development. The watershed covers approximately 5737 km$^2$ and is located on the southernmost fringes of the Precambrian Shield, approximately 180 km north of Toronto (Figure II.17.1). Four large (>5,000-ha) lakes — Lake Muskoka, Lake Rosseau, Lake Joseph, and Lake of Bays — form the recreational focus of the region. These were opened up to summer recreation with the arrival of railways in the 1880s and the formation of steamship service on larger lakes shortly thereafter. Approximately 150 smaller lakes also support shoreline recreation. Three major towns are located in the watershed, each with a population exceeding 8,000 permanent residents. Eight smaller communities of 500 to 2,000 people also exist. Some of these are serviced with sewage treatment plants discharging to the watershed, while others are serviced by septic systems.

Development on these lakes consists of approximately 18,000 shoreline residences, 4,300 resort units, and 1,400 trailer sites. Approximately 4,800 ha of urbanized areas are located adjacent to shorelines and 2,700 kg/year of phosphorus are added from point source STPs in urban centers. The Lake Rosseau watershed also includes approximately 4,800 ha of agricultural land use. There are approximately 4,400 vacant shoreline lots across the watershed, which represents a substantial resource base of future development potential. Approximately 1,400 back lots (i.e., set back from the shoreline) exist, and about one third of these are vacant.

The Province of Ontario and various municipal governments in recreational areas have maintained water quality programs since the late 1970s, to manage recreational growth in recognition of the important economic link of tourism to water quality. These programs generally consist of four elements:

1. Policies to maintain water quality through limits to shoreline development
2. Predictive models linking shoreline development to water quality
3. Lake-specific policies, including development objectives based on water quality and,
4. Monitoring programs to track changes in water quality in lakes

This chapter describes a process for developing a water quality model, validating its predictions of water quality, and setting development objectives on the basis of water quality. Issues of water sensitive land-use planning with respect to rural lakes are also covered in Chapters II.1, II.12, II.13, II.14, and II.16.

The author has made use of water quality and land-use data for a set of lakes situated within the Muskoka River watershed. The concepts and observations herein are those of the author alone and are intended to guide technical practitioners of water quality planning in recreational lakes in Ontario and elsewhere. They are not presented as specific recommendations for water quality planning for lakes in the Muskoka River Watershed — but the Muskoka watershed is used as an example of how these models can be applied throughout the Precambrian Shield in Ontario and elsewhere.

## *The history and origins of lakeshore capacity planning in Ontario*

The first lakeshore capacity planning initiatives in Ontario grew out of the efforts to control eutrophication of Lake Erie in the 1970s. Lake Erie was a large and visible example of the threats posed by enrichment of surface waters with the algal nutrient phosphorus and of the success of remedial programs that were centered on managing the lake's phosphorus budget. In the same era, the eutrophication of inland lakes was also documented in response to inputs of partially treated sewage effluent. Among these was Gravenhurst Bay on Lake Muskoka, which suffered a history of algal blooms until tertiary sewage treatment was implemented in 1972 (Michalski et al., 1975) and enhanced treatment and relocation of the outfall were implemented in May 1994.

The primary water quality concern in Ontario's cottage country is also nutrient enrichment. Excessive phosphorus input promotes the growth of algae, causing a loss of water clarity. The lake user sees this as "greener" water of less aesthetic appeal or as surface blooms of nuisance algal growth. Algae settle to the bottom of the lake, where their decomposition consumes oxygen, reducing the amount of cold, oxygen-rich habitat available for sensitive aquatic life such as lake trout (*Salvelinus namaycush)* and triggering remineralization of sediment-bound phosphorus. Residential or cottage development on a shoreline may increase the input of phosphorus to a lake. Domestic septic systems may be a significant component of the loading, but clearing of the shoreline, fertilizer application, and increased erosion are also important.

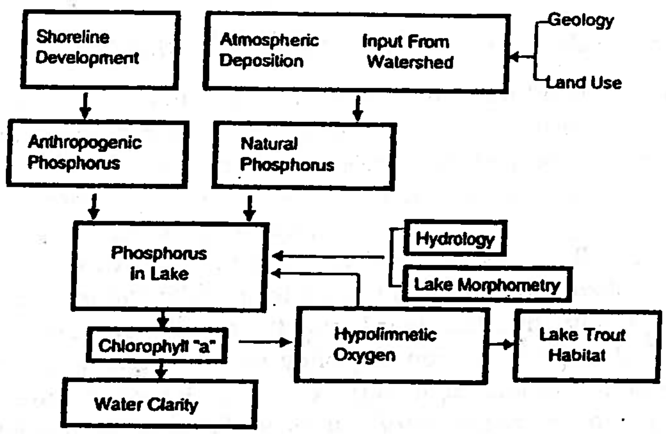


*Figure II.17.2* Schematic of water quality models used in Ontario. Lake trout habitat is not modeled by municipalities, as fish habitat management is a federal government responsibility in Ontario.

(Dillon and Molot, 1997). Finally, the settling velocity for loss of phosphorus to lake sediments used in the models is an average of local estimates made from calibrated watersheds and lakes with oxic and anoxic hypolimnia (Dillon et al., 1986). In-lake phosphorus retention may vary regionally and from lake to lake. The use of these models must always, therefore, be preceded by calibration to specific local conditions.

Three major assumptions underlie the use of the Ontario Trophic Status model:

1. The first is that 100% of the phosphorus loaded to a shoreline septic system will ultimately be expressed as increased trophic status (Dillon et al., 1986, 1994). This assumption has only been tested indirectly as a function of the fit of predicted with measured phosphorus in study lakes (Hutchinson et al., 1991; Dillon et al., 1994). Recent investigations of septic system geochemistry (Robertson et al., 1998) and the mechanisms of phosphorus mineralization in soils (Isenbeck-Schroter et al., 1993; Jenkins et al., 1971) suggest that this assumption is debatable where soils are present between a septic system and a water body and that 100% phosphorus export is, in fact, unlikely.
2. The second assumption is that all anthropogenic phosphorus sources within 300 m of the lakeshore, or any inflowing tributary, must be included in the lake's phosphorus budget (Dillon and Rigler, 1975; Dillon et al., 1986). Although this assumption is based on the need to place boundaries on the inclusion of phosphorus sources, the distance of 300 m is arbitrary and has neither been substantiated nor tested.
3. Finally, the lakes used to calibrate the Ontario model are small, headwater lakes (Dillon et al., 1986). Although straightforward adaptation of the model to an entire watershed is recommended (Dillon and Rigler, 1975; Dillon et al., 1986), published validations have focussed only on headwater lakes (Dillon et al., 1994) or on simple watersheds with a limited number of lakes (Hutchinson et al., 1991). The additional complexity inherent in expanding the original concept and development allocation schemes to large watersheds must be addressed as part of any policy development process.

These assumptions form the basis of challenges to the concept of determining lakeshore capacity through trophic status (Michalski, 1994) and guided much of the technical recasting of trophic status models reported herein.

In summary, trophic status models have been developed that can accurately predict the responses of lakes to natural and anthropogenic phosphorus sources. The models are regionally specific and must be validated against local conditions before they can be used with confidence. In addition, assumptions guiding anthropogenic loading estimates and watershed implementation are open to some debate on both empirical and mechanistic grounds and should be addressed before beginning any management exercise.

## *Model validation*

The accuracy of predictions made by a trophic status model must be confirmed against measurements of water quality in the subject lakes before the models can be used with confidence in policy setting. This requires:

1. Establishment of a water quality monitoring program to determine existing levels of phosphorus in lakes for comparison against present-day model predictions
2. Maintenance of the monitoring program for the long term to determine any trends in water quality
3. Calibration of natural phosphorus loadings and basic model operation in undeveloped lakes, with no human phosphorus sources
4. Calibration of the model on developed lakes to determine if assumptions on anthropogenic phosphorus sources are valid
5. A process to resolve inaccuracies in model predictions and to update the model on the basis of monitoring results

Model validation must also meet the requirements of the planning process it supports. The intent of a lake management program is to achieve stable and predictable water quality. This must occur in concert with the requirements for a stable and consistent planning and policy environment. Water quality programs should set stable targets for a minimum of 20 years — time to resolve new steady states in water quality in a monitoring program and to provide a stable economic and planning environment, without jeopardizing the resource through over-allocation of development.

## *Monitoring programs*

The monitoring program supporting a lakeshore capacity policy must strike a balance between practical implementation, accuracy, and expense. Ontario's water quality models were developed on the basis of an intensive, long-term research program on a small number of study lakes which was undertaken by the provincial Ministry of the Environment in the Muskoka and Haliburton regions of Ontario. The program was based on dedicated laboratory procedures and analytical staff, long-term personnel, and routine scientific review. In contrast, a municipal program may have to be implemented with limited financial support, summer or term staff, a variety of commercial or government laboratory analyses, lack of in-house expertise, and staff who manage water quality only as one aspect of their career. None of these, on its own, jeopardizes the integrity of a water quality program, but all represent the potential for error and the need for stable policy support. The water quality program must, therefore, be supported by a cost-effective monitoring program that can be maintained for the long-term.

The water quality model is based on predictions of total phosphorus, and so the best comparisons of model accuracy are obtained by measuring total phosphorus directly. Published water quality relationships for Muskoka–Haliburton lakes (Clark and Hutchinson, 1992) suggest that long-term trends in water quality can be determined by making one phosphorus measurement each year at the time of spring overturn, when the lake is completely mixed from top to bottom, thus reducing program costs. Accordingly, an effective water quality monitoring program may consist of:

1. An annual spring overturn measurement of total phosphorus for comparison with water quality predictions made by the model
2. Biweekly measurements of Secchi depth during the summer to track long-term changes in water clarity, the recreational attribute that forms the basis of the water quality program
3. An annual measurement of the dissolved oxygen profiles made at the end of summer, when oxygen stress is most likely, to determine the oxygen status of lakes for model input, to establish the suitability of aquatic habitat, and to track long-term changes

The results of the monitoring program can be used to calibrate the water quality model and to identify those lakes for which the model does not produce accurate estimates of trophic status.

## *Model calibration*

The primary requirement for a water quality model, once accuracy has been demonstrated, is that it be based on a solid mechanistic understanding of watershed and lake dynamics. A purely empirical approach, in which understanding and technical substantiation are ignored and the model "fit" is the only rationale for model acceptance, may not be technically defensible. The coefficients and assumptions that make up the predicted water quality must be clear and documented, as they will form the basis of challenges to the model. Clear technical rationale is also required so that all model users can understand the model and improve it in the future. Wherever possible, the water quality model should be substantiated by reference to the primary scientific literature.

The Ontario Lakeshore Capacity Study calibrated its trophic status model (Dillon et al., 1986) on research lakes that had no shoreline development. This was done in order to quantify basic lake processes in the absence of the additional uncertainty inherent in assumptions of human phosphorus loading from shoreline septic systems. While this is mechanistically sound and represents the ideal approach, it may not be practical for municipalities or other jurisdictions. Water quality programs are focussed on those lakes where shoreline development is present, because they are the lakes that require management activities such as development capacities. Uninhabited lakes are not generally monitored as part of a water quality program, posing difficulties in calibrating basic model elements such as settling velocity, natural watershed loads, or in-lake retention. These factors must be well quantified before considerations of the uncertainties in quantifying human phosphorus loadings.

Where the number of uninhabited lakes in the calibration set is not considered adequate for model development, then the calibration exercise should include lakes in which only limited shoreline development is present. A cutoff where less than 10% of the total potential load is added from shoreline development may suffice in these cases. Measurements of water quality may be available for larger numbers of these "sparsely inhabited" lakes to assist with model calibration.

*Calibrating natural phosphorus sources.* Consideration of two factors is recommended to improve model fit for natural phosphorus loading.

*Incorporation of wetlands to describe natural phosphorus export.* Recent research on south-central Ontario watersheds showed that phosphorus export from wetlands (plus atmospheric deposition) determined natural phosphorus loading to a lake. This was stated as the following estimate of phosphorus load (from Dillon and Molot, 1997):

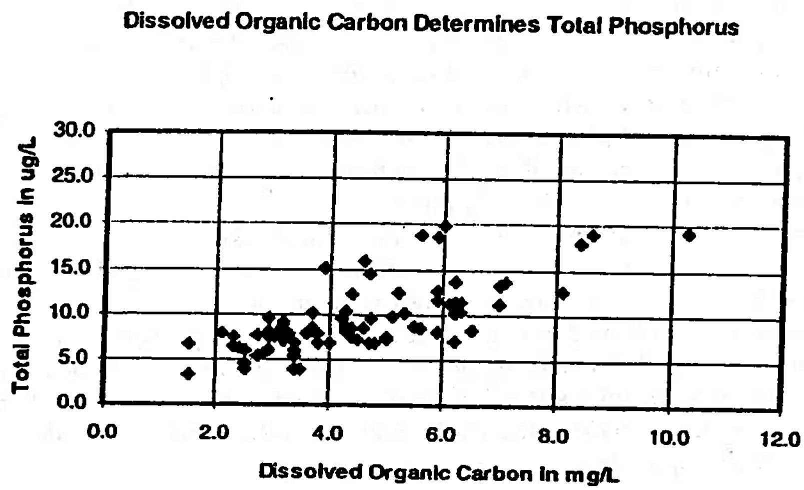


*Figure II.17.3* Influence of dissolved organic carbon (DOC) on average long-term total phosphorus concentrations in Precambrian Shield lakes.

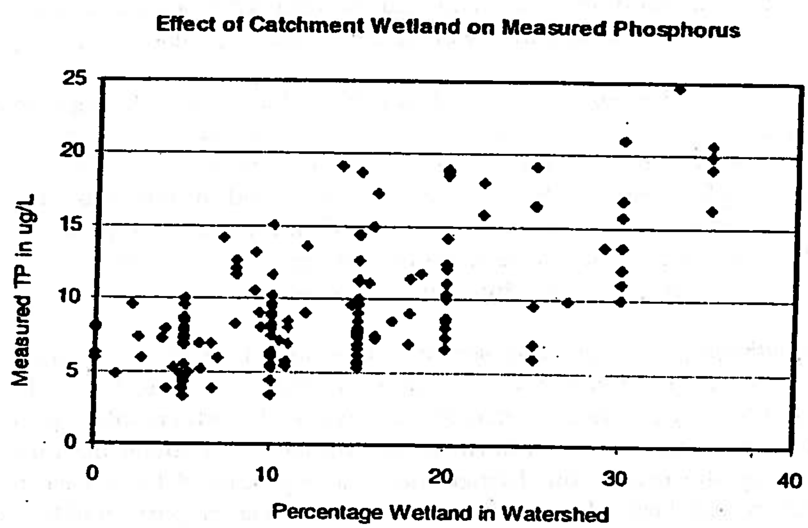


*Figure II.17.4* Relationship of average long-term measured phosphorus in Precambrian Shield lakes with wetland area in catchment.

$$\text{kg TP/year} = \text{catchment area (km}^2) * (3.05 + (0.54 * \%\ \text{wetland}))$$

This relationship is driven by the export of phosphorus with dissolved organic carbon from wetlands in the catchments of the lakes. This is shown for lakes of the Muskoka River watershed in Figure II.17.3. Total phosphorus concentrations were significantly related ($p < 0.000001$, $r^2 = 0.39$) to the amount of wetland in the catchments of lakes in the Muskoka River watershed (Figure II.17.4.). Natural phosphorus loading from all catchments containing wetland can therefore be estimated from wetland area.

*Phosphorus retention in shallow lakes.* Both the Dillon–Rigler and Lakeshore Capacity Study models were developed and calibrated for use in lakes that are deep enough to stratify. The models are not intended for use in shallow lakes, in which phosphorus does not settle to hypolimnetic waters. The major requirement for modeling a shallow lake is to devise an accurate estimate of in-lake retention of phosphorus. The models estimate retention as a function of the areal water load ($m^3$ of runoff/year per $m^2$ of lake surface) and the settling velocity of phosphorus, as modified by the hypolimnetic oxygen status (Dillon et al., 1986). For shallow lakes, this method overestimates retention as it does not allow for wind-driven resuspension of phosphorus.

Several methods of estimating retention in shallow lakes can be attempted, most of which involve modification of the apparent settling velocity. In the end, shallow lakes are best calibrated individually. In-lake phosphorus retention can be modified to achieve a best fit between measured and predicted estimates of total phosphorus concentration. Calibration of the many shallow lakes present in the watershed, most of which have no shoreline development, provides the final improvement to the model and, together with the other modifications, provides the confidence in model accuracy to allow its use for setting development capacities.

*Atmospheric phosphorus loading.* Trophic status models must include a phosphorus contribution from the atmosphere directly to the lake surface. Atmospheric loading includes contributions from precipitation and from dry loading (or dust). Values such as 20.7 mg/$m^2$/year for the Precambrian Shield in south-central Ontario can be obtained from published monitoring studies, such as those of the Ontario Ministry of the Environment (Dillon et al., 1992).

*Hydrologic loading.* Trophic status models require input of hydrologic loading for each lake. The models are long-term steady-state models and so are cast to use average annual depth of runoff. These values can be obtained from published figures (Canada Dept. of Fisheries and Environ., 1978). They can then be coded into the model in a "look-up" table and determined for each lake on the basis of latitude and longitude of the lake, as entered by the user. They should be reviewed periodically, however, in light of the potential changes in runoff stimulated by climate change.

*Calibrating anthropogenic phosphorus sources.* Consideration of wetlands and shallow lakes improves the fit of the model so that measured and modeled phosphorus concentrations do not vary in undeveloped lakes. Review of model results for developed lakes showed that the measured phosphorus concentrations were lower than the concentrations predicted by the model but higher than those predicted for a case where no development was present. From this, it can be concluded that some portion of the potential anthropogenic phosphorus load is being expressed in the lake but that the model must be recalibrated to account for the portion of the development load that is not expressed. Several modifications are required to reduce the large positive bias (overprediction of measured phosphorus) observed in developed lakes.

*Incorporation of phosphorus retention by soils.* The water quality model for lakes in the Muskoka River watershed incorporates a substantial departure from other Ontario models in its geochemical assumptions regarding phosphorus movement from septic systems to lakes. Both the original Dillon and Rigler (1975) and Ontario models (Dillon et al., 1986) assumed that all septic system phosphorus generated within 300 m of the shoreline would ultimately migrate to the lake. This assumption may be considered reasonable as a conservative approach but has never been tested directly.

Since the publication of the original models, direct monitoring studies and mechanistic understanding of soil and phosphate interactions have provided evidence that conflicts with the original assumptions. Mechanistic evidence (Stumm and Morgan, 1970; Jenkins


*Figure II.17.5* Quaternary Geology of Muskoka, Ontario (from Barnett et al., 1991). Pink areas (1) denote igneous and metamorphic bedrock, which is exposed or covered with thin drift. Blue areas (24) are glaciolacustrine deposits of clay and silt. Remaining colors are glaciolacustrine deposits of outwash and till, sand and gravel (18, 23, 25) or organic peat and muck (32).

et al., 1971; Isenbeck-Schroter et al., 1993) and direct observations made in septic systems (Willman et al., 1981; Zanini et al., 1997; Robertson et al., 1998) all show strong adsorption of phosphate on charged soil surfaces and mineralization of phosphate with Fe and Al in soil. The mineralization reactions, in particular, appear to be favored in acidic and mineral-rich groundwater in Precambrian Shield settings (Robertson et al., 1998), such that over 90% of septic phosphorus may be immobilized. The mineralization reactions appear to be permanent (Isenbeck-Schroter et al., 1993), and direct observations suggest that most septic phosphorus may be stable within 0.5 m of the tile drains in a septic field on the Precambrian Shield (Robertson et al., 1998).

The mechanistic and geochemical evidence is supported, in part, by trophic status modeling. Dillon et al. (1994) reported that only 26% of the potential loading of phosphorus from septic systems around Harp Lake, Muskoka, could be accounted for as measured phosphorus in the lake. The authors attributed the variance between measured and modeled estimates of phosphorus to retention of septic phosphorus in thick tills in the catchment of Harp Lake. Although the Muskoka watershed is frequently characterized as an area of thin to no soils over bedrock, this description is in no way universal. The central corridor of the watershed (in which Harp Lake is located) occupies a glacial outwash plain of alluvial sands and gravels, and many catchments contain substantial soil deposits. Western and southwestern Muskoka represent the more typical topography of thin soils and granite ridges and outcrops (Figure II.17.5). Even in these areas, however, tile fields are often, by necessity, built on imported fill and so some attenuation is possible.

Revisions to trophic status models should use the findings of these recent studies to improve the positive bias (i.e., overprediction of measured phosphorus) in the model by accounting for a 74% retention of septic phosphorus for those lakes with suitable soils in their catchments (Dillon et al., 1994). The positive bias is apparent in model results for all developed lakes, but most pronounced in heavily developed lakes. All of the study lakes can be located upon a map of surficial geology (i.e., Southern Ontario Engineering Geology Terrain Study; Mollard, 1980). Those lakes situated within the same types of thick till as are found around Harp Lake, and those lakes situated within outwash or alluvial plains can also be assumed to retain 74% of their septic phosphorus in soils without expression in the lake. The model should therefore be coded to reduce the septic phosphorus contribution by 74% for those lakes. Soil classifications (from Mollard, 1980) for which 74% retention can be assumed are:

MG/R = ground moraine over bedrock
LD = glaciolacustrine delta
GO = outwash plain

Adoption of soil retention provides substantial improvement to the fit between measured and modeled estimates of phosphorus concentration in recreational lakes. The estimate of 74% may still be considered conservative, as Robertson et al. (1998) and Wood (1993) both reported retentions well in excess of 90% for septic systems located in the Precambrian Shield in general and at Harp Lake, specifically.

*Distance of development from shorelines.* Much of the recreational shoreline development in Ontario is located in one tier around a lakeshore, with a minimum setback of 15 to 30 m between the septic system and the water's edge. As development intensifies, however, and prime shoreline space is occupied, development may occur in a second or third tier back from the lakeshore, with water access from communal docking or bathing facilities. Although it is reasonable to assume that second- and third-tier development will have some influence on water quality, it is unlikely that it will have the same influence as development located on the shoreline. The water quality model, therefore, must find a means of accommodating distance of development from a lake while still including some contribution of phosphorus to the lake.

The Dillon–Rigler and Ontario models both adopted a convention of assuming that 100% of the septic system phosphorus generated within 300 m of a lake would ultimately migrate to the lake. Although it is a useful figure to define the limits of a modeling exercise, it is very difficult to defend technically, given the knowledge of phosphorus geochemistry described above. It also leads to counterintuitive interpretations, in which a septic system located 299 m from the shore has a 100% impact, while one located 301 m back has no impact. The absence of substantiation or a mechanistic basis for the assumption, and the complexity involved in site-specific derivation for a watershed modeling exercise, make it very difficult, however, to propose a valid alternative approach.

Setback of development can be accommodated by modifying the assumption of phosphorus contribution with distance; for example, by coding the model so that:

1. Development within 100 m of the shoreline provides 100% contribution of septic phosphorus (as modified by soil thickness and type, see above)
2. Development between 100 and 200 m has its phosphorus contribution reduced by one third
3. Development between 200 and 300 m has its phosphorus contribution reduced by two thirds
4. Development beyond 300 m has no phosphorus contribution

Although this approach still suffers from arbitrary distinctions it does accommodate the concept of distance of development from the shoreline and phosphorus attenuation by soils.

*Table II.17.1* Usage of Shoreline Residences in the District Municipality of Muskoka

| | |
|---|---|
| Zone 1: Outlying area | 0.82 |
| Zone 2: Close to major highway | 1.23 |
| Zone 3: Close to major urban center | 2.09 |
| Resort unit usage | 1.23 |
| Trailer and camp sites | 0.41 |

All values are given in capita years per year for each residential type.

*Revised per-capita phosphorus contribution.* The original Dillon–Rigler and Ontario models used a figure of 800 gm/C/year as an estimate of per-capita contributions of phosphorus to septic systems from human waste and household cleaning products. This figure was originally derived, in part, from measurements of total phosphorus in septic tanks (Dillon et al., 1986) made between 1965 and 1980. Phosphorus concentrations ranged from 5 to 21.8 mg/L, with a mean of 13.2 mg/L. More recent research conducted by the Ontario Ministry of the Environment. (Gartner Lee Limited, 2002, in preparation) found a range of 4.3 to 13.3 mg/L of total phosphorus in septic tanks serving shoreline residences and an overall average of 8.2 mg/L. The more recent values reflect the limitation of phosphate in laundry detergent in the early 1970s and represent 62% of the phosphorus concentrations used in Ontario's Lakeshore Capacity Study. Strict application of the reduced concentration produces an estimate of 500 gm/C/year (i.e., 800 * 0.62). No figure will be completely accurate, however, and so 100 gm/year was added to the measured value in order to maintain a protective and conservative approach to estimating phosphorus loadings. A water quality model should therefore consider a per-capita phosphorus contribution of 600 gm/year to the septic system.

Shoreline development also adds phosphorus to a lake from the conversion of a forested landscape by clearing and lawn planting, and hardening of soils. Previous Ontario models did not include a contribution from these sources. A model should account for this clearing by including an estimate of 2000 m$^2$ for the average size of the developed portion of each shoreline lot and an increased export coefficient of 17 mg/m$^2$/year ( = 34 gm/lot/year) from those areas.

*Validation of cottage usage figures.* The final requirement for estimating human phosphorus loadings from shoreline development is to obtain estimates of the number of days that residences are used in a year. Seasonally occupied residences will be occupied for fewer days per year than permanent homes, and different usage figures apply to resort units or trailer sites. In the 1970s, the Ontario Lakeshore Capacity Study estimated seasonal and permanent usage of shoreline residences figures as 0.89 and 2.55 capita years/year, respectively, on the basis of a cottage survey conducted in the Muskoka–Haliburton region of Ontario (Downing, 1986). Twenty years later, the District Municipality of Muskoka (1995) undertook a similar study and determined that the overall lot usage had not changed substantially (Table II.17.1). Usage figures did not support the commonly held perception that large numbers of cottages were converting from seasonal to permanent use to accommodate retirees or "telecommuters." Only two lakes had high proportions of permanent residents, and these were close to a major urban center (the town of Huntsville). Regionally specific lot usage surveys should accompany development of lake trophic status models to ensure accuracy.

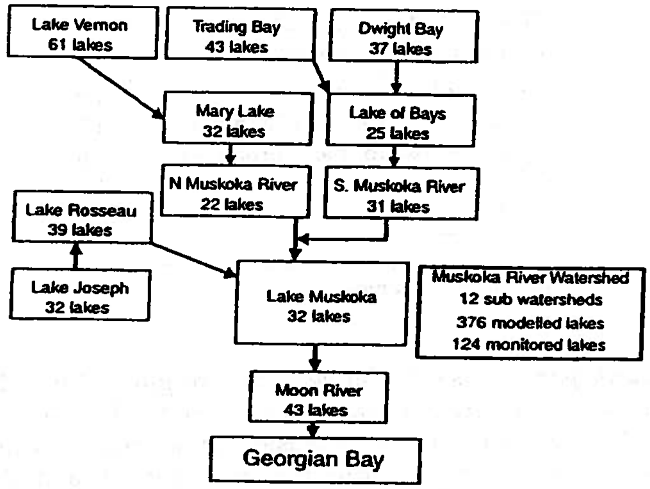


*Figure II.17.6* Schematic of Muskoka River Watershed.

*Other phosphorus sources.* The model should also include anthropogenic sources of phosphorus from other activities in the watershed:

1. Loadings from sewage treatment plants which can be obtained from operating records.
2. Areas of agricultural land use can be determined from air photo interpretation. A coefficient of 45 mg/m$^2$/year can be used to describe export from agricultural land use (Winter and Duthie, 2000).
3. Urban runoff of 45 mg/m$^2$/year for those portions of urban areas within 300 m of a lake or a river. These loadings are not reduced with setback distance, as urban areas are hardened and offer little opportunity for attenuation.

## *Final model results*

The Muskoka River drains a watershed with a total area of 5,737 km$^2$. The watershed model is built around 12 subwatersheds of the Muskoka River, as defined by the major river or lake at the downstream end of each (Figure II.17.6). The hydrologic and nutrient output from each subwatershed enters the next subwatershed downstream, from the western fringes of Algonquin Park at the furthest extent upstream, to Georgian Bay of Lake Huron. The total area of surface water included in the model is 56,000 ha, and the size of modeled lake segments ranges from 3 to 6,200 ha.

The model is both large and complex. It provides explicit characteristics and model output for 376 individual lakes, rivers, or portions of each. The model contains nearly 12,500 individual calculations or data points to calculate total phosphorus concentrations, and approximately 15,000 additional calculations to estimate and track development capacities.

### *Model accuracy*

The final version of the Muskoka Watershed water quality model produced excellent estimates of total phosphorus concentrations in 123 lakes for which monitoring data were available. Mean negative bias in predictions (i.e., underestimate of measured water quality) occurred in 3 lakes and was within 1% of measured values (Table II.17.2). The median

*Table II.17.2* Summary Statistics Showing the Percentage Agreement between Modeled and Measured Estimates of Water Quality in the Muskoka Watershed Model

| | All Lakes | | D.I. < 1.21 | | D.I. ≤ 1.11 | |
|---|---|---|---|---|---|---|
| | +Error | –Error | +Error | –Error | +Error | –Error |
| Average error | 14.74 | –0.84 | 5.89 | –0.28 | 2.83 | –0.28 |
| Median error | 8.08 | –0.51 | 2.81 | –0.28 | 2.34 | –0.28 |
| No. lakes | 120 | 3 | 47 | 1 | 32 | 1 |
| No. > 40% error | 9 | 0 | 1 | 0 | 0 | 0 |

overestimate was 8.1% in 120 lakes. The average overestimate of 14.7% included 9 lakes for which the bias exceeded 40%. The criterion of 40% was chosen as the mean coefficient of variation in measured phosphorus concentration for all lakes. Model errors in excess of 40% were considered unacceptable as they were outside the range of natural variance in water quality. These errors only occurred as a result of positive bias, however, as an indication that not all of the potential human phosphorus load modeled was actually expressed in the lake.

Model accuracy was further expressed by comparison of model error with the proportion of the total phosphorus load contributed by human sources. These sources were described using the "development index" (DI), which was the ratio of total phosphorus load to potential anthropogenic phosphorus load for each lake. A DI of 1.0 represents a lake with no anthropogenic loading, a DI of 1.5 denotes a 50% increase, 2.0 a doubling, and so on.

For 32 "undeveloped" lakes (DI < 1.11) the average overprediction was <3%. Positive bias increased with development intensity such that median and average error were 2.8 and 5.9%, respectively, for 47 moderately developed lakes (DI < 1.21, Table II.17.2). A scatterplot comparison of measured and modeled phosphorus concentrations (Figure II.17.7) shows a) the good correspondence between the two and (b) the tendency for the model to overpredict phosphorus concentration. The positive bias persisted, even after accounting for attenuation of septic phosphorus (see Model Validation section above).

The absolute difference between measured and modeled phosphorus was <1 μg/L for 75 lakes (Figure II.17.8), and error for 77% of the lakes was within 2 μg/L. All of these errors represent positive bias, however, such that the model has a strong tendency to overestimate phosphorus concentrations in developed lakes.

The positive model bias was related to the estimate of phosphorus loads from shoreline development. Median and average positive bias increased with the intensity of development (DI, Table II.17.2), but negative model bias did not change. Positive bias in excess of 10% was confined to developed lakes. The trend to overprediction appeared on undeveloped and sparsely developed lakes but was less than 5% and likely reflected model variance.

The positive bias was reduced by accounting for the retention of phosphorus in soils (see Model Validation). Average and median errors were 9.7 and 5.7%, respectively, for the 66 lakes in which 74% phosphorus retention was assumed on account of the soil characteristics. Error only exceeded 40% on two of these lakes. For lakes where retentive soil mantles were not assumed in the model, average and median error increased to 21.2 and 13.0%, respectively, and 8 lakes had positive bias above 40%. This suggests that some retention may be occurring around all lakes and not just those with thick soil mantles in their catchments.

In spite of the positive bias in model results, it is clear that shoreline development has influenced the phosphorus concentrations in lakes to some extent. Phosphorus concentrations have increased with development, but by far less than predicted on the basis of an

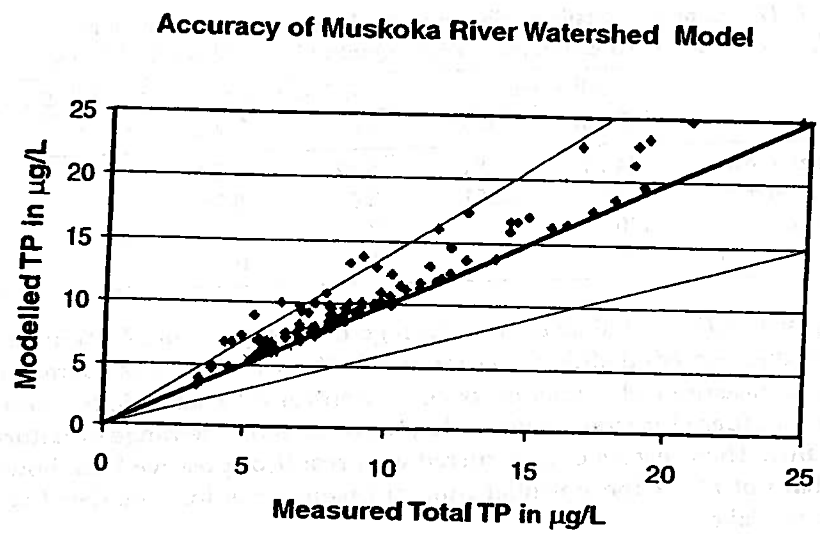


*Figure II.17.7* Accuracy of water quality predictions made with final version of the Muskoka watershed water quality model. Thick line shows perfect fit (1:1), and narrow lines enclose accuracies of ±40%.

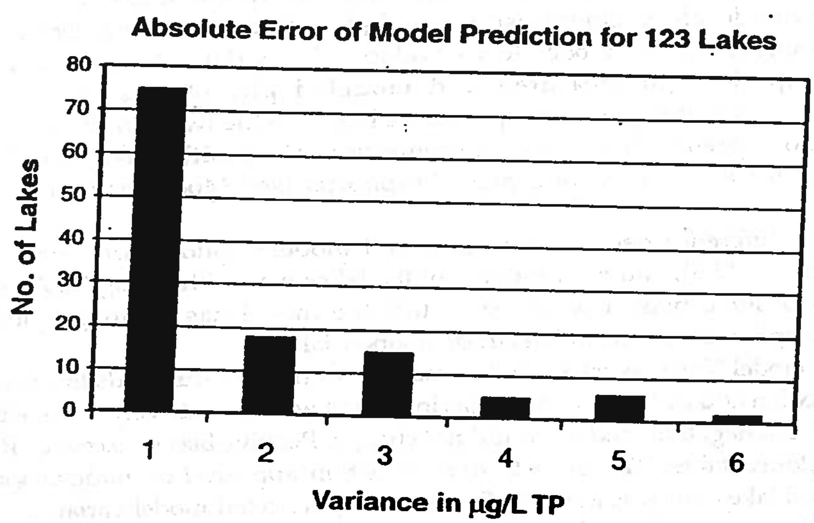


*Figure II.17.8* Absolute difference (in µg/L) between modeled and measured estimates of total phosphorus concentrations in Precambrian Shield lakes.

assumption that 100% of all septic phosphorus is mobile. This observation was supported by the modeling exercise, by the detailed observations of phosphate mobility in septic systems summarized in Robertson et al. (1998) and by geochemical descriptions of phosphate behavior (Isenbeck-Schroter et al., 1993; Stumm and Morgan, 1970). The setting of water quality objectives and development limits must account for these understandings in phosphate behavior and be aware that significant overestimates of the impacts of shoreline development may hinder the implementation and defense of water quality-based development limits.

## Setting water quality objectives and development limits

The intent of a water quality program is to use the monitoring and modeling exercises to support water quality-based shoreline development capacities. Some review of approaches to setting phosphorus objectives is therefore warranted.

### Surface-water management in Ontario

The Ontario Ministry of the Environment (MOE) manages environmental quality primarily through two pieces of provincial legislation: The Environmental Protection Act and The Ontario Water Resources Act. Policies and procedures for management of surface water quality that arise from this legislation are elaborated in implementation documents such as *Water Management: Policies, Guidelines, Provincial Water Quality Objectives of the Ministry of Environment and Energy* (OMEE, 1994). The goal of surface water management in Ontario is *"to ensure that the surface waters of the province are of a quality which is satisfactory for aquatic life and recreation"* (OMEE, 1994).

Ontario established Provincial Water Quality Objectives (PWQOs) in the 1970s in order to meet this goal. The first objectives were mostly adopted from other agencies, such as The International Joint Commission, but were later developed in Ontario (OMEE, 1992).

> *"Provincial Water Quality Objectives (PWQOs) are numerical and narrative ambient surface water quality criteria. They are applicable to all waters of the province (e.g., lakes, rivers and streams) except in those areas influenced by MOEE approved point source discharges. In specific instances where groundwater is discharged to surface waters, PWQOs may also be applied to the groundwater. PWQOs represent a desirable level of water quality that the MOEE strives to maintain in the surface waters of the province. In accordance with the goals and policies in Water Management (OMEE, 1994), PWQOs are set at a level of water quality which is protective of all forms of aquatic life and all aspects of the aquatic life cycle during indefinite exposure to the water. The Objectives for protection of recreational water uses are based on public health and aesthetic considerations"* (OMEE, 1994).

Two policies are used to interpret the water management goal and application of the PWQOs to specific water bodies (OMEE, 1994).

> *"Policy 1: In areas which have water quality better than the Provincial Water Quality Objectives, water quality shall be maintained at or above the Objectives. Although some lowering of water quality is permissible in these areas, degradation below the Provincial Water Quality Objectives will not be allowed, ensuring continuing protection of aquatic communities and recreational uses.*
>
> *Policy 2: Water quality which presently does not meet the Provincial Water Quality Objectives shall not be further degraded and all practical measures shall be taken to upgrade the water quality to the Objectives."*

### Municipal responsibilities for water quality in recreational lakes

The planning system in Ontario is established in the Planning Act, which provides the legislative jurisdiction for municipalities to monitor and regulate land use subject to policy provided in the 1997 Provincial Policy Statement. The Planning Act requires that in reviewing any planning application, a municipality must have regard to provincial policy. The provincial policy respecting water quality states:

> *"The quality and quantity of groundwater and surface water and the function of sensitive groundwater recharge/discharge areas, aquifers and headwaters will be protected or enhanced."*

In practice, municipal policy for water quality protection is interpreted against Ontario PWQOs, and against Policies 1 and 2 for protection of surface water quality, as outlined above. Municipalities are allowed to be flexible in their specific interpretations but must meet the intent of provincial policy. Municipalities can make use of trophic status models to develop shoreline development objectives, which can then be lodged in official plan documents.

Where a development objective exists, policy may then direct that new development shall not result in the predicted nutrient level exceeding the objective for that particular lake. Where a lake is at capacity, development or lot creation may only be permitted in limited circumstance, and in particular where:

1. There is an existing vacant lot of record.
2. The redevelopment of the property would not result in an increase of phosphorus loading to the water body.
3. The septic system and leaching bed can be set back from the waterbody by 300 m.
4. The septic system and leaching bed can be placed in another watershed that is not at capacity

More general policy should also encourage the maintenance of shoreline vegetation, the restoration and preservation of the waterfront shoreline where it has been artificially altered, and aesthetic factors such as setback, footprint or coverage of built structures. Site Plan Control is a planning instrument which can be used to implement site-specific development details.

Ontario's Planning Act requires that municipal official plans be reviewed from time to time to ensure that they continue to provide appropriate policy direction. Such review should include the need to update water quality objectives and lake models to ensure that the most recent scientific understanding of nutrient loading is properly incorporated.

## *Ontario's existing PWQO for total phosphorus*

The existing PWQO for total phosphorus was developed in the late 1970s (OMEE, 1979). It drew on the trophic status classification scheme of Dillon and Rigler (1975) to protect against aesthetic deterioration and nuisance concentrations of algae in lakes, and excessive plant growth in rivers and streams. The rationale (OMEE, 1979) acknowledges that elemental phosphorus can be toxic but that it is rare in nature and so toxicity is rarely of concern. (In fact, there is only one documented case of elemental phosphorus poisoning an aquatic (marine) system in Canada.) Instead, the purpose of the objective was to protect the aquatic ecosystem from nontoxic forms of phosphorus: *"phosphorus must be controlled, however, to prevent any undesirable changes in the aquatic ecosystem due to increased algal growth ..."* (OMEE, 1979).

The 1979 PWQO was given the status of "guideline" to reflect the uncertainty about the effects of phosphorus and to acknowledge the difference between managing toxic and nontoxic pollutants.

> *"Current scientific evidence is insufficient to develop a firm objective at this time. Accordingly, the following phosphorus concentrations should be considered as general guidelines which should be supplemented by site-specific studies:*
>
> *To avoid nuisance concentrations of algae in lakes, average total phosphorus concentrations for the ice-free period should not exceed 20 μg/L;*
>
> *A high level of protection against aesthetic deterioration will be provided by a total phosphorus concentration for the ice-free period of 10 μg/L or less. This should apply to all lakes naturally below this value;*
>
> *Excessive plant growth in rivers and streams should be eliminated at a total phosphorus concentration below 30 μg/L."*

## *Total phosphorus and the PWQO development process*

There are several shortcomings with Ontario's existing PWQO for total phosphorus and the province has been reviewing its approach to phosphorus management. The approach is derived from that first proposed in Hutchinson et al. (1991), and elements of it are summarized in this section. These can be considered in cases where a municipality wishes to derive its own phosphorus objectives to assist with managing shoreline development.

### *Phosphorus as a pollutant*

Development of a water quality objective for total phosphorus is distinctly different from that for toxic substances. Most aquatic pollutants are directly toxic to some target tissue, such as the fish gill, even if some of them are required nutrients at trace amounts, i.e., copper or zinc. As a result, the health of aquatic organisms, and hence the ecosystem, declines rapidly at concentrations slightly above ambient levels (Figure II.17.9). Phosphorus, on the other hand, is a major nutrient. Concentrations can increase substantially with no direct toxic effects. In fact, the first response of the aquatic system is increased productivity and biomass. Beyond a certain point, however, indirect detrimental effects become apparent, which ultimately decrease system health.

The first detrimental responses of a lake to enrichment (i.e., water clarity, algal blooms) are aesthetic and of concern mostly to humans. Assessment of aesthetic impacts is highly subjective; perceived changes in water clarity are based largely on what one is used to (Heiskary and Walker, 1988). The development of a phosphorus objective must therefore acknowledge an element of subjectivity in dealing with human concerns. The objective-development process may also consider that aesthetic impacts begin where a change in water clarity is first noticeable to the human eye, or where the mean water clarity first exceeds natural variation. Unfortunately, human perception of water clarity has not been established. Existing guidelines are based on trophic status classification schemes. They do not consider other water clarity influences such as inorganic turbidity or dissolved organic carbon or how lake users perceive changes in water clarity.

Finally, trophic status indicators such as water clarity, chlorophyll "*a*," or dissolved oxygen cannot be managed directly, but only through management of phosphorus. In addition, there may be delays of up to decades between the addition of phosphorus sources

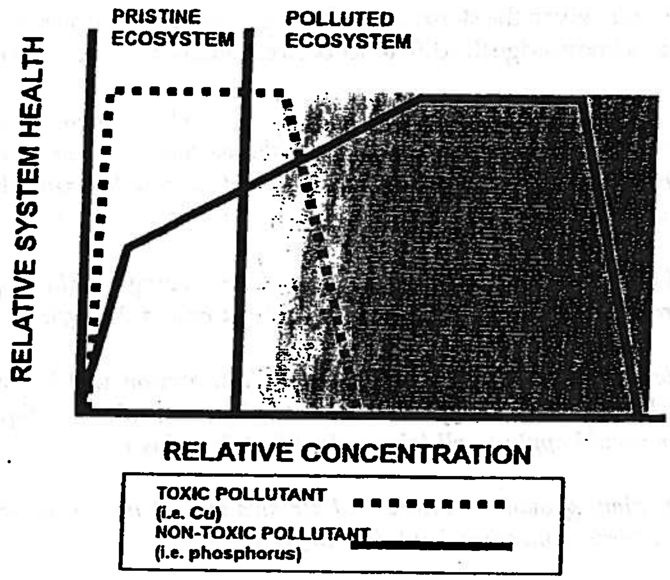


*Figure II.17.9* Generalized responses of an ecosystem to toxic and nontoxic pollutants.

to a watershed (i.e., septic systems), its movement from the source to surface water (Robertson, 1998), and its expression as a change in trophic status. As a result, phosphorus management in Ontario requires the extensive use of models relating shoreline development to the trophic status of the receiving water. Phosphorus management may therefore be considered as a process of "predicting the predictor." Previous sections of this chapter have emphasized the importance of validating or accounting for these assumptions.

### *The shortcomings of a numeric phosphorus objective*

Ontario's existing two-tiered numeric objectives for total phosphorus obscure fundamental differences between lake types and their nutrient status in the absence of human impact. Most Precambrian Shield lakes are characterized by excellent water quality as represented by low concentrations of total phosphorus. For example, the average of the mean annual measured concentration for 123 lakes in the Muskoka watershed was 9.3 ±4.2 µg/L and 62% of the lakes averaged 10 µg/L or less (Figure II.17.10). Within any set of lakes, however, there is still a large diversity of water clarity, controlled by both total phosphorus concentrations and dissolved organic carbon.

The provincial phosphorus objective allows lakes that currently contain less than 10 or 20 µg/L of phosphorus to increase to a maximum of 10 or 20 µg/L. The logical outcome of this two-tiered objective is that, over time, all recreational waters would converge on one or the other of the water quality objectives. This would produce a cluster of lakes slightly below 10 µg/L and another slightly below 20 µg/L, decreasing the existing diversity in water quality in lakes and, with it, the diversity of their associated aquatic communities.

The second shortcoming is that, over time, some lakes would sustain unacceptable changes in water quality while others would be unimpacted, producing both ecological and economic asymmetries as the resource was developed. A lake with a natural phosphorus concentration of 4 µg/L is a fundamentally different lake than one that exists at 9 µg/L. Both lakes, however, would be allowed to increase to 10 µg/L under the existing PWQO. One lake would experience no perceptible change (9 to 10 µg/L) and be overprotected, but the other (4 to 10 µg/L) would be underprotected and change dramatically. In both cases, human perceptions of aesthetics are ignored in the objective. Allocation of phosphorus loadings between these two lakes would be unfair as well. The higher-phosphorus lake could

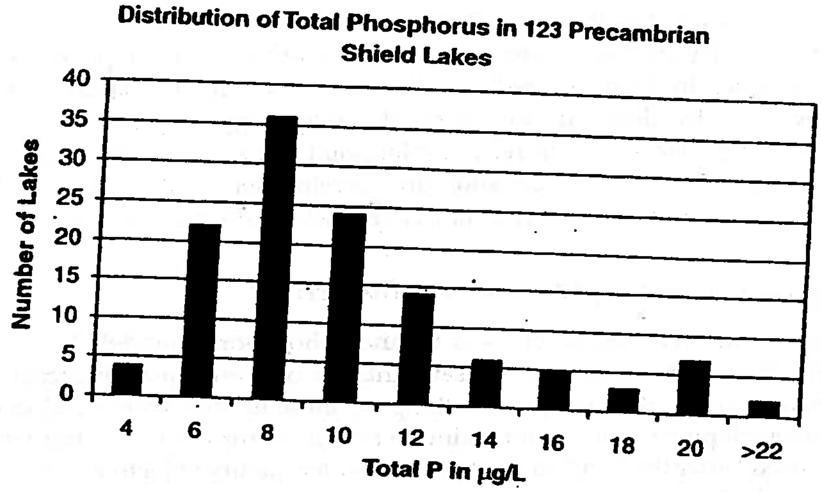


*Figure II.17.10* Distribution of total phosphorus measurements in 123 lakes in the Muskoka River watershed, 1990–1998.

sustain a greater change than the low-phosphorus lake but would be restrained to a much lower load.

In summary, the existing two-tiered numeric objectives overprotect some lakes and do not protect others adequately. Allocation of phosphorus loadings is unnecessarily restricted in some lakes and overly generous in others. Neither biotic nor aesthetic attributes are adequately protected. Over time the diversity of trophic status presently represented in Ontario will decrease as lakes converge on one of two numeric objectives.

### *Environmental baselines and measured water quality*

An emerging concern in environmental assessment is the need for a standard baseline for comparison against environmental change. Inland lakes respond quickly to point source phosphorus inputs. Detection of change is much more difficult, however, for nonpoint sources such as leachate from domestic septic systems.

The incremental nature of shoreline development (no lake is ever developed all at once) results in a slow and gradual increase in trophic status. The high degree of seasonal and annual variance in phosphorus levels in lakes (Clark and Hutchinson, 1992) means that changes may not be detectable without an intensive monitoring program, based on many samples and a precise and replicable analytical method.

Human observers may not notice a slow increase in trophic status over a generation. Environmental change that occurs over one generation becomes the status quo for the next. Over a long period, therefore, any assessment baseline based on measurements of total phosphorus will increase.

Any phosphorus objective that relies exclusively on measured water quality will therefore suffer from:

- Detection problems due to natural variance and analytical problems
- The lag time between addition of phosphorus to a watershed and its expression in a lake
- Failure to detect incremental changes in water quality
- Human perceptual conditioning, which reduces the apparent change in water quality over time

As a result, an increasing assessment baseline and incremental increases in nutrient loading may slowly degrade water quality past any objective. Impacts will accumulate by virtue of delay in their expression, repetition over time and space, extension of the impact boundary by downstream transport, or by triggering indirect changes in the system, such as anoxic sediment release. Nonpoint source phosphorus pollution, particularly from septic systems serving shoreline development, is thus an excellent example of a pollutant that may produce cumulative impacts to the aquatic environment.

## *An ecologically sound approach to objective setting*

The emergence and validation of mass balance phosphorus models for lakes offers an opportunity to correct some of the disadvantages of water quality measurements and conventional assessment techniques. In the past, these models were used to establish the amounts of development that would maintain trophic status within the numeric objectives. They were used "after the fact," to implement a water quality objective. The recommended approach would be to use the water quality model itself to set lake-specific phosphorus objectives.

### *The merits of a modeled assessment baseline*

The basis of the revised approach is increased reliance on water quality modeling in the objective setting process. Recent advances in trophic status models allow us to calculate the "predevelopment" phosphorus concentrations of inland lakes (Hutchinson et al., 1991). This is done by:

1. Modeling the total phosphorus budget for the lake
2. Comparing the predicted concentration to a reliable water quality measurement to validate the modeled result
3. Subtracting that portion of the budget attributable to human activities

The main advantage of the modeling approach is establishment of a constant assessment baseline. A modeled "predevelopment" baseline is based on an undeveloped watershed and so will not change over time. This serves as the starting point for all future assessments. Every generation of water quality managers will therefore have the same starting point for their decisions, instead of a steadily increasing baseline of phosphorus measurements.

The new approach therefore proposes phosphorus objectives based on modeled "predevelopment" phosphorus concentrations. This will provide water quality managers with:

1. A constant assessment baseline
2. A buffer against incremental loss of water quality
3. A buffer against variable water quality measurements

The "predevelopment" phosphorus concentration should not be interpreted as the objective itself. Pristine or "predevelopment" phosphorus levels have not existed in Ontario's Precambrian Shield lakes for over a century, and their attainment is not cost-effective on heavily developed lakes. The modeled "predevelopment" concentration only serves as the starting point for the objective and a reference point for future changes.

The degree of lake development can also guide the selection of a modeled baseline for use in objective derivation. The recreational settlement of lakes in Ontario began in the late 1800s and was very well advanced when trophic status models emerged 100 years later. There was no way to confirm how much of the phosphorus from a century of development was already being expressed in the lakes, and new development was added every year. Measured phosphorus concentrations were therefore higher than the natural baseline but potentially lower than their final, steady-state levels. The natural variability in phosphorus concentrations, and the lag time before the expression of septic-derived phosphorus (as discussed above), prevents the use of measured values as a planning baseline. Modeled, predevelopment baselines of phosphorus concentrations, validated against local lakes which are not yet developed, are therefore recommended as the starting point for objective development.

A model-based objective has two additional advantages. First, the modeled response of the watershed to future changes is instantaneous. It applies new development directly against capacity, without the intervening decades it takes for phosphorus to move to a lake and be expressed as a measured change in water quality (this approach, however, also requires assumptions on the ultimate mobility of phosphorus, which may not be valid (see Calibrating Anthropogenic Phosphorus Sources)). Second, the trophic status model is based on entire watersheds and so allows explicit consideration of downstream phosphorus transport in the assessment.

One disadvantage of the model-based baseline, however, lies with the inevitable changes in scientific understanding of lakes and watersheds. Any baseline derived from a water quality model is therefore subject to change as improved understanding or refinement produces changes in export coefficients, atmospheric deposition, or quantification of in-lake dynamics. For this reason, a modeling exercise must begin with locally validated coefficients and calibration of the water quality model. Model improvements should also be implemented within a defined schedule of Official Plan review, so that scientific understanding is incorporated at a schedule consistent with measured responses of lakes and the planning process.

### *The merits of a proportional increase*

The second component of the objective is a proportional increase from the modeled predevelopment baseline. The proportional increase accommodates regional variation in natural or "background" water quality through the use of one numeric objective for all lakes. It is, in fact, a broader, yet simpler, application of the regionally specific, multitiered objectives proposed in other jurisdictions as a means of accommodating regional variation in background water quality (e.g., Heiskary and Walker, 1988).

One consideration is to adopt an allowable phosphorus increase of 50% above the modeled predevelopment level from anthropogenic phosphorus sources (Hutchinson et al. 1991). This approach is being considered by the Province of Ontario. Under this proposal, a lake modeled to a "predevelopment" phosphorus concentration of 4 μg/L would be allowed to increase to 6 μg/L from shoreline development or other human activities. Predevelopment concentrations of 6, 10, or 12 μg/L would increase to 9, 15, or 18 μg/L, respectively. A cap at 20 μg/L would still be maintained to protect against nuisance algal blooms.

There are numerous advantages to this approach.

1. Each waterbody would have its own water quality objective, but this could be described with one number (i.e., predevelopment plus 50%).
2. Development capacity and ultimate phosphorus status would be proportional to a lake's original trophic status.
3. As a result, each lake would maintain its original trophic status classification. A 4-μg/L lake would be developed to 6 μg/L and therefore maintain its distinction as oligotrophic. A 9-μg/L lake would be developed to 13.5 μg/L, would maintain its mesotrophic status, and development would not be unnecessarily constrained to 10 μg/L.

4. The existing diversity of trophic status in a region such as Ontario would therefore be maintained over the long term, instead of the ultimate outcome of a set of lakes at 10 μg/L and another at 20 μg/L.

### Alternative water quality endpoints and objectives

Several other potential water quality objectives can also be considered in order to establish reasonable and defensible recommendations for development limits in recreational lakes. There is no one obvious "best" water quality objective. The uncertainty in objective setting for phosphorus and its status as a nontoxic pollutant (see Phosphorus as a Pollutant) mean that there will be an element of subjectivity in any figure that is ultimately used.

*Background + 40%.* One refinement could consider the "predevelopment" background phosphorus concentration plus an increase corresponding to the variance in phosphorus concentrations detected by a monitoring program. The average of the mean annual measured concentration for 123 lakes in the Muskoka River watershed was 9.3 ± 4.2 μg/L. Interannual variance in the mean total phosphorus concentration can be substantial, however, and ranged from 10 to 125% in individual lakes. The mean coefficient of variation for annual phosphorus measurements was ± 40% or 4 μg/L. A routine water quality monitoring program is, therefore, capable of determining the mean annual phosphorus concentrations in lakes to within 40% of the true value in any given year. More intensive, research-level programs, however, can detect changes of ± 20% between years (Hutchinson et al., 1991; Clark and Hutchinson, 1992), but this level of precision is not always attainable in routine programs.

This variance of ± 40% represents the ability of a routine monitoring program to detect changes and the natural variance in runoff or biotic factors from one year to the next. Any changes of 40% or less may not, therefore, be detectable over the long term, would be perceived as routine variance in water quality by lake users, and so may represent an objective based on the ability to detect change.

*Background + 4 μg/L.* A standard increase of 4 μg/L of phosphorus from background may also be considered and modeled as an objective. The figure of 4 μg/L represents the natural variance in phosphorus concentration determined by a routine monitoring program but is expressed as an absolute limit instead of a proportional change. This approach is not recommended, however, because the "proportional" increase (40 or 50%) provides a fairer allocation of development to a diversity of lakes. An absolute increase of 4-μg/L allows a doubling of phosphorus from development in nutrient-poor waters (i.e., 4–8 μg/L) but only a 25% increase for mesotrophic waters (i.e., 16–20 μg/L).

*Existing + 10%.* Reviews of the model accuracy showed that much of the septic phosphorus loading predicted by the model was not expressed as increases in the measured phosphorus concentration in lakes. A final approach would acknowledge this lack of response but still provide a margin of error for future changes. In this scenario, the existing measured phosphorus concentration would be used as a baseline and development allocated to a modeled increase of 10% higher than the measured mean. The small increase would acknowledge the uncertainty in the assumptions of phosphorus migration from septic fields so that development would proceed cautiously. Changes in the measured response would be reviewed against monitoring results every 10 years. If lakes began to respond to shoreline development, then the objectives could be reviewed against the nature and degree of response and development limits revised accordingly. Although this approach represents an adaptable response to uncertainty in model assumptions and maintains a proportional increase, it does not provide the desired attribute of a stable baseline over time.

*Background + 50%.* The Province of Ontario has considered a revised water quality objective for total phosphorus in surface waters which is based on a 50% increase in anthropogenic loadings above the modelled natural background. Although the figure of 50% can be debated, it does reflect the merits of a proportional increase and a modelled baseline, as discussed previously. Municipalities must consider Provincial Policy, and so an objective of "Background + 50%" is used here to illustrate the implications of a potential Provincial Water Quality Objective.

### *"Filters" and water quality objectives*

A water quality objective is not the only determinant of development capacity for lakes. Other physical factors can be considered as "filters," additional constraints to development that will modify any numeric objective developed for water quality. They apply equally to any water quality objective. Review of water quality objectives against other development filters helps to determine which aspects of a lake are most limiting to development and to place the water quality objective in the context of other capacity determinants.

*Perimeter filter.* The first such filter is shoreline perimeter, or the availability of waterfront lots based on physical constraints to development. The amount of lakeshore is limiting for any lake; some Ontario municipalities, for example, require a minimum lot frontage of 200 ft for new lots. Other municipalities may adopt larger or smaller lot sizes as a response to narrow embayments or other biophysical limits. Sensitive wetland areas, steep topography, or lack of soil may impose additional physical constraints that exclude portions of a lakeshore from development or require larger lot frontages.

In many cases, particularly lakes at the low end of the watershed, perimeter may be a more restrictive development limit than water quality. Total shoreline perimeter for each lake in the model can be determined using a geographic information system (GIS). The perimeter is then divided into 200-ft lots to provide an estimate of maximum shoreline available for development. This presents an overestimate, however, as steep shorelines or other physical constraints to development may further reduce the number of developable lots. These must be assessed on a lake-by-lake basis.

*Crown land filter.* A second physical filter possibly reducing development potential is consideration of "Crown" land. Many lakes in Ontario are surrounded by "Crown" land, publicly owned land managed by the province in the name of Her Majesty, the Queen. These lands are not developed at present and cannot be subdivided by private interests unless their status is revised by the province. This has been an uncommon occurrence in southern Ontario and is unlikely to occur in the immediate planning horizon. Modeling of all lakes in a watershed to the water quality objective of "Background + 50%" (including Crown land lakes) thus overestimates their ultimate phosphorus loading. In a watershed-based water quality model, the phosphorus loading from these lakes is accumulated in the anthropogenic load for downstream lakes, thus restricting future development. Because Crown lands are unlikely to be developed (and any consideration will involve extensive consultation between the Crown and municipal governments), the potential future load from shoreline development on these lakes can be removed from the modeling exercise as an additional limit or "filter" on development.

*Vacant lot filter.* Many lakes contain vacant lots on their shorelines. These lots have been legally created but have not yet been developed. Owners of these lots retain the legal right to build on them at any time in the future and so their potential phosphorus load must be subtracted from the future development capacity to account for their ultimate development. The 123 monitored lakes that are used in this exercise contain 4,400 vacant shoreline lots in addition to over 18,000 developed shoreline lots. These vacant lots represent a limit to future development of new lots and should be removed from the future capacity of all lakes in the model as an additional filter.

*Lake trout habitat filter.* The final filter which can be considered is based on the presence of lake trout (*Salvelinus namaycush*) in lakes. The requirement of the species for cold, hypolimnetic waters and high-oxygen tensions makes it particularly vulnerable to phosphorus loading and associated oxygen demand (MacLean et al., 1990). Management of lake trout and their habitat remains a responsibility of the federal and provincial governments in Ontario and so lake-specific habitat requirements are not managed by municipalities. Nevertheless, any program must consider the reduced phosphorus contribution imposed by development restrictions on lake trout lakes to acknowledge their potential contribution to downstream capacity.

In practice, some development of these lakes may be allowed, subject to review by provincial agencies. The Ministry of Natural Resources (MNR) and the Ministry of the Environment (MOE) model those lakes that support lake trout populations. In application the MNR will allow development on lake trout lakes up to a limit based on hypolimnetic oxygen status. The MOE and the MNR determine this limit through a separate modeling exercise. When the MNR has identified a problem of depleted oxygen in a lake that supports lake trout, no further development is assumed for that lake.

*Watershed-based development limits*

Water quality models are set up to account for phosphorus movement from one lake to the next lake downstream, and hence throughout the watershed (Dillon and Rigler, 1975; Dillon et al., 1986). The need for such "watershed-based planning" has long been encouraged by provincial management agencies, but specific guidance has been limited. Watershed-based trophic status models include all hydrologic and phosphorus sources in a watershed and so meet the intent of watershed-based planning. The models add the total phosphorus load from one lake to the load for the next lake downstream, after accounting for in-lake retention. As a result, export of development-derived phosphorus from upstream lakes makes up part of the development loading for downstream lakes and must be included as part of the contribution from shoreline development. High levels of shoreline development on upstream lakes will therefore limit the development potential of downstream lakes. Sensitive downstream lakes will also limit development upstream.

The greatest problem with implementing development limits on a watershed basis is determining the extent of upstream influence of a lake that has reached capacity. Strict interpretation of watershed-based planning involves extending development limits to all lakes upstream of any lake that has reached its development capacity. This is logically consistent, as some of the phosphorus that enters the upper end of a watershed will ultimately reach the lower end of the watershed.

Although it is logically consistent to account for downstream transport of phosphorus in setting development capacities it is difficult to develop a spatial limit governing how far upstream of a capacity lake any limit on development should extend. The implications are considerable. If a lake in the lower watershed had no additional development capacity, then strict interpretation of a watershed-based planning policy would see future development restricted on all upstream lakes to prevent downstream transport of phosphorus.

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In many cases, significant urban development and shoreline development potential may be located in the upper watershed. The trophic status of downstream lakes could, therefore, in the strictest sense, prevent any further development upstream.

In the practical sense, the watershed manager must determine the balance between protecting recreational water quality through watershed-based planning and the strictest approach, which would limit all development upstream of a capacity lake. Watershed-based planning remains an attractive concept, but quantitative advice on its implementation limits its utility.

### *Implications of watershed-based phosphorus limits*

One approach to watershed-based planning is to first assess its implications:

1. Will the standards of water quality protection be reduced if development is not limited upstream of a lake that has reached capacity?
2. Does the added protection of a watershed-based approach translate into measurable or predictable improvements in water quality?

The implications can be assessed by running the watershed model in two scenarios:

1. In the first, future development is added to every lake in the watershed up to the limits prescribed by the "Background + 50%" or any other objective. These total future loads to upstream lakes are added to the downstream loading before downstream capacities are set. This approach represents true watershed-based planning.
2. In the second scenario, all lakes are developed to their individual limits, based on the difference between present day development and development to "Background + 50%" without accounting for the additional load from future upstream development. The phosphorus from this future development is then added to the downstream lakes, which are already at their own, independent capacity limits. This approach represents the implications of ignoring watershed-based planning.

The results of this exercise show that the implications of allocating development capacity independently of downstream transport are not significant for most lakes. Figure II.17.11 shows little overall deviation between watershed-based and independent allocations of development to the limits of the "Background + 50%" objective for lakes in the Muskoka River watershed. Lakes where the two development outcomes do not differ are shown as a straight 1:1 relationship. Departure from the 1:1 relationship, where the figure shows points above the 1:1 line, represent the degree to which the final water quality exceeds or "overshoots" the "background + 50%" objective because of upstream loading. The maximum deviation is 4.4 µg/L, in which a lake that should be at 11.1 µg/L ends up at 15.5 µg/L. (Figure II.17.11 shows one point with a very high deviation. This lake has already exceeded its objective as a result of upstream agricultural inputs.)

The ultimate, post-development phosphorus concentrations remained within 20% of the "Background + 50%" objective in 93% of the lakes modeled (Figure II.17.12). Water quality in 12 of the 376 modeled lakes could overshoot the objective by 40% or more as a result of downstream phosphorus transport. The water quality in these individual lakes would still be better, however, than that allowed if all were developed to Ontario's present-day Provincial Water Quality Objectives of 10 or 20 µg/L. The implications of exceeding the revised objective are therefore minor, and less than those of adhering to the present objective.

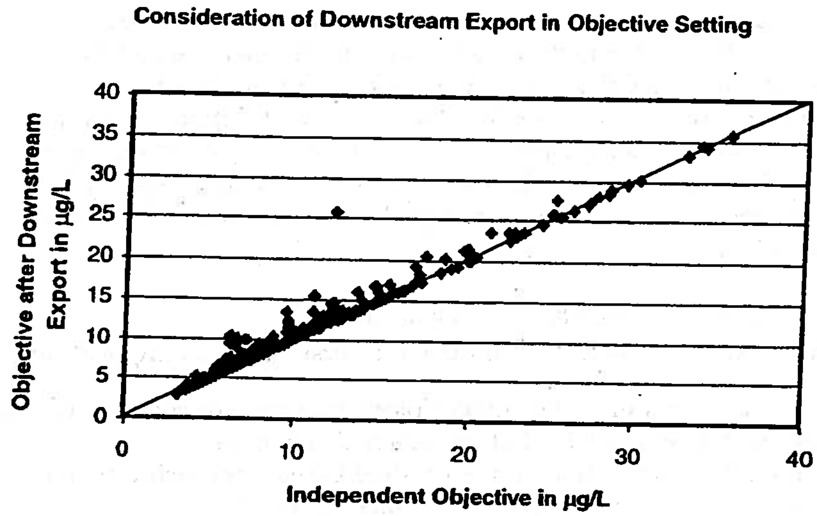


*Figure II.17.11* Implications of independent versus watershed-based development allocation.

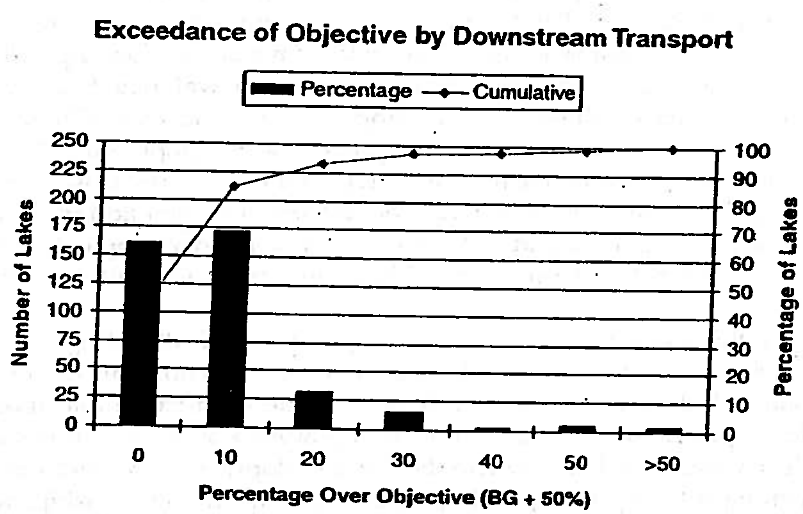


*Figure II.17.12* Number of lakes in which water quality exceeds objective if downstream transport of phosphorus is not considered.

Although it is logically consistent to account for downstream transport of phosphorus in setting development capacities, it is difficult to develop a spatial limit governing how far upstream of a capacity lake any limit should extend. It is therefore clear that watershed-based planning should be interpreted and implemented in a reasonable fashion. The model already accounts for downstream transport of anthropogenic phosphorus from all existing development. The scenarios tested here reveal that the implications of not implementing a development freeze upstream of capacity lakes are minor and not necessary to ensure that water quality is protected. It is therefore evident that a watershed-based orientation of the trophic status model provides for sustainable levels of water quality. We note, however, that this conclusion is based on the relatively low loads associated with shoreline cottage development. Higher loadings, for example from point source, agricultural, or urban inputs, may have a greater impact on downstream water quality.

## *Setting development objectives*

The objective-setting approach must protect water quality as well as assess the implications of model assumptions to policy for shoreline development. Development and calibration of the model for the Muskoka River watershed produced accurate estimates of water quality of the subject lakes, after accounting for reduced phosphorus loading to septic systems and the role of soils in attenuating phosphorus migration from shoreline septic systems to lakes. An objective of "Background + 50%" was tested to correspond to provincial initiatives and filters were developed to compare water quality and other determinants of development capacity. The resultant development capacities can then be summarized for each lake, each subwatershed, and the entire watershed. The capacities can then be compared with existing development density to provide insight into the degree of social stability and public expectations of policy.

Development objectives are best expressed as a phosphorus load in policy, so that the model can be used to compare resort, point source, or other development loadings against objectives. For ease of interpretation at the implementation stage that load can be converted to "Seasonal Residential" development using the occupancy figures for individual lakes (see "Validation of cottage usage" figures), or to any other type of development.

For this exercise, water quality objectives were stated as the total allowable anthrpogenic phosphorus load in kg. Objectives were set for each of the 376 lakes, bays, or rivers in the Muskoka River watershed as an index of potential development load, and for the 123 "monitored" lakes as an index of the loading of those lakes that are currently managed.

Although some jurisdictions may wish to optimize the allocation of future development to each lake, optimization was not attempted in this exercise. Certain popular lakes may well benefit from the adoption of stricter upstream controls on development, in order to maintain development opportunities in the popular lakes. Optimization would involve reducing the development capacity upstream of highly desired lakes, in order to maximize development of preferred locations. There is a near-infinite number of optimization strategies inherent in a large watershed, and optimization would involve a variety of stakeholders with different interests. Optimization could be considered on a case-by-case basis where there is a need to reallocate development opportunities.

### *Reconciliation of model accuracy and assumptions with objectives*

Although the Muskoka River watershed model produced a very good correspondence between measured water quality and modeled estimates on average, there were many lakes in which a large discrepancy between measured and modeled phosphorus concentrations remained in the final model. The objective of a 50% increase in phosphorus against background was used here to illustrate a potential starting point for setting development limits. For this exercise, the 50% increase was modified, based on the agreement of the modeled estimate of phosphorus concentrations with phosphorus measurements for individual lakes, to accommodate the observed (versus the theoretical) expression of phosphorus in lakes. Where water quality measurements do not exist, the model should be assumed to be accurate.

The logic of reconciling objective development with model accuracy can be summarized as follows.

*Criterion #1.* If the measured phosphorus concentration exceeds the modeled "Background + 50%" objective and the modeled total phosphorus exceeds "Background + 50%,"

Then no additional development is allocated (accurate model, lake at capacity).

*Criterion #2.* If the measured phosphorus concentration is 80% or more of the modeled total phosphorus concentration and the modeled total phosphorus exceeds "Background + 50%,"

Then no additional development is allocated (accurate model, lake at capacity); agreement of 80% or better is considered an acceptable indicator of model accuracy.

*Criterion #3.* If the measured phosphorus concentration is between 40% and 80% of the modeled total phosphorus concentration,

Then additional development is allocated up to the "Background + 50%" objective, but the objective is modified (increased) to account for the discrepancy between measurements and modeled estimates (inaccurate model, lake at capacity).

A conservative (protective) approach assumes that 80% of the modeled phosphorus would ultimately be expressed. The objective is therefore modified to Background + (1/0.8 * 50%) = Background + 62.5%.

*Criterion #4.* If the measured phosphorus concentration is less than 40% of the modeled total phosphorus concentration,

Then additional development is allocated up to the "Background + 50%" objective, but the objective is modified (increased) to account for the discrepancy between measurements and modeled estimates (inaccurate model, lake at capacity).

A conservative (protective) approach assumes that 40% of the modeled phosphorus would ultimately be expressed. The objective is therefore modified to Background + (1/0.4 * 50%) = Background + 125%.

*Criterion #5.* If no water quality measurements exist,

Then additional development is allocated up to the "Background + 50%" objective, under the assumption that the model is accurate.

## *Resultant phosphorus objectives for the Muskoka River watershed example*

The total number of shoreline lots on the 123 "measured" lakes was approximately 14,000 in 1999. This corresponds to an anthropogenic phosphorus load of 9,204 kg. The "Background + 50%" objective allows an additional 3,403 kg of phosphorus from shoreline development, after consideration of the perimeter and Crown land filters. This is reduced to 3,010 kg when the vacant lot filter is added. Application of the filter restricting development on lake trout lakes reduces the allowable load to 2,638 kg. Development of all 123 lakes to the water quality objective of "Background + 50%" will thus allow a 37% increase in phosphorus loading from existing levels, or 29% when lake trout and vacant lots are considered.

Including all of the 376 modelled lakes in the exercise produces a corresponding increase in phosphorus loading. There are approximately 18,000 lots on these 376 lakes, corresponding to an anthropogenic phosphorus load of approximately 11,700 kg. The "Background + 50%" objective allows an additional 34,143 kg of phosphorus from shoreline development, or 9,451 kg after consideration of the perimeter and Crown land filters. This is reduced to 8,860 kg when the vacant lot filter is added. Application of the filter restricting development on lake trout lakes reduces the allowable load to 8,071 kg. Development of all 376 lakes to the water quality objective of "Background + 50%" will thus allow a 289% increase in phosphorus loading from existing levels, or 68% when all filters (perimeter, Crown land, vacant lots, and lake trout) are considered. The implications of each of the filters, particularly the perimeter and Crown land filters, are shown in Figure II.17.13. It is clear that water quality alone is not the most sensitive determinant of development capacity, and that physical restrictions are more limiting.

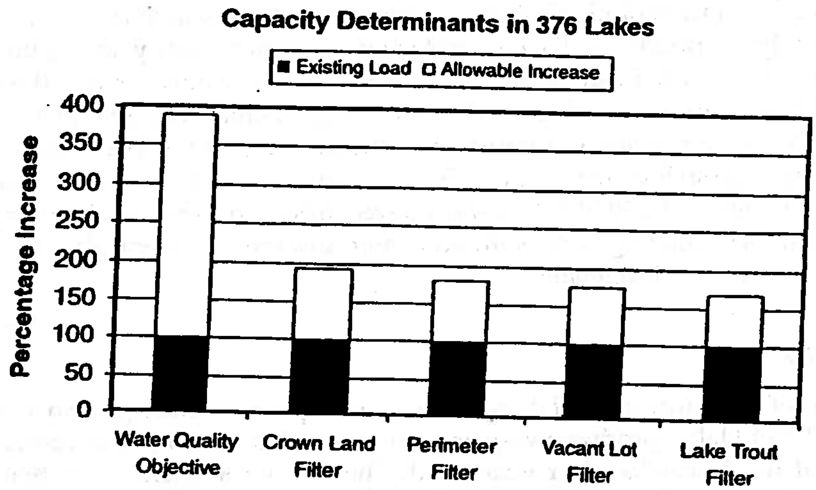


*Figure II.17.13* Comparison of development constraints ("filters") in 525 lakes in the Muskoka River watershed.

## *Value of nutrient-based water quality objectives*

Phosphorus is the nutrient limiting the growth of algae in the nutrient-poor lakes of the Precambrian Shield. When the phosphorus load to a lake increases because of anthropogenic sources and water quality declines, the recreational value of a lake will be diminished. In many municipalities on the southern Precambrian Shield in Ontario, lake-based recreation and tourism are the foundations of the local economy. A mechanism that allows local decision makers to define and understand the carrying capacity (whether based on water quality or otherwise) of the lakes within a municipality will ensure that further development does not unduly stress the natural resources upon which the area depends.

The intent of water quality-based development policies is to protect water quality from eutrophication induced by overdevelopment. It is therefore surprising that water quality is not always the strongest limitation on development capacity in lakes. The Muskoka River example was tested with a "Background + 50%" water quality objective and filters that limited development based on

1. Physical limits of available shoreline ("perimeter")
2. The presence of undevelopable Crown Lands
3. Vacant lots that were already committed to development
4. The presence of lake trout in lakes

Water quality alone did not represent the most significant restriction on shoreline development potential.

The exercise of modeling and monitoring an entire watershed is complex and costly. Water quality-based development limits are a worthy exercise in cases where the effort produces a substantial improvement in water quality protection; for example, if large point sources or urban areas are present. Biophysical and regulatory concerns may constrain development capacity far more, however, than water quality. Consideration of Crown Land and the physical shoreline limits may also reduce development capacity below that allowable under a very conservative phosphorus water quality objective of "Background + 50%." Lake trout habitat may be the most conservative filter and result in the lowest estimate of development capacity.

Simple consideration of physical development constraints may therefore, in a whole watershed analysis, provide sufficient protection of water quality in all but the most sensitive lake trout lakes. In all cases, however, the manager must review the large-scale findings at a finer resolution, as there will always be individual lakes to which the physical constraint does not apply and water quality is the most sensitive determinant. This conclusion applies to shoreline development in the form of recreational or residential development built on individual 200-ft lots. It must be reconsidered, however, for more intensive forms of phosphorus loading such as urban runoff, sewage treatment plants, agriculture, and high-density resort development.

## Conclusions

Incorporation of the most up-to-date science on phosphorus loadings and dynamics in Precambrian Shield lakes produced very accurate estimates of phosphorus concentrations in the lakes of the Muskoka River watershed. The major conclusion was that previous modeling exercises, which assumed that 100% of the phosphorus in septic systems within 300 m of a lakeshore was mobile, could not be substantiated on an empirical or a mechanistic basis. Adoption of soil-based attenuation produced a substantial improvement in predictive capability and defensibility of the planning exercise. Although the lakes showed some response to shoreline development, the degree of response was much less than that originally predicted.

In the final analysis it is clear that water quality is not the most sensitive determinant of shoreline development and that adoption of simple standards such as a 200-ft minimum frontage on shoreline lots may achieve a high level of water quality protection without the need to rely on a whole watershed model of phosphorus dynamics and lake-specific water quality objectives. Some form of water quality assessment is essential, but it may not have to take the form of a complex predictive model — a sensitivity analysis may suffice. Water quality was also well protected in most cases without the need to "freeze" all development upstream of a lake which had reached capacity.

Social pressures and user conflicts are becoming increasingly important in cottage country. These may be partly managed if lake area ("recreational space") is used to help determine cottage density, independently of water quality. It is also clear that properly managed near-shore development, in which minimum lot sizes are coupled with: enhanced septic system setback, ensured naturalization of shorelines, protection of significant natural areas, wetlands, and scenic vistas, modern septic systems, mandatory septic inspection and a strong stewardship program, will likely be as successful in maintaining water quality as will development limits. Expansion of resorts and cluster or subdivision-type cottage development provide the potential to use package sewage plants that are capable of achieving very high effluent quality and that can be monitored to confirm their inputs, thus reducing the need to rely on assumptions regarding the effectiveness of septic systems. Finally, the development of lake-specific management plans by lake users or residents in cooperation with provincial or municipal authorities may bring all users together to draft a common vision of a lake's future and actions to achieve it based on site-specific local concerns and consensus.

In summary, lake management planning extends beyond consideration of plumbing. There will always be some phosphorus enrichment as lakes are developed, whether from land clearing, septic systems, urbanization, or expanded point sources. Recreational water quality must therefore remain a component of lake management. Development capacities based only on phosphorus, however, are costly, complex, and vulnerable to challenge.

Managers are encouraged to consider and implement a broader spectrum of management approaches, in addition to phosphorus-based development capacities.

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# APPENDIX G – MITIGATION OF PHOSPHORUS IMPACTS FROM SEPTIC SYSTEM PHOSPHORUS

## 1.0 Introduction

The purpose of this appendix is to outline briefly the status of research underlying phosphorus removal for application to sewage treatment systems in Ontario's Precambrian Shield. Many of the investigations and related approvals have focused on treatment systems that discharge less than 10,000 litre/day (L/day); however, their application is also suited to facilities discharging greater than 10,000 L/day. As well, all of the research has tested different media through which the partially treated sewage would infiltrate, rather than the batch treatment with coagulant aids such as ferric chloride or alum. A further intent of this review is to set out the technical underpinnings for an approach to reducing phosphorus for an expanded Delawana Inn.

## 2.0 Research Findings on "B" Horizon Canadian Shield Soils

The 1997 **Ontario Building Code** requires that sewage systems be set back 15 metres (m) from shorelines of lakes, rivers and streams. This is designed to protect surface water quality originating in leaching beds on the basis of public health concerns. In recent years, the Ministry of the Environment (MOE) and Ministry of Natural Resources (MNR) have been requesting setbacks for sewage systems and buildings on lakes which exceed those currently required for sewage systems only. As well, some area municipalities also require building and sewage system setbacks in excess of provincial requirements. These requirements have related to both real and potential changes in lake trophic state due to phosphorus migration from leaching beds and related shoreline development, and the resulting effects on lake aesthetics and fish and wildlife habitat. Approval agencies are now viewing more extended setbacks as effective mechanisms to reduce impacts resulting from shoreline development.

The underlying principle of a setback is that, the greater the distance the tile field and development are from the lake, the greater the capacity of the intervening land base to intercept and retain phosphorus. Some of the factors contributing to phosphorus retention by soils include soil chemistry and adsorption capacity, phosphorus concentration in the septic tank effluent and loading rate, composition and density of the vegetation cover, and slope conditions. It is logical to expect variations in the degree of effectiveness of different landscapes to mitigate or eliminate potential trophic state impacts. For example, deep, sandy loam soils supporting continuous forest growth on level sites would naturally retain more phosphorus than shallow, discontinuous mantles over steeply sloping bedrock. More specifically, soil textural and chemical characteristics will influence, to some degree, a soil's ability to retain phosphorus; soil chemistry (for example, aluminum, iron, and calcium concentrations) is perhaps of equal or greater importance in this regard. Information on the specific phosphorus retention capacities of various publications are informative and allow some generalizations. For example, studies on the subsurface movement of effluent from private sewage disposal systems (Brandes 1974 and 1975) indicate that a decrease in phosphorus concentration below and downgradient of the tile fields is generally relatively rapid, regardless of soil type. Based on the studies of Brandes (1974 and 1975), concentrations decreased most rapidly with increasing silt and clay content in the soil; up to 98% of phosphorus was removed in soils containing 40% silt and clay, with significant reductions in concentrations within the first 3 m of the tile fields. Although the reductions are not as dramatic with sandy soils, as much as 78% phosphorus removal was reported by Brandes (1974), and phosphorus concentrations were typically reduced to 0.1 milligrams per litre (mg/L) within 20 m of the tile fields. Brandes (1974) concluded that the bulk of the effluent phosphorus was fixed within the native soil matrix directly below and downgradient of the tile fields, so that phosphorus concentrations reaching the groundwater and/or nearby water bodies were very low, even where sandy soils predominated.

With respect to the movement of phosphorus in soils, it attaches to the soil colloids, either permanently or semi-permanently. Permanent attachment comes in the form of a calcium, magnesium, iron or aluminum phosphate precipitate, depending on the geo-chemistry of the soil, or through covalent bonding to the soil colloid itself. Semi-permanent attachment is via electrostatic attraction of the phosphate with the soil colloid. During periods of high hydraulic loading, the phosphorus cannot get as close to the soil colloid as it can under low hydrological conditions; in effect, the attraction is weak and the phosphorus can travel, at least until the electrostatic binding can hold the phosphorus, or until it is geo-chemically complexed. Over time, the movement of a phosphorus front can be observed in the soil, as reaction sites are used up.

There is no question that identifying, understanding, and confirming the mechanisms which determine the limits to phosphorus attenuation in different shoreline landscapes is important. For example, if the control is soil surface adsorption, then phosphorus mobility will increase when the number of adsorption sites are filled. However, if the controlling process continues indefinitely, as in the case of chemical combination with other effluent and soil constituents to form minerals, then phosphorus transport will not be a concern as long as the septic system continues to operate properly. In our opinion, it is probably not one mechanism or the other which operates in any one situation; rather, both mechanisms are likely in effect at most sites.

Several key studies have been undertaken in Canadian Shield soils, which greatly improved our knowledge on the potential of phosphorus movement from tile beds and the ability of select "B" Horizon soils to mitigate this movement. The relevant research is summarized in the following paragraphs.

First, in 1991, Robertson *et al.* (**Ground-Water Contamination from Two Small Septic Systems on Sand Aquifers**) undertook detailed groundwater monitoring of two single-family residences to determine the impact of septic systems on shallow unconfined aquifers in southern Ontario. One site at Cambridge, located on carbonate-rich sand aquifer, was in operation for over twelve years, while the other site on the Muskoka River near Bracebridge, on a poorly buffered, carbonate-depleted sand aquifer, was in operation for one year. While high levels (about 10 mg/L) of phosphorus were found in the septic tank effluent, concentrations were substantially attenuated immediately below the tile bed, with no detectable phosphate phosphorus (<0.02 mg/L) was observed in the groundwater zone at the Bracebridge site. While the precise mechanism of attenuation was not confirmed, it was suggested that the presence of sparsely soluble phosphate minerals such as stregnite (an iron complex), or varisite (an aluminum based compound), or adsorption were instrumental in controlling movement. A copy of the Robertson et al paper is included as **Attachment A.**

Second, and perhaps the most comprehensive investigation to date to examine septic system derived phosphorus in groundwater and upgradient of Precambrian Shield lakes, was undertaken by Wood (1993), in partial fulfilment of a Master of Science degree at the University of Waterloo. The septic tank-tile field in question was installed in 1962 to serve a shoreline seasonal residence on Harp Lake, northeast of Huntsville. The septic system was situated 0.66 m above the highest elevation of the water table and 15.8 m from the shoreline of the lake at its closest point. Between 1962 and 1992, there was no maintenance to either the tile bed or the approximately 1,800 L steel septic tank. Wood reported that while slightly elevated phosphorus was detected in the groundwater of the terrestrial and aquatic zones, most of the phosphorus from 30 years of use was found directly under the tile bed (within 14 centimetres [cm] of the drains). Soil phosphorus concentrations below and downgradient from this horizon were at background levels, and most of the immobilized phosphorus corresponded to the operationally defined fraction considered to be reactive phosphorus sorbed to metal complexes.

Third, in analyzing hydrologic and mass balance measurements, Dillon *et al.* (1992) reported accurate predictions of total phosphorus concentrations in lakes having no shoreline development. However, the model predicted lower than observed concentrations on the basis of natural phosphorus loads in four developed lakes. In three of the four, the predictions and observations were virtually identical only if the potential total phosphorus contributions from shoreline development was included in the budgets. In the exception, Harp Lake (the same lake studied by Wood [1993]), the only one of the four having thick deposits of till/soil.

The primary conclusions of the Robertson *et al.* (1991), Wood (1993), and Dillon *et al.* (1992) studies are that:

1. little **long term** retention of phosphorus appears to result from shoreline development on lakes having sparse soils; and

2. where reasonable soil conditions prevail, considerable long term retention of a significant part of the artificially-derived phosphorus occurs immediately beneath the leaching bed.

Fourth, in August 1995, Michalski Nielsen Associates Limited (then Michael Michalski Associates) requested further clarification from the MOE's soil experts regarding the movement of phosphorus in tile beds. The following is reproduced from a letter from the Ministry (August 22, 1995, Healy to Michalski); the relevant paragraphs from the transmittal are reproduced below.

> "Phosphorus does not move only during the freshet or high hydraulic loading. The fact is that the phosphorus does not move per se. What happens is that the phosphorus attaches to the soil colloids either permanently or semi-permanently (for lack of a better phrase). Permanent attachment comes in the form of precipitation of the phosphorus in the form of Ca, Fe or Al phosphate depending on the geochemistry, or through covalent bonding of the phosphorus to the soil colloid itself. Semi-permanent attachment is through the electrostatic attraction of the phosphate with the soil colloid. This attachment is semi-permanent because the phosphorus is still mobile because of the soil water chemistry response to other cations and anions.
>
> What happens during period of hydraulic loading is that the phosphorus cannot get as close to the soil colloid so the electrostatic attraction is weaker and the phosphorus travels further. At least until the electrostatic attractions can hold the phosphorus or precipitate it. Over time, we can observe the movement of a phosphorus front in the soil as the reaction site on the soil colloids are used up. Precipitation of the phosphorus will always be a factor but is a response to what cations are available at any given time.
>
> The phosphorus front movement is easily seen in the research work done by the University of Waterloo at the Cambridge research facility over the past 10 years. In fact, most of their work does indicate that phosphorus is mobile, albeit not as mobile as chloride or nitrate but mobile nevertheless."

Fifth, in early November, 1995, information was presented at the annual North American Lake Management (NALMS) Society conference held in Toronto which indicated that while phosphorus movement in sandy calcareous soils occurs at a rate of about 1 m per year, the upper horizon of soils of Precambrian Shield origin has a high capacity for retaining phosphorus due to its naturally high iron content and low pH. The notes in **Attachment B** are reproduced from the conference poster session (**Phosphate Minerals in the Vadose Zone at Septic System Sites** by L. Zanini, W. D. Robertson, D. J. Ptacek, and S. L. Schiff), a presentation (**Septic System Phosphorus in Precambrian Shield Country** by S. L. Schiff, W. D. Robertson, L. Zanini, J. Wood and R. Elgood), and presentation and poster session (**Laboratory Studies on the Development of a Reactive Mixture to Remove Phosphates from Septic System Effluent** by M. J. Baker, D. W. Blowes, W. D. Robertson, and C. J. Ptacek).

Sixth, the recent **Review of Phosphate Mobility and Persistence in 10 Septic System Plumes** by Robertson, Schiff, and Ptacek (1998) is informative and quite conclusive. This paper reviewed phosphate distribution in ten mature septic system plumes, and revealed that in six cases (primarily those on calcareous sands, and south of the southern limit of the Precambrian Shield ), relatively large plumes were present (> 10 metres in length), and phosphate concentrations of 0.5 mg/L to 5.0 mg/L were about two orders of magnitude higher than normally found in uncontaminated aquatic ecosystems. At the other four sites, which are acidic and on Precambrian Shield non-calcareous sands and silt-and clay-rich sediments, high phosphate concentrations occurred only within three metres of the infiltration pipes, clearly demonstrating the attenuating capability of such soils. Concentrations of phosphorus in the plumes appeared to be strongly controlled by mineral precipitation reactions that occur in close proximity to the infiltration pipes. For example, at one site, approximately 85% of the total sewage phosphorus was retained within the 2.0 metre thick vadose zone after 44 years of effluent loading. Because of the importance of this paper, a copy is enclosed as **Attachment C**.

Seventh, **Limnology, Plumbing and Planning: Evaluation of Nutrient-based Limits to Shoreline Development in Precambrian Watersheds** by Neil Hutchinson (2002) sheds new light on the assumptions of phosphorus loadings to a watershed from human sources, reporting from his review of a 25-year data base for over 125 lakes in the District Municipality of Muskoka that the MOE's assumption that 100% of phosphorus associated with septic tank sewage eventually getting to the lake is, in fact, unlikely (**Attachment D**). As well, he challenges the scientific validity of the Ministry's reliance on a 300 m setback as a safe distance beyond which any septic tank related phosphorus will not migrate to the lake. Regarding the former, Hutchinson (2002) points out that, "Both the original Dillon and Rigler (1975) and Ontario models (Dillon *et al.* 1986) assumed that all septic system phosphorus generated within 300 m of the shoreline would ultimately migrate to the lake. This assumption may be considered reasonable as a conservative approach but has never been tested directly... Since publication of the original models, direct monitoring studies and mechanistic understanding of soil and phosphate interactions have provided evidence that conflicts with the original assumptions. Mechanistic evidence (Stumm and Morgan 1970, Jenkens *et al.* 1971, Isenbeck-Schroter *et al.* 1993) and direct observations made in septic systems (Willman *et al.* 1981, Zanini *et al.* 1997, Robertson *et al.* 1998) all show strong adsorption of phosphate on charged soil surfaces and mineralization of phosphate with Fe and Al in soil. The mineralization reactions, in particular, appear to be favoured in acidic and mineral-rich groundwater in Precambrian shield settings (Robertson *et al.* 1998), such that over 90 % of septic phosphorus may be immobilized. The mineralization reactions appear to be permanent (Isenbeck-Schroter *et al.* 1993), and direct observations suggest that most septic phosphorus may be stable within 0.5 m of the tile drains in a septic field on the Precambrian Shield (Robertson *et al.* 1998)." With respect to the second matter (i.e., distance of development from shorelines), Hutchinson (2002) comments, "... The Dillon-Rigler and Ontario models both adopted a convention of assuming that 100 % of the septic system phosphorus generated within 300 m of a lake could ultimately migrate to the lake. Although it is a useful figure to define the limits of a modeling exercise, it is very difficult to defend technically, given the knowledge of phosphorus geochemistry described above. It also leads to counter-intuitive interpretations, in which a septic system located 299 m from the shore has a 100 % impact, while one located 301 m back has no impact." Dr. Hutchinson recommended that the Province's protocol for modeling lakes (i.e., the lake trophic state model) be revised such that the phosphorus contribution from sewage septic systems be reduced by 74% for those lakes with suitable soils in their catchments, which is a major departure in the way the model is presently applied.

Eighth, Robertson's (2003) paper (**Enhanced Attenuation of Septic System Phosphate in Noncalcareous Sediments**), which is one of a series dating back to 1991, confirms his fundamental conclusion that phosphorus is strongly attenuated in acidic Precambrian Shield soils. This paper provides additional support for the likely geochemical attenuating mechanism, showing that under acidic conditions, permanent attenuation is caused by high levels of aluminum combining to produce an aluminum/phosphate compound. The reaction is substantiated by electron microscopic analysis confirming such a complex on sand grains below the infiltration bed. The acidic conditions relate to the breakdown of the components of sewage, particularly ammonia from human wastes (**Attachment E**).

Ninth is the Branson matter which is described in **Section 4**.

**Tenth**, in a special issue, **Scope Newsletter** (January 6, 2006) presented an overview of existing knowledge regarding nitrogen and phosphorus releases from septic tanks and autonomous sewage treatment systems. The newsletter confirmed significant differences between the behaviours of the two nutrients. In this regard, nitrogen is only retained in septic tanks to a small extent, and once the effluent is infiltrated into the soil, it is converted to nitrates which are then very mobile and move with the ground water. On the other hand, phosphorus is significantly retained in the septic tank (i.e., up to 48%), and then precipitated or adsorbed by soils, so that significant quantities of phosphorus rarely move more than a few metres from the point of infiltration. Because of the importance of this newsletter, it is reproduced in its entirety in **Attachment F**.

**Eleventh**, relates to direction provided in the recent scientific publication "A Review of the Components, Coefficients and Technical Assumptions of Ontario's Lakeshore Capacity Model" (**Lakes and Reservoir Management**, 22[1]:7-18, 2006) (**Attachment G**). Of some importance is that all of the authors are employed by Ontario's MOE, except for Dr. Hutchinson who is presently with a private consulting firm. To accommodate the difference between the historical position of the MOE regarding the attenuating ability of soils insofar as sewage-related phosphorus is concerned and the recent science described herein, a three-step graduated approach was suggested. It recognizes that attenuation may occur in some watersheds and probably increases with distance from the lake's shoreline. "First, in watersheds (or portions of watersheds) with shallow [generally < 3 m] or absent soils, and with exposed or fractured bedrock, the existing assumption of zero retention is applied . . . Second, at sites where deeper [generally > c m], non-calcerous native soils are present, the modeller may use the coefficients outlined in Table 3. Here, the degree of attenuation increases with distance from the shoreline, with an assumption of zero export at a distance of > 300 m [Hutchinson 2002]. Third, in cases where site-specific characteristics demonstrate that retention of septic system phosphorus may occur over the long-term, attenuation factors may be developed for consideration by local planning authorities and plugged into the model." It is application of the third option that is relevant for the subject application, primarily because considerable information on soil characteristics were collected and analyzed as part of this trophic state impact analysis.

**Twelfth**, is the recent publication by Roger Lacasse and Naider Fanfan which demonstrates that the Ecoflow biofilter coupled with a 12 inch thick drain field reduces the concentration of phosphorus in the effluent by 98%. The Ecoflow unit itself is responsible for 12% reduction, while the soils attenuate the balance through adsorption into particles of iron and aluminum (i.e., a permanent reaction) (Attachment G).

**APPENDIX H – DISTRICT MUNICIPALITY OF MUSKOKA LAKE SYSTEM HEALTH PROGRAM, AS SET OUT IN OFFICE CONSOLIDATION OF THE OFFICIAL PLAN OF THE MUSKOKA PLANNING AREA (NOVEMBER 2007)**

# Office Consolidation of the Official Plan of the Muskoka Planning Area

November 12, 2007

Prepared by
The District of Muskoka
Planning and Economic
Development Department

**F.10 Notwithstanding that application of the Plan is primarily directed to natural resources within the Muskoka District Area, it is acknowledged that many resource issues do not respect political jurisdiction, particularly those that relate to air or watersheds. As such, the provisions of the Environmental Limitations Section shall be encouraged to be applied in documentation affecting such air or watersheds under other jurisdiction.**

*Inter-municipal Resource Management*

**F.11 The enhancement and preservation of the natural and man-made environments shall be incorporated within any development or redevelopment proposal.**

*Enhance Environments*

**ENVIRONMENTAL LIMITATIONS**

Deleted Sections F.12- F.23 and replace d per OPA #32

**Lake System Health**

Water significantly contributes to Muskoka's geography, biology and cultural heritage. The water that connects Muskoka plays a key role in its economy and lifestyle and represents part of its important natural assets. Therefore, the District of Muskoka has an interest in the protection of all of the water resources within its jurisdiction and it is important that the District continues to be a leader in the protection of this key asset.

In January, 2003, Muskoka District Council approved the Muskoka Water Strategy. The Strategy is a comprehensive framework of integrated strategic initiatives to protect Muskoka's water resources. A significant component of that Strategy is Lake System Health. This program has evolved from the review of the Muskoka recreational water quality model. It incorporates the best available science and a variety of implementation techniques designed to minimize the impact of human activities on water resources, protect and enhance the environmental health and quality of life in Muskoka and protect the future of Muskoka as a premier recreational region and represents good planning. The program is to be implemented in concert with the Area Municipalities and other stakeholders.

**Key Program Components**

The following sections describe key components of the Lake System Health program.

**F.12 The District of Muskoka will, in collaboration with the Area Municipalities and other stakeholders, undertake limits to growth assessments for waterbodies in Muskoka, with those lakes considered to have surpassed an acceptable threshold for phosphorus taking priority. Limits to growth assessments are intended to identify the development limits of a waterbody by using existing base data and applying the various applicable official plan policies to determine potential development capacity. These limits to growth assessments will provide background information for local municipal planning decisions and initiatives and lake plans.**

**F.13 The District of Muskoka will, in collaboration with any affected Area Municipalities, the lake community and other stakeholders, facilitate and participate in remedial action programs for lakes considered to have surpassed an acceptable threshold for phosphorus. The purpose**

**of remedial action programs is to identify areas of degradation or sources of contamination in and around these lakes and to develop a plan with actions to remediate and improve the situation.**

**F.14 The District of Muskoka will promote, and where possible and appropriate, require stormwater management approaches and practices that will protect the health of lakes and rivers within Muskoka.**

**F.15 Stewardship programs engage the local community and empower individuals to care for or remediate specific portions of the watershed. Local stewardship is especially important on waterbodies that have been identified as being Over Threshold or of high sensitivity, as defined elsewhere in this Plan. The District, will, in collaboration with the Area Municipalities, the Muskoka Watershed Council, lake communities and other stakeholders, participate in stewardship initiatives as appropriate.**

**F.16 Recreational water quality monitoring and modeling is an important component of Lake System Health. Recreational water quality will continue to be monitored and modeled by The District of Muskoka as one measure of a waterbody's capacity to sustain development.**

**F.17 The District of Muskoka will continue, through its development review and approvals function, to ensure that water quality is protected and will require Municipalities to adopt provisions in Area Municipal official plans and zoning by-laws in order to achieve this objective.**

### Recreational Water Quality

The single most significant impact on water quality on most recreational lakes and rivers in Ontario is the increased levels of phosphorus, that are entering surface waterbodies. Sources of phosphorus are both natural and man made. Natural sources of phosphorus include such things as precipitation and natural drainage form the watershed. Man made sources of phosphorus include increases in overland flow as a result of disruption in the natural vegetation (leading to erosion) in and beyond the riparian zone, use of fertilizers, increased stormwater run-off from impervious surfaces and effluent from septic systems and sewage treatment plants.

Based on the recreational water quality model as detailed in the report prepared by Gartner Lee Limited in 2005 entitled *Recreational Water Quality Management in Muskoka,* the lakes and rivers in Muskoka have been classified as having high, moderate or low sensitivity to phosphorus. This classification is based on the responsiveness of a waterbody to phosphorus and its mobility within the watershed and will not change.

Lakes of low sensitivity respond only minimally to the input of phosphorus and it is unlikely that development related phosphorus will increase concentrations by more than 50% of the undeveloped phosphorus load. Lakes of moderate sensitivity have some ability to receive phosphorus without a significant decrease in water quality. Where a lake is classified as being of high sensitivity, there is the potential for development to input more phosphorus into a lake than it can sustain, causing the measured phosphorus levels to increase beyond the acceptable threshold.

Where the phosphorus loading to a waterbody exceeds 50% of the undeveloped phosphorus load, the lake or river is considered as being "Over Threshold" for phosphorus loading. "Over Threshold" lakes require a higher level of development control as a precautionary action to protect the long-term health of the lake.

***General Development Policies***

**F.18 The District of Muskoka will maintain a recreational water quality model and monitoring program and will review it on an ongoing basis. This model has been designed to address recreational water quality only and does not include factors to address fisheries values.**

**F.19 Lake and river classifications are identified in Schedule F. Any lake or river not listed is assumed to be of moderate sensitivity unless otherwise identified by Muskoka.**

**F.20 Through the review of the Muskoka recreational water quality program, it has been determined that the overall health of lakes and rivers in Muskoka is very good to excellent and that the cautious approach to development taken in Muskoka has been beneficial. This cautious approach will be continued. In this regard, new lot creation, development or redevelopment will only be permitted where it is determined that phosphorus impacts on water quality can be effectively eliminated.**

**F.21 The role of natural vegetated shorelines in buffering waterbodies from erosion, siltation and nutrient migration adjacent to the sensitive littoral zone is critical to the protection of water quality. Preservation and restoration, where appropriate, of shoreline buffers is therefore required. At a minimum, a target of 75% of the linear shoreline frontage of a lot will be maintained in a natural state to a target depth of 15 metres from the shoreline where new lots are being created and where vacant lots are being developed. Where lots are already developed and further development or redevelopment is proposed, or where the lot is located within an urban centre or community, these targets should be achieved to the extent feasible. Where these targets cannot be met, a net improvement over the existing situation is required.**

**F.22 A minimum 30 metre setback from any shoreline will be required for leaching beds. Where this is not feasible, on-site phosphorus management, as outlined in section F.26, will be required.**

**F.23 A minimum 20 metre setback from any shoreline will be required for all development, excluding shoreline structures. Where this setback cannot be achieved, a lesser setback may be considered where on-site phosphorus management is implemented and in the following circumstances:**

**a) Sufficient lot depth is not available;**

**b) Terrain or soil conditions exist which make other locations on the lot more suitable;**

**c) The proposal is for an addition to an existing building or replacement of a leaching bed where the setback is not further reduced;**
**d) Redevelopment is proposed on an existing lot and a net improve-**

ment is achieved; or

e) The lot is located within an urban centre or community and a net improvement over the existing situation is achieved.

Low Sensitivity Waterbodies

F.24 Area Municipalities are encouraged to require site plan approval or a development permit for substantial development on lots abutting low sensitivity waterbodies. In addition, Area Municipalities are encouraged to require site plan approval or a development permit for all shoreline and non-shoreline commercial, institutional and industrial development in order to ensure that stormwater management and construction mitigation techniques are implemented.

Moderate and High Sensitivity and Over Threshold Waterbodies

F.25 In order to ensure no negative impact on recreational water quality, all substantial development on a lot within the waterfront designation (including backlots), and on shoreline lots in the urban centre and community designations, of moderate and high sensitivity and Over Threshold waterbodies will be subject to site plan control or development permitting.

Site Plan Control and Development Permits

F.26 Where site plan control or a development permit is required, or where on-site phosphorus management is required, the following matters will be addressed:

a) appropriate location of buildings, structures and sewage disposal systems;

b) retention or restoration of a natural vegetative buffer in accordance with Section F.21 to prevent erosion, siltation and nutrient migration;

c) maintenance or establishment of native tree cover and vegetation on the lot wherever possible;

d) appropriate location and construction of roads, driveways and pathways, including use of permeable materials; and

e) implementation of stormwater management and construction mitigation techniques, including proper re-contouring, discharging of roof leaders, use of soak away pits and other measures to promote infiltration.

Public Lands

F.27 The release of Crown land, other than lands under water, for private development is discouraged, particularly in the Waterfront designation. Should the Province dispose of Crown land for private development, such land will not be further divided unless it is to alleviate problems associated with existing development and no more than one single family dwelling will be permitted on those lands as of right.

F.28 The maintenance, enhancement or restoration of native vegetative buffers along shorelines in municipal parks and other municipal lands is strongly encouraged.

***High Sensitivity Waterbodies*** – *Specific Policy*

**Lot Creation**

F.29 In general, no lot creation will be permitted on waterbodies identified as being of high sensitivity unless the lot is connected to municipal water and sewer services.

F.30 Notwithstanding Section F.29, lot creation on private services may be permitted where the Area Municipality has the resources and tools in place and is prepared to implement the following requirements:

a) The lot creation may only proceed where a water quality impact assessment, undertaken and implemented to the satisfaction of Muskoka and the Area Municipality demonstrates that the development can proceed without negatively impacting water quality and which outlines the circumstances under which development should occur.

b) The water quality impact assessment shall consist of the following main elements at a minimum.

*Phase 1*

Site condition analysis to determine if the required conditions exist on site so that development can occur in a manner that will ensure the protection of water quality and shall include analysis of the site and surrounding area, soil characteristics, and vegetation cover. The Phase 1 report must be completed to the satisfaction of the District of Muskoka and the Area Municipality before proceeding to Phase 2.

*Phase 2*

i. Identification of recommended building and septic system (including the leaching bed) envelope and mitigation measures, including but not limited to, detailed construction mitigation plans, shoreline setbacks and buffers, measures for protecting natural vegetation, and stormwater management;

ii. Monitoring will be required to confirm that the vegetative buffer and stormwater mitigation measures are in place until such time as construction is complete and an occupancy permit is issued and at a time approximately two years following the issuance of an occupancy permit;

iii. **The use of a septic system with soils that have a demonstrated ability to effectively eliminate phosphorus will be required; and**

iv. **The recommendations of such a report and the monitoring and septic system requirements are required to be implemented through an official plan or zoning amendment and in Section 51(26) (subdivision, condominium or consent) and site plan agreements or development permits.**

**A detailed terms of reference is contained in Appendix J.**

**Development of Vacant Lots on Private Services**

**F.31 Development of a vacant lot on private services will only be permitted where it is demonstrated through a Phase 2 Water Quality Impact Assessment that building and septic system envelopes, together with appropriate mitigation measures, including but not limited to, detailed construction mitigation plans, shoreline setbacks and buffers will protect water quality and where these requirements are implemented in site plan agreements.**

**F.32 The use of a septic system with soils that have a demonstrated ability to effectively eliminate phosphorus will be required.**

**Redevelopment on Private Services**

**F.33 Redevelopment on private services will only be permitted where mitigation measures are implemented in order to prevent negative impacts on water quality, including phosphorus management measures.**

**F.34 Where the setback requirements cannot be met due to insufficient lot depth or the existence of terrain or soils conditions which make other locations on the lot more suitable, or where existing buffers or stormwater management practices do not satisfy the requirements outlined in this Plan, an overall net improvement shall be achieved through onsite phosphorus management measures.**

**F.35 A net reduction of phosphorus loading to the lake will be required for commercial redevelopment.**

***Over Threshold Waterbodies*** *– Specific Policy*

**Lot Creation - General**

**F.36 In general, no lot creation will be permitted on waterbodies identified as being Over Threshold unless the lot is connected to municipal water and sewer services.**

**Lot Creation - Moderate and Low Sensitivity Waterbodies**

**F.37 Notwithstanding Section F.36, lot creation on private services may be permitted on waterbodies identified as being of moderate or**

**low sensitivity where the Area Municipality has passed a municipal site alteration and tree cutting by-law or a development permit by-law and is prepared to implement the following requirements:**

**a) An amendment to the local Official Plan will be required to implement specific development policy.**

**b) Lot creation may only proceed where a water quality impact assessment, undertaken and implemented to the satisfaction of Muskoka and the Area Municipality demonstrates that development can proceed without impacting water quality and which outlines the circumstances under which development should occur.**

**c) The water quality impact assessment shall consist of the following main elements at a minimum:**

***Phase 1***

**Site condition analysis to determine if the required conditions exist on site so that development can occur in a manner that will ensure the protection of water quality and shall include analysis of the site and surrounding area, soil characteristics, and vegetative cover. The Phase 1 report must be completed to the satisfaction of the District of Muskoka and the Area Municipality before proceeding to Phase 2.**

***Phase 2***

**i. Identification of recommended building and septic system (including the leaching bed) envelope and mitigation measures, including but not limited to, detailed construction mitigation plans, shoreline setbacks and buffers, measures for protecting natural vegetation, and stormwater management;**

**ii. Monitoring will be required to confirm that the vegetative buffer and stormwater mitigation measures are in place until such time as construction is complete and an occupancy permit is issued, and on an annual basis until such time as the waterbody is no longer considered to be Over Threshold;**

**iii. The use of a septic system with soils that have a demonstrated ability to effectively eliminate phosphorus; and**

**iv. The recommendations of such a report and the monitoring and septic system requirements will be implemented through the official plan amendment and in the zoning amendment and Section 51(26) (subdivision, condominium or consent) agreements and site plan agreements or development permits.**

**A detailed terms of reference is contained in Appendix J.**

Development of Vacant Lots on Private Services

F.38 Development of a vacant lot on private services will only be permitted where it is demonstrated through a Phase 2 Water Quality Impact Assessment that building and septic system envelopes, together with appropriate mitigation measures, including but not limited to, detailed construction mitigation plans, shoreline setbacks and buffers will protect water quality and where these requirements will be implemented in site plan agreements.

F.39 The use of a septic system with soils that have a demonstrated ability to effectively eliminate phosphorus will be required.

Redevelopment on Private Services

F.40 Redevelopment on private services will only be permitted where phosphorus mitigation measures are implemented in order to prevent negative impacts on water quality, including measures such as setbacks, vegetative buffers and stormwater management.

F.41 Where the setback requirements cannot be met due to insufficient lot depth or the existence of terrain or soils conditions which make other locations on the lot more suitable, or where existing buffers or stormwater management practices do not satisfy the requirements outlined in this Plan, an overall net improvement should be achieved through the use of phosphorus management techniques.

F.42 A net reduction of phosphorus loading to the waterbody will be required for commercial redevelopment.

F.43 Site plan agreements or development permits will be required to implement buffers, stormwater and phosphorus management and building and septic system (including the leaching bed) envelopes.

*Section F.25—F. 96 renumbered per OPA #32*

Acidic Deposition

*Reduce Acid Rain*

F.44 The District encourages efforts to reduce acidic deposition. To this end, the District:

a) supports the continuation of research programs undertaken by the Provincial Government, the Federal Government, universities and the private sector.

b) supports the continuation of the lake monitoring programs undertaken by agencies within Muskoka.

*Emission Control Standards*

F.45 Industries shall be required to comply with all emission control standards as established by Federal and Provincial agencies from time to time.

June 7, 2007

## SCHEDULE 'F'
## Lake Classification by Phosphorus Sensitivity

| Lake Name | Area Municipality | Sensitivity | Lake Name | Area Municipality | Sensitivity |
|---|---|---|---|---|---|
| Ada | ML | Moderate | Buchanan | HT | High |
| Adams | GB | Moderate | Buchanan | LOB | Moderate |
| Allen Lakes | LOB | Moderate | Buck | GB | Moderate |
| Angel | LOB | Moderate | Buck | HT | Moderate |
| Arrowhead | HT | Low | Buck | LOB | Moderate |
| Atkins | BB | Low | Buck (Ryde) | GR | Moderate |
| Axe | HT | Moderate | Buckhorn | GB | Moderate |
| Axle | LOB | Moderate | Burns | LOB | Moderate |
| Barkway | GR | Moderate | Butterfly | ML | Low |
| Barnes | ML | Moderate | Cabin | GR | Moderate |
| Barron's | GB | Moderate | Camel | ML | Moderate |
| Bass | GR | Moderate | Camp | LOB | Moderate |
| Bass | ML | Moderate | Campstool | LOB | Moderate |
| Bastedo | ML | Moderate | Carcass | LOB | Moderate |
| Baxter | GB | Moderate | Cardwell | ML | Moderate |
| Bear | GB | Moderate | Cassidy | ML | Moderate |
| Bear | ML | Moderate | Chain | HT | Moderate |
| Bearpaw | ML | Moderate | Chub | HT | Moderate |
| Beaton | ML | Moderate | Chub | LOB | Moderate |
| Beatty | HT | Moderate | Circular | LOB | Moderate |
| Bella | LOB | Moderate | Clark | HT | Low |
| Ben | GR | Low | Clarke Pond | ML | Moderate |
| Benson | LOB | Moderate | Clear | BB | Moderate |
| Big Hoover | LOB | Moderate | Clear (Torrance) | ML | Moderate |
| Big Orillia | BB | Moderate | Clearwater | GR | Moderate |
| Big Otter | ML | Moderate | Clearwater | HT | Moderate |
| Big Stephen | LOB | Moderate | Coldwater (Swan) | GB | Moderate |
| Bigwind | BB | Moderate | Concession | ML | Moderate |
| Bing | HT | Moderate | Cooper | LOB | Low |
| Bird | BB | Moderate | Cornall | GR | Moderate |
| Black (Black R) | LOB | Moderate | Cotter | LOB | Moderate |
| Black (Muskoka L) | ML | Low | Cowan | ML | Moderate |
| Black (Muskoka R) | LOB | Moderate | Cream | LOB | Moderate |
| Blue Chalk | LOB | Moderate | Crosson | BB | Moderate |
| Bogart | ML | Moderate | Crotch | LOB | Moderate |
| Boleau | ML | Moderate | Dan | LOB | High |
| Bonnie | BB | Moderate | Dark | ML | Moderate |
| Brandy | ML | Moderate | Davies | GB | Moderate |
| Brooks | LOB | Moderate | Deer | GR | High |
| Brophy | GB | Moderate | Devine | HT | Moderate |
| Brotherson's | ML | Moderate | Dickie | LOB | Low |
| Bruce | ML | Moderate | Dividing | HT | Moderate |

June 7, 2007

| Lake Name | Area Municipality | Sensitivity | Lake Name | Area Municipality | Sensitivity |
|---|---|---|---|---|---|
| Docker | LOB | Moderate | Gray | GB | Moderate |
| Doe (Doeskin) | GR | Moderate | Greenish | LOB | Moderate |
| Dotty | LOB | Low | Grindstone | LOB | Moderate |
| Doughnut | LOB | Moderate | Grouse | LOB | Moderate |
| Duffy | ML | Moderate | Groves | HT | Moderate |
| Dunn | HT | Moderate | Gull | GR | Moderate |
| Eagle | GB | Moderate | Gullfeather | BB | Moderate |
| East Buck | BB | Moderate | Gullwing | ML | Moderate |
| Eastell | LOB | Moderate | Haggart | GB/ML | Low |
| Echo | LOB | Moderate | Halfway | BB | Moderate |
| Echo | ML | Moderate | Hardup | LOB | Moderate |
| Eighteen Mile | LOB | Moderate | Hardy | ML | Moderate |
| Ellis | LOB | Moderate | Harp | HT | Moderate |
| Ennis | BB | Moderate | Hart (Wood S.) | ML | Moderate |
| Fairy | HT | Moderate | Harts (Medora-Wood) | ML | Moderate |
| Fawn | BB/HT | Moderate | Healey | BB | Moderate |
| Fawn | HT | Moderate | Heck | LOB | Moderate |
| Fawn (Lowe) | BB/GR | Moderate | Heeney | LOB | Low |
| Fifteen Mile | LOB | Moderate | Helve | LOB | High |
| Fitzell | LOB | Moderate | Henderson | HT | Moderate |
| Flatrock | GB | Moderate | Henshaw | ML | Moderate |
| Fleming | HT | Moderate | Hesner's | ML | Moderate |
| Fleming | LOB | Moderate | High | ML | Moderate |
| Flossie | LOB | Moderate | Hillman | ML | Moderate |
| Fly | LOB | Moderate | Hoc Roc River | GR | Moderate |
| Foote | LOB | Moderate | Horse | LOB | Moderate |
| Fowler | LOB | High | Indian River | ML | Moderate |
| Fox | HT | Moderate | Insula | LOB | Moderate |
| Gagnon | ML | Moderate | Irvine | GB | Moderate |
| Galla | GB | Low | Jerry | LOB | Moderate |
| Gartersnake | GR | Moderate | Jessop | HT | Moderate |
| Gibson | GB | Moderate | Jevins | GR | Moderate |
| Gibson R. | GB | Moderate | Jill | LOB | Moderate |
| Gilleach | BB | Moderate | Joseph - *Cox Bay* | ML | Moderate |
| Gloucester Pool | GB | Moderate | Joseph - *Little Lake Joe* | ML | Moderate |
| Go Home | GB | Moderate | Joseph - *Main* | ML | High |
| Go Home R. | GB | Moderate | Joseph River | ML | Moderate |
| Golden City | HT | Low | Juniper | ML | Moderate |
| Goldstein | GB | High | Kahshe | GR | Moderate |
| Goodman | LOB | Moderate | Kenney | GB | Moderate |
| Gooley | GB | Moderate | Keyhole | BB | Moderate |
| Gosling | LOB | Moderate | Lafarce | GB | Moderate |
| Grandview | LOB | Moderate | Lake of Bays - *Haystack Bay* | LOB | Moderate |

| Lake Name | Area Municipality | Sensitivity | Lake Name | Area Municipality | Sensitivity |
|---|---|---|---|---|---|
| Lake of Bays - *Main & Dwight* | LOB/HT | Moderate | Lower Twin | LOB | Moderate |
| Lake of Bays - *Rat Bay* | LOB | Moderate | Lunnen | GB | High |
| Lake of Bays - *S. Muskoka River Bay* | LOB | Moderate | Lynch | HT | Moderate |
| Lake of Bays - *S. Portage Bay* | LOB | Moderate | Lynx | HT | Moderate |
| Lake of Bays - *Ten Mile Bay* | LOB | Moderate | Mainhood | HT | Moderate |
| Lake of Bays - *Trading Bay* | LOB | Moderate | Mansell | LOB | High |
| Lamberts | HT | Moderate | Margaret | LOB | Moderate |
| Lamorie | GR | Moderate | Marion | LOB | Moderate |
| Lancelot | HT | Moderate | Martin | LOB | Moderate |
| Lassetter | LOB | Moderate | Mary | HT | Moderate |
| Leclaric | GB | Moderate | Mary Jane | ML | Moderate |
| Lee | LOB | Moderate | Mathews | HT | Moderate |
| Leech | BB | Moderate | McReynolds | LOB | Moderate |
| Lena | HT | Moderate | McCan | ML | Moderate |
| Leonard | ML | Moderate | McCrea | GB | Moderate |
| Lili | ML | High | McDonald | GB | Moderate |
| Little | GB | Moderate | McEwen | LOB | Moderate |
| Little | GR | Moderate | McKay | BB | Moderate |
| Little Arrowhead | HT | Moderate | McMaster | GB | Moderate |
| Little Clear | LOB | Moderate | McRey | BB | Moderate |
| Little Go Home Bay | GB | Moderate | Medora | ML | Moderate |
| Little Hellangone | GB | Moderate | Menominee | HT/LOB | Low |
| Little Nelson | LOB | Moderate | Mink | LOB | Moderate |
| Little Orillia | BB | Moderate | Mirror | ML | Moderate |
| Little Otter | BB | Moderate | Montgomery | HT | Moderate |
| Little Oxbow | LOB | Moderate | Moon R. | ML/GB | Moderate |
| Little Pell | LOB | Moderate | Moose | GR | Moderate |
| Little Spaniel | LOB | Moderate | Moot | LOB | Moderate |
| Little Sunny | GR | Moderate | Morrison | GR | Moderate |
| Lone | GB | Moderate | Mosquito | ML | Moderate |
| Long | ML | High | Mug | LOB | Moderate |
| Longline | LOB | Moderate | Muskoka - *Bala Bay* | ML | Moderate |
| Long's (Utterson) | HT | Moderate | Muskoka - *Dudley Bay* | ML | Moderate |
| Loon | GR | Moderate | Muskoka - *Main* | ML | Moderate |
| Loon | LOB | Moderate | Muskoka - *Muskoka Bay* | GR | Moderate |
| Lower Eagle | GB | Moderate | Muskoka - *Whiteside Bay* | ML | Low |
| Lower Galla | GB | Moderate | Muskoka River | HT/BB | Moderate |
| Lower Raft | LOB | Moderate | Musquash River | GB | Moderate |
| Lower Schufelt | LOB | High | Myers | GB | High |

June 7, 2007

| Lake Name | Area Municipality | Sensitivity | Lake Name | Area Municipality | Sensitivity |
|---|---|---|---|---|---|
| Narrow | ML | Moderate | Rosseau – *Main* | ML | Moderate |
| Neilson | ML | Moderate | Rosseau - *Portage Bay* | ML | High |
| Nelson | LOB | Moderate | Rosseau - *Skeleton Bay* | ML | Moderate |
| Nine Mile | ML | Moderate | Roundabout | LOB | Moderate |
| North Bay | GB | Low | Rutter | ML | Moderate |
| North Dotty | LOB | Moderate | Sage | LOB | Moderate |
| North Healey | BB | Moderate | Sahanatien | GB | Moderate |
| North Muldrew | GR | Moderate | Saucer | LOB | Moderate |
| Nutt | ML | Moderate | Saw | BB | Moderate |
| Onawan | HT | Moderate | Sawyer | ML | Moderate |
| Otter | HT | Moderate | Schufelt | LOB | Moderate |
| Oudaze | HT | Moderate | Seventeen Mile (Dwight) | LOB | Moderate |
| Oxbow | LOB | Moderate | Seventeen Mile (Vernon) | LOB | Moderate |
| Pairo (Twin) 1 | LOB | Moderate | Severn River | GR/ML/ GB | Moderate |
| Pairo 2 | LOB | Moderate | Shack | BB | High |
| Palette | HT | Moderate | Shapter | LOB | Moderate |
| Palmer | HT | Moderate | Shaw | ML | Moderate |
| Paul's (Reay) | BB | Moderate | Shoe | LOB | Moderate |
| Peeler | LOB | Moderate | Siding | HT | Low |
| Pell | LOB | Moderate | Silver | GR | Moderate |
| Penfold | HT | Moderate | Silver | ML | High |
| Peninsula | HT/LOB | Moderate | Silversands | GB | Moderate |
| Pennsylvania | ML | Moderate | Sims | HT | Moderate |
| Perch | HT | Moderate | Six Mile - *Cedar Nook Bay* | GB | Moderate |
| Pigeon | GR | Moderate | Six Mile - *Channel* | GB | Moderate |
| Pine | BB | Moderate | Six Mile – *Main* | GB | Moderate |
| Pine | GR | Moderate | Six Mile - *Prov. Park Bay* | GB | High |
| Porcupine | LOB | Moderate | Sixteen Mile | LOB | Moderate |
| Pretzel | LOB | Moderate | Skeleton | HT/ML | Moderate |
| Prospect | BB | Moderate | Slim | LOB | Moderate |
| Pup | LOB | Moderate | Slocombe | HT | Moderate |
| Rat (Cana) | GR | Moderate | Sly | LOB | Moderate |
| Rebecca | LOB | Low | Solitaire | LOB | Moderate |
| Red Chalk | LOB | Moderate | South Bay | GB | Moderate |
| Ricketts | ML | Low | South Muldrew | GR | Moderate |
| Ridout | LOB | High | South Nelson | LOB | Moderate |
| Ril | LOB | Moderate | South Tasso | LOB | High |
| Riley | GR | Moderate | Spaniel | LOB | Moderate |
| Ripple | HT | Moderate | Sparrow | GR | Moderate |
| Roderick | ML | Moderate | Spence | BB | Moderate |
| Rose | HT | Moderate | Spider | HT | Moderate |
| Rosseau - *Brackenrig Bay* | ML | Moderate | Splatter | LOB | Moderate |

June 7, 2007

| Lake Name | Area Municipality | Sensitivity | Lake Name | Area Municipality | Sensitivity |
|---|---|---|---|---|---|
| Spring | BB | Moderate | Turtle (Long Turtle) | GR | Moderate |
| St. Germaine | ML | Moderate | Twelve Mile Bay | GB | Moderate |
| St. Mary (Paint) | LOB | Moderate | Twin Lakes | GB | Moderate |
| Steeple | LOB | Moderate | Upper Eagle | GR | Moderate |
| Stewart | GB/ML | Moderate | Upper Oxbow | LOB | Moderate |
| Stinking | HT | Moderate | Upper Raft | LOB | Moderate |
| Stoneleigh | BB | Moderate | Upper Twin | HT | Moderate |
| Stuart | GB | Moderate | Verner | LOB | Moderate |
| Sugar Bowl | LOB | Moderate | Vernon | HT | Moderate |
| Sunny | GR | Moderate | Walker | LOB | Moderate |
| Surerus | GB | Moderate | Waseosa | HT | Moderate |
| Surprise | LOB | Moderate | Webster | GB | Moderate |
| Tackaberry | LOB | Moderate | Weeduck | HT | Moderate |
| Tadenac | GB | Moderate | Weismuller | BB | Moderate |
| Tadenac Bay | GB | Moderate | Wells | LOB | Moderate |
| Tar | ML | Moderate | West Buck | BB | Moderate |
| Tasso | LOB | Moderate | White | GB | High |
| Teapot | LOB | Moderate | Whitehouse | LOB | Moderate |
| Thinn (Reay) | BB/GR | Low | Wier | ML | Moderate |
| Three Island | LOB | Moderate | Wildcat | LOB | Moderate |
| Three Mile | GR | Moderate | Wilson | LOB | Moderate |
| Three Mile | ML | Moderate | Wolfkin | LOB | Moderate |
| Toad | LOB | Moderate | Wood | BB | Moderate |
| Tock (Otter) | HT | High | Woodbine | BB | Moderate |
| Tom | LOB | Moderate | Woodland | ML | Moderate |
| Toms | HT | Moderate | Woods | ML | Moderate |
| Tongua | HT | Moderate | Wrist | BB | Moderate |
| Tooke | LOB | Moderate | Young | ML | Moderate |
| Toronto | GB | Low | | | |
| Trackler | HT | High | | | |
| Tucker | HT | High | | | |
| Turtle | ML | Moderate | | | |

LEGEND

BB – Bracebridge
GB – Georgian Bay
GR – Gravenhurst
HT – Huntsville
LOB – Lake of Bays
ML – Muskoka Lakes

APPENDIX 'J'
LAKE SYSTEM HEALTH
TERMS OF REFERENCE
WATER QUALITY IMPACT ASSESSMENTS
June 7, 2007

Water Quality Impact Assessments will be carried out by a professional who can be qualified by the Ontario Municipal Board as an expert witness on these matters, if required, on the basis of education and experience in one or more of the following disciplines: soils science, hydrogeology, or limnology and with demonstrated experience working in Precambrian Shield environments. Water Quality Impact Assessments consist of three main steps. Firstly, a site condition analysis is required. Should this analysis determine that site conditions exist such that development can proceed without affecting water quality, the second step would involve the identification of a suitable building envelope and any required mitigation measures. As a third step, the final report will be reviewed by municipal staff and may also be subjected to a peer review.

***Phase 1: Site Condition Analysis***

A site condition analysis will be undertaken to determine if the required conditions exist on site so that development can occur in a manner that will ensure the protection of water quality. This analysis will include:

***a. Site and Surrounding Area***

A plan will be provided that identifies the physical features associated with the site and surrounding lands including land use, topographic features, watercourses, ponds, designated protected areas, and wetlands.

***b. Site Description***

A Plan will be provided showing a detailed description of the site including:

- Lot size including frontage, depth, area and general shape.
- Location of public and private access roads.
- Location of significant features, both geological and man-made, including such features as wetlands, off-site streams and other surface water.
- Site contours at an interval not more than 5 metres (OBM).
- Areas of slope between 0 to 9%; 10 to 25%; and over 25%.
- The location of all depressions and gullies that will channel stormwater toward the lake.
- The location of all permanent and seasonal or intermittent streams as well as details concerning observations of the amount of flows experienced within the streams at various times of year (minimum of spring freshet and summer drought periods) and an outline of the expected path of surface runoff from the development site to the lake of interest.
- Areas of aquatic vegetation and ecological description (dominant species, emergent/submergent/floating leaved).
- A description of the terrestrial vegetation community – size, composition, age and general health, as detailed below.

***c. Soil Characteristics***

The Impact Assessment will include a documentation and mapping of soil conditions in order to characterize the soils to be used in the construction of septic system leaching beds as well as the native soils in the mantle between the leaching beds and any surface water receptors.

The location of the proposed septic system leaching bed and the expected pathway of the subsurface nutrient flow (septic plume) in relation to the ultimate receptor (waterbody) of the nutrient flow must be delineated on the Plan submitted. The proponent will also:

- Undertake manual auguring to map soil depth along the flow path of each septic plume within 30m of the tile field, with soil depths inferred from a minimum of twenty (20) points, or as many as required to ensure the integrity of the soil mantle.
- Document the location of sources of suitable soil to construct the partially or totally raised tile fields.
- Provide descriptions of soil characteristics –type, texture and colour for any soils (native or off site) used to construct the tile field and present in the mantle, as determined from soil profiles taken at the site of the tile field or source of the soil, as appropriate, and the mantle area.
- For lakes which are either highly sensitive or over the water quality threshold - provide an analysis of soil chemistry (lab analyses of phosphorus adsorption capability, mineral content and particle size) for any soils proposed for use in the tile field, and from the native soil mantle.
- Map the location of all on-site sample locations, and off-site locations of soils that are to be imported.

### d. *Vegetation cover*

The Impact Assessment will map the location and characteristics of shoreline and upland vegetation communities and provide an explanation of the site characteristics that will provide natural buffer protection for the adjacent waterbody from overland and subsurface flow of sediment, nutrient and other potential pollutants. The Impact Assessment will include a photographic documentation of the property showing vegetative cover. The record shall include the following photographs, at a minimum:

- The shoreline across the entire width of the lot as viewed from the lake,
- The tile field and mantle areas, along the direction of subsurface flow towards the lake
- The building envelope, along the shortest distance between the envelope and the lake

### e. *Findings*

A determination of the suitability of the site conditions to ensure development will not adversely impact water quality will be provided.

## *Phase 2: Identification of Recommended Building and Septic Envelope and Mitigation Measures*

Where a site has been determined to have the conditions required to permit development based on the findings of the Site Condition Analysis, a Plan will be provided showing a detailed description of the manner in which development should occur to protect water quality, including:

- Building location, septic system location, paths, decks, accessory buildings, shoreline structures, parking areas and any other hard surfaces;
- Proximity to significant features, both geological and man-made, including such features as wetlands, off-site streams and other surface water.

- The location of proposed leaching beds in relation to permanent and intermittent streams or other drainage courses.

Specific mitigation measures necessary for the effective elimination of the impacts of nutrient and sediment loading on water quality should also be identified, including:

- Detailed construction mitigation plans including methods to deal with sediment and nutrient loading. Map the proposed location of all proposed facilities.
- Detail and map stormwater mitigation measures including methods to deal with sediments and nutrient loading during construction and occupation.
- The location, design and construction of septic systems and leaching beds.
- Shoreline setbacks and buffer areas.
- The delineation of building envelopes for proposed building structures and uses, including septic systems for each lot. Building envelopes are defined as the area bounded by the minimum setback from the shoreline and minimum yard setbacks for all development.
- Measures for protecting the natural vegetation, slopes and soil mantle for the area located outside of the building envelopes. Design criteria (including size and construction materials) for uses, buildings and structures that may be permitted within this area. (e.g. boat docks, meandering walkways to the shoreline, and driveways).

***Step 3: Municipal Review***

The District of Muskoka and/or the Local Municipality will review the Impact Assessment, or submit it to peer review to establish:

- The completeness of the assessment regarding the requirements herein,
- Interpretation of the assessment by the proponent,
- The effectiveness of the mitigation measures proposed
- The likelihood that the assessment supports a conclusion of no nutrient impact to the subject water body.

The assessment will be maintained on file for the possibility of re-assessment of the site to ensure that mitigation measures have been implemented and maintained over time.

June 7, 2007

## APPENDIX K
## "Over Threshold" Lakes for Recreational Water Quality

| Lake Name | Area Municipality |
|---|---|
| Baxter | Georgian Bay |
| Bird | Bracebridge |
| Brooks | Lake of Bays |
| Clear (Torrance) | Muskoka Lakes |
| Clearwater | Gravenhurst |
| Clearwater | Huntsville |
| Dark | Muskoka Lakes |
| Flatrock | Georgian Bay |
| Go Home | Georgian Bay |
| Gull | Gravenhurst |
| High | Muskoka Lakes |
| Joseph – *Cox Bay* | Muskoka Lakes |
| Leech | Bracebridge |
| Leonard | Muskoka Lakes |
| Long | Muskoka Lakes |
| Longline | Lake of Bays |
| Long's (Utterson) | Huntsville |
| Loon | Gravenhurst |
| McKay | Bracebridge |
| Medora | Muskoka Lakes |
| Mirror | Muskoka Lakes |
| Muskoka - *Muskoka Bay* | Gravenhurst |
| Myers | Georgian Bay |
| Nutt | Muskoka Lakes |
| Pine | Bracebridge |
| Ril | Lake of Bays |
| Rosseau - *Brackenrig Bay* | Muskoka Lakes |
| Rosseau – *Portage Bay* | Muskoka Lakes |
| Rutter | Muskoka Lakes |
| Silver | Muskoka Lakes |
| Six Mile - *Cedar Nook Bay* | Georgian Bay |
| Six Mile - *Prov. Park Bay* | Georgian Bay |
| South Bay | Georgian Bay |
| Spring | Bracebridge |
| Stewart | Georgian Bay / Muskoka Lakes |
| Three Mile | Muskoka Lakes |
| Tooke | Lake of Bays |
| Turtle (Long Turtle) | Gravenhurst |
| Walker | Lake of Bays |
| Waseosa | Huntsville |
| Wood | Bracebridge |

## ATTACHMENT A – GROUND-WATER CONTAMINATION FROM TWO SMALL SEPTIC SYSTEMS ON SAND AQUIFERS

# Ground-Water Contamination from Two Small Septic Systems on Sand Aquifers

by W. D. Robertson, J. A. Cherry, and E. A. Sudicky[a]

## Abstract

Distinct plumes of septic system-impacted ground water at two single-family homes located on shallow unconfined sand aquifers in Ontario showed elevated levels of $Cl^-$, $NO_3^-$, $Na^+$, $Ca^{2+}$, $K^+$, alkalinity, and dissolved organic carbon and depressed levels of pH and dissolved oxygen. At the Cambridge site, in use 12 years, the plume had sharp lateral and vertical boundaries and was more than 130 m in length with a uniform width of about 10 m. As a result of low transverse dispersion in the aquifer, mobile plume solutes such as $NO_3^-$ and $Na^+$ occurred at more than 50 percent of the source concentrations 130 m downgradient from the septic system. At the Muskoka site, in use three years, the plume also had discrete boundaries reflecting low transverse dispersion. After 1.5 years of system operation, the Muskoka plume began discharging to a river located 20 m from the tile field. Almost complete $NO_3^-$ attenuation was observed within the last 2 m of the plume flowpath before discharge to the river. This was attributed to denitrification occurring within organic matter-enriched riverbed sediments.

The very weakly dispersive nature of the two aquifers was consistent with the results of recently reported natural-gradient tracer tests in sands. Therefore, for many unconfined sand aquifers, the minimum distance-to-well regulations for permitting septic systems in most parts of North America should not be expected to be adequately protective of well-water quality in situations where mobile contaminants such as $NO_3^-$ are not attenuated by chemical or microbiological processes.

## Introduction

About one-third of the population of the United States uses septic systems for waste-water disposal (U.S. EPA, 1986). Septic systems thus represent the largest volumetric source of effluent discharged to the ground-water zone. The literature, however, has few detailed field evaluations of septic system impacts on ground water (Childs et al., 1974; Walker et al., 1973b; Rea and Upchurch, 1980; Barber et al., 1988). Septic systems located on sand and gravel aquifers are a potential source for producing large-scale contaminant plumes in aquifers that are also likely to be used for drinking-water supply. Additional field evaluations are appropriate at this time for two reasons. First, recent studies have indicated that the dispersive capabilities, and therefore the contaminant dilution potential of many sand and gravel aquifers, are much less than previously thought (Sudicky et al., 1983; Freyberg, 1986; Garabedian, 1987; Moltyaner and Killey, 1988a, b). This is significant because regulators frequently rely on dilution for attenuation of septic system contamination in ground water. Second, there has been recent concern regarding persistence of toxic trace organic constituents in the ground-water zone. Such contaminants frequently occur in sewage effluent (Viraraghavan and Hashem, 1986; Barber et al., 1988).

In this study, exceptionally detailed ground-water monitoring networks were used to investigate ground-water impacts caused by septic systems at two single-family homes located on shallow unconfined sand aquifers in Ontario. At the older site (Cambridge), where the septic system has been in operation since 1977 and where our field investigations began in 1987, a long narrow plume of septic system-impacted ground water, more than 130 m in length, has been identified within a carbonate-rich sand aquifer. At the younger site (Muskoka), a ground-water monitoring network was installed six months after the beginning of full-time use of the septic system in 1987, and monitoring was then carried out for a period of over two years. During that time, the plume migrated a distance of 20 m within a poorly buffered, carbonate-depleted, sand aquifer, and began discharging into an adjacent river. Both septic systems are of the conventional design used in Ontario and most other parts of North America for permeable soils. In this study the character of the plumes are described and major-ion geochemistry is evaluated along clearly defined subsurface flow paths originating from the septic system tile beds. In addition, because of the detail of the monitoring networks, we are able to infer dispersive characteristics of the aquifer at the Cambridge site and are able to evaluate the mobility of the major-ion species at the Muskoka site by monitoring plume breakthrough at downgradient locations. In subsequent papers the microbial conditions and the persistence and fate of a number of consumer product-derived organic constituents will be described in the Cambridge plume (Shimp and Lapsins, 1990; Rapaport and White, 1990; Robertson, 1990).

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## Field Sites

The Cambridge site is located near Cambridge, Ontario at an agricultural research station operated by the University of Guelph (Figure 1). The site is located on a flat-lying sand plain where glaciolacustrine and outwash sand occurs to a depth of 4-8 m and overlies a silt till of low permeability (Karrow, 1987). The surficial aquifer is comprised of moderately permeable fine sand to very permeable coarse sand with coarser material dominating. From 1977 to present, a family of two adults and two children have been permanent residents at the site. Since that time, household waste water including laundry effluent, has been discharged to a typical domestic septic system consisting of a holding tank and weeping tile bed 100 $m^2$ in area (Figure 1). The weeping tiles, consisting of perforated PVC pipe, are positioned in trenches 2 meters apart and are encased for a radius of 15 cm by gravel. The tiles lie at a depth of 0.6 m at a location where the water table is about 2 m below ground surface. The tile bed lies under a garden plot but most of the contaminant plume originating from the septic system extends under intensively cultivated agricultural land to the northeast.

The Muskoka site is located on the edge of the Muskoka River near Bracebridge, Ontario (Figure 2). At this site, fine fluvial sand occurs to a depth in excess of 10 m and overlies granitic bedrock which outcrops within 100 m of the site. From 1987 to present, a family of two adults have lived at the site. Since that time household waste water, including laundry effluent, has been discharged to a typical domestic septic system consisting of a holding tank and tile bed about 80 $m^2$ in area. The tile bed is positioned 20 m from the Muskoka River at a location where the water table is about 3 m deep. Tile bed construction is the same as at Cambridge except that the tile lines are trenched into coarse sand fill that occurs to a depth of about 1 m. The tile bed and plume underlies grass lawn, and the upgradient groundwater flow system underlies uninhabited forested terrain.

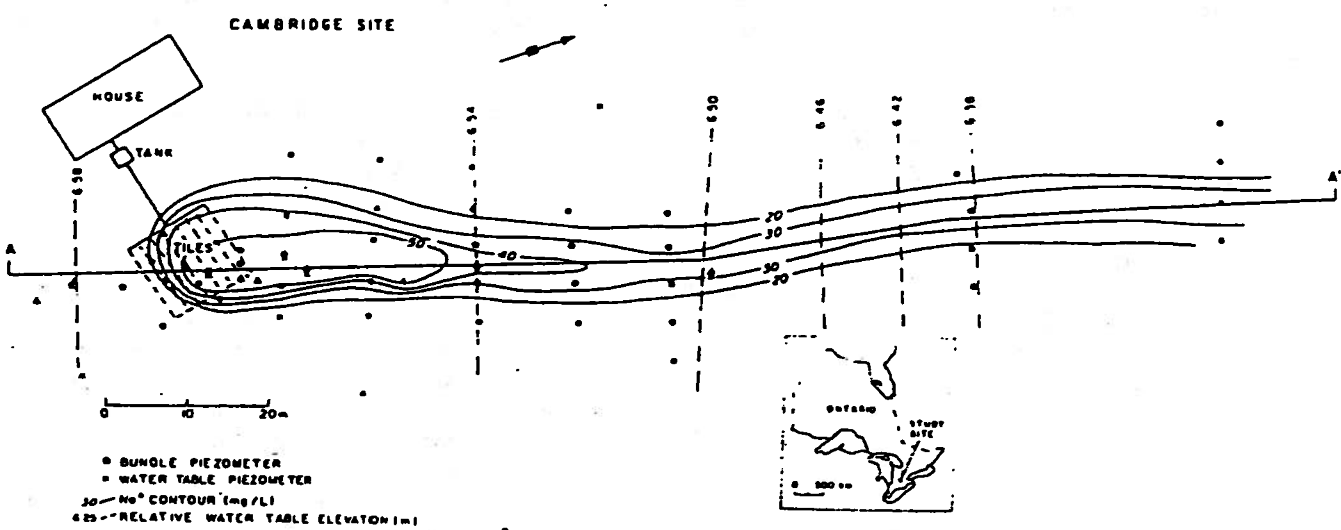



**Fig. 1. Vertically averaged ground-water $Na^+$ distribution in the surficial aquifer, Cambridge site, 1987.**

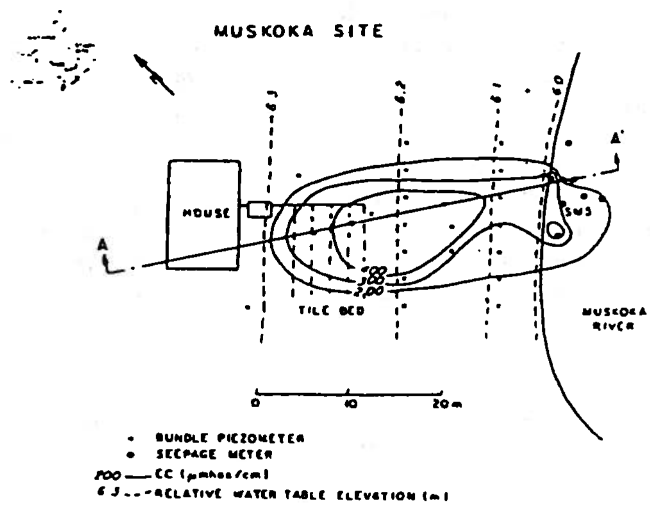



**Fig. 2. Vertically averaged ground-water electrical conductance at the Muskoka site, September 1988.**

## Methods

The initial field investigation at each site involved installation of a network of water-table piezometers to establish the regional ground-water flow direction, the installation of a multiple piezometer bundle near the tile field, and the retrieval of a continuous 5-cm-diameter sediment core from the aquifer. The water-table piezometers were constructed of 1.6-cm-diameter PVC pipe with slotted and screened tips and were installed into the shallow water-table zone using a hand auger. The multiple piezometer bundles consisted of 13 2-mm-diameter polyethylene sampling tubes attached at 30-cm-depth intervals to a center stock of 1.6-cm-diameter PVC pipe. The bundles were installed into the aquifer with the aid of 5-cm-diameter steel casing and an expendable drive tip. The casing was advanced using a handheld vibrating hammer (Atlas Copco, COBRA) and was extracted after bundle insertion. Continuous sediment cores were obtained in 5-cm-diameter thin-walled aluminum core tubes using a newly developed portable coring apparatus (Starr and Ingleton, 1989). The core tubes, each 1.5 m in length, were fastened to the end of a 5-cm-diameter steel casing. Next, a plunger-type drive point was inserted in the driving end of the core tube and was held in position with a steel center rod. The core was obtained by first advancing the coring apparatus to the desired depth using the handheld vibrating hammer. The plunger was then restricted from further movement using an anchored cable, and the core was obtained finally by advancing the core tube an additional 1.5 m. In excess of 90 percent core recovery was obtained using this technique.

After the preliminary field investigations, detailed ground-water monitoring networks were installed at each site. At Cambridge an additional 41 multiple piezometer bundles were inserted to the bottom of the aquifer at 4-6-m depth (Figure 1). At Muskoka, an additional 20 piezometer bundles were installed to depths of 3-5 m (Figure 2). Each piezometer bundle contained from 13 to 18 sampling tubes; hence the monitoring networks consisted of over 500 sampling points at Cambridge and over 250 sampling points at Muskoka.

After sampling and delineation of the plume locations in three dimensions, additional piezometer bundles were installed along the plume centerlines for the purpose of trace organics sampling. These bundles were each comprised of six Teflon tubes 0.64 cm in diameter arranged at 0.6-m-depth intervals. At Cambridge, nine such bundles were installed (piezometer bundles 33-41, Figure 3) while at Muskoka three were installed (piezometers 31-33, Figure 4).

During installat[illegible] of each Teflon piezometer bundle, a continuous core of aquifer sediment was retrieved for the purpose of aquifer solids characterization.

At Muskoka, eight riverbed seepage meters similar to the type described by Lee (1977) were installed in the area of the riverbed where the plume was discharging. These were constructed from 0.3-m-diameter plastic buckets which were inverted and pressed into the sediment. These devices were successful in collecting discharging ground water which was sampled by means of a 0.6-cm-diameter access tube using a peristaltic pump.

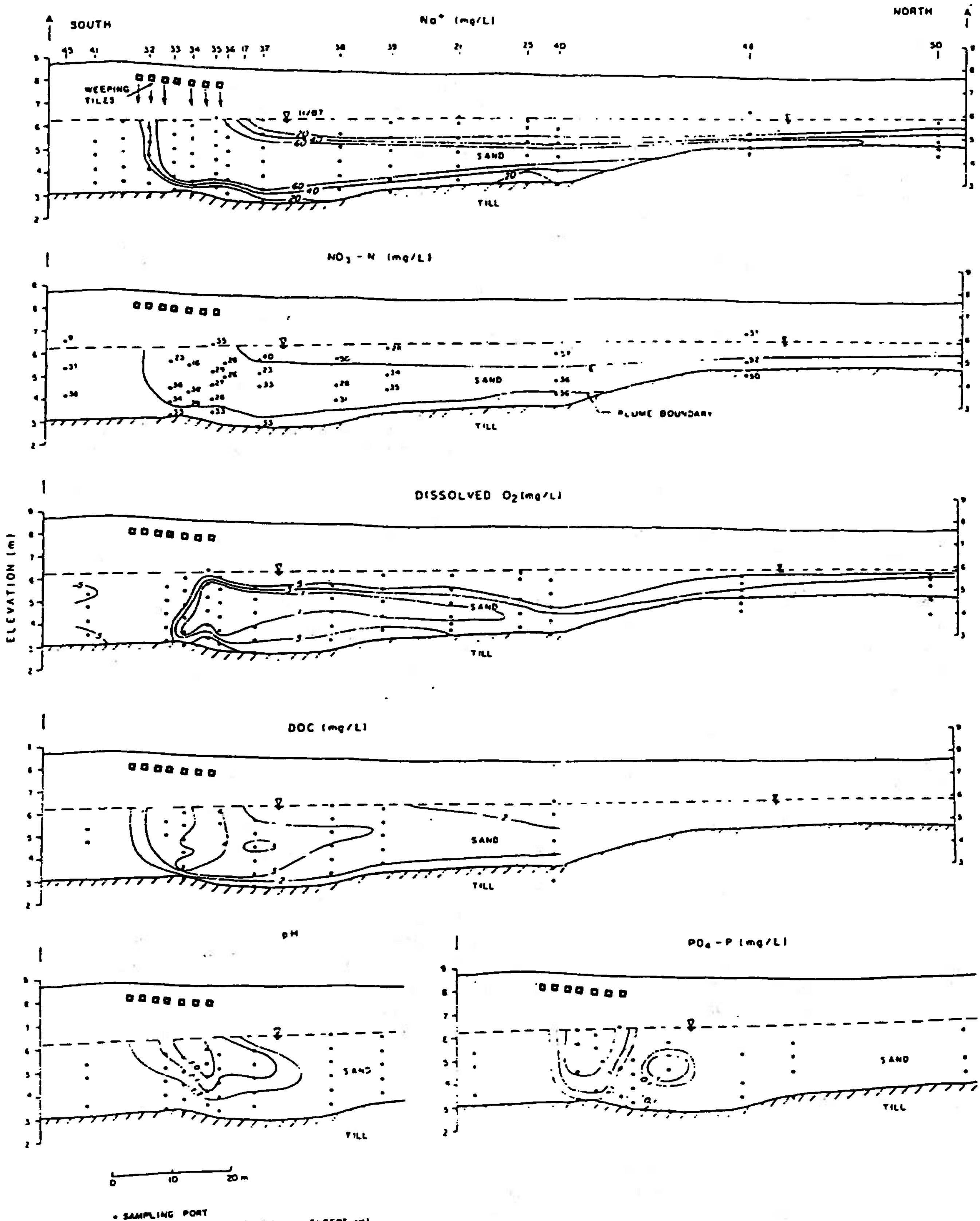
Sampling access to the septic tank effluent was provided by means of a 0.6-cm-diameter sampling tube inserted into the center weeping tile at each site, from which effluent could be withdrawn using a peristaltic pump.

Routine sampling of the monitoring networks demonstrated that the zones of septic system-impacted ground water could be distinguished readily by field measurements of ground-water electrical conductance (EC) or by laboratory measurements of sodium ($Na^+$) levels. The entire monitoring networks were thus sampled regularly for electrical conductance using a temperature-corrected field-portable meter and a multiple-port sampling manifold, and about 50 percent of the monitoring network at Cambridge was also sampled for $Na^+$ content using the multiple-port sampling manifold or a peristaltic pump. Additional sampling at both sites was then concentrated along the plume centerlines and included sample collection for analysis of all major dissolved ions ($Na^+$, $K^+$, $Ca^{2+}$, $Mg^{2+}$, $NH_4^+$, $NO_3^-$, $Cl^-$, $SO_4^{2-}$, and $PO_4^{3-}$), dissolved oxygen (DO), pH, alkalinity, and dissolved organic carbon (DOC). DO was measured in the field using the Winkler titration method (U.S. EPA, 1974) which provided a detection limit of about 0.1 mg/l. Alkalinity was measured in the field by acid titration, and pH was measured in the field using a field-portable meter. Sodium analyses were performed using an atomic absorption method which provided a lower detection limit of 1 mg/l. Nitrate and bromide analyses were done by ion chromatography which provided a detection limit of 0.2 mg/l for $NO_3^-$-N and 0.5 mg/l for $Br^-$. Other major ion analyses were done colorimetrically.

Bromide tracer tests to establish effluent residence times in the septic tank, in the unsaturated zone, and in the ground-water zone at each site were conducted by instantaneously injecting a 1 kg mass of NaBr into the septic tanks. After injection, 20-ml samples were collected for analysis of $Br^-$ content, from the tile effluent at Cambridge for a period of 11 days, and from the ground-water zone for a period of 350 days at Cambridge and 90 days at Muskoka.

Sediment grain-size distribution was determined using sieves for the sand fraction and a hydrometer for the silt and clay fractions. Total organic carbon fraction (foc) of the sediment was measured by combustion (Leco furnace) after acid leaching of the inorganic carbon. Sediment cation exchange capacity (CEC) was determined by a barium stripping technique (Peech et al., 1962). Sediment carbonate content ($CaCO_3$ equivalent) was determined by the method of Allison and Moodie (1965).

Falling head permeameter tests were conducted on repacked samples of core material to determine sediment hydraulic conductivity. Tests were done at 30-cm-depth intervals on each core using the apparatus and techniques described by Sudicky (1986).




7- 3. Major ion geochemistry along the plume centerline at the Cambridge site, 1987-88.

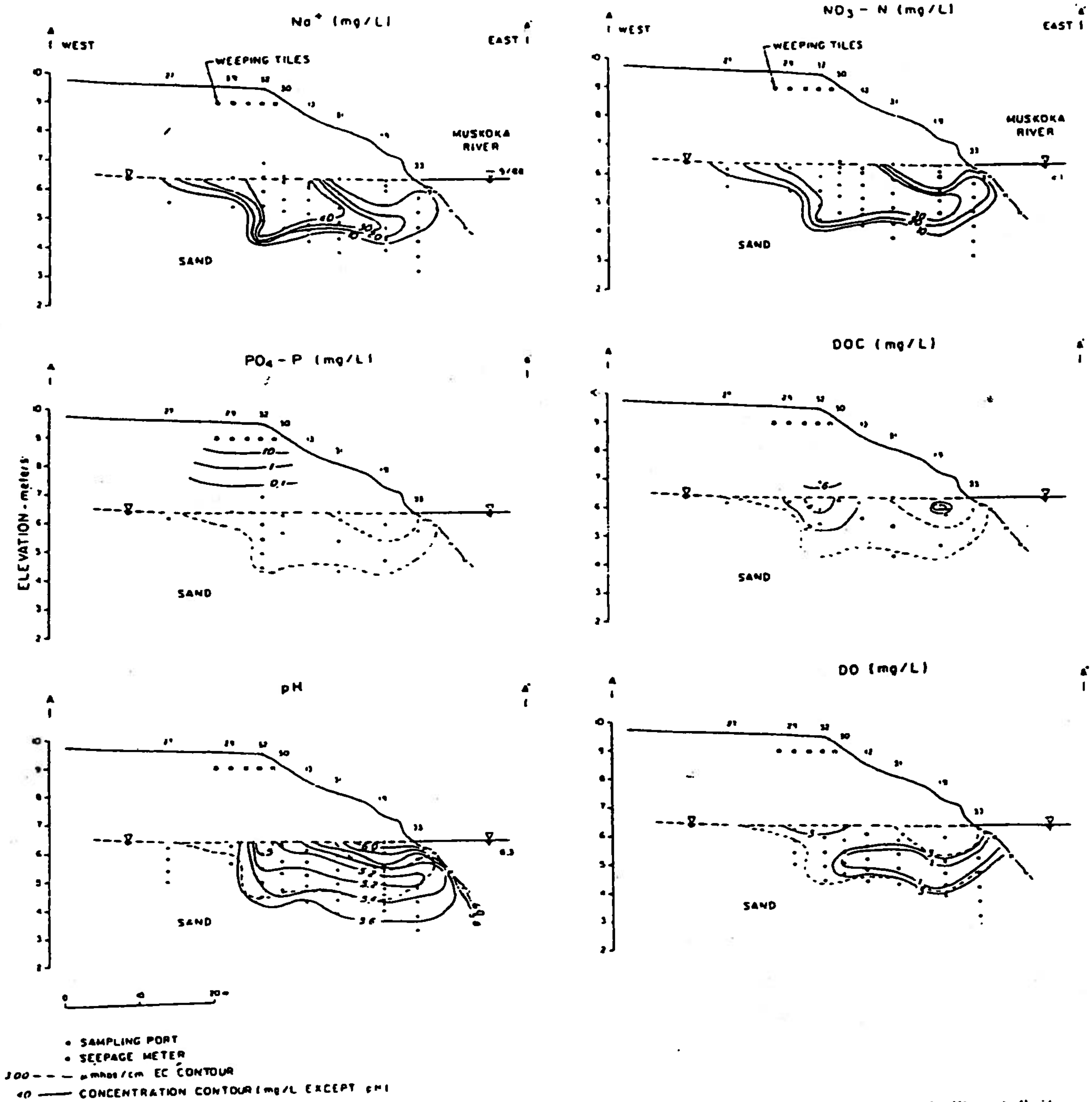



Fig. 4. Major ion geochemistry along the plume centerline at Muskoka, 500 days after the beginning of effluent discharge (September 1988).

## Results

### *Stratigraphy*

At the Cambridge site, sediment cores retrieved along section A-A' (piezometer bundle locations 33-51, Figures 1 and 3) showed that the surficial aquifer in the plume area is 4-6 m thick and overlies a slightly undulating till surface. Permeameter tests indicated that two major zones of contrasting hydraulic conductivity existed within the aquifer. In the area downgradient (north) of piezometer bundle 38, the entire thickness of the aquifer was comprised of clean medium to coarse sand (Table 1) having an average hydraulic conductivity (K) of $3 \times 10^{-2}$ cm/s (sample size N = 21). In the area upgradient of bundle 38, including the area under the tile field, most of the saturated portion of the aquifer was comprised of less permeable fine- to medium-grained sand with average K of $5 \times 10^{-3}$ cm/s (N = 22). The lower K zone was, however, underlain at all locations along the bottom of the aquifer by a high K layer of coarse sand approximately 0.3 m thick. Figure 5 shows a typical hydraulic conductivity profile for the tile bed area (core 36). The relatively impermeable underlying till unit consisted of compact pebbly sandy silt (K = $2 \times 10^{-5}$ cm/s, N = 3). The contact between the aquifer and the till unit was generally a sharp erosional feature, except in the area between cores 35 and 38, where a topographic depression in the till surface was infilled with silt.

At the Muskoka site the zone below the water table consisted entirely of homogeneous silty fine sand (K = $1.3 \times 10^{-3}$ cm/s, N = 38) to a depth in excess of 7 m below the water table. The only contrasting sediment type observed at this site was a zone of clean medium sand (K = $3 \times 10^{-2}$ cm/s, N = 4) occurring in the vadose zone below the tile bed (core 32, Figure 5) and a more silt-rich zone about 0.5 m thick on the riverbed.

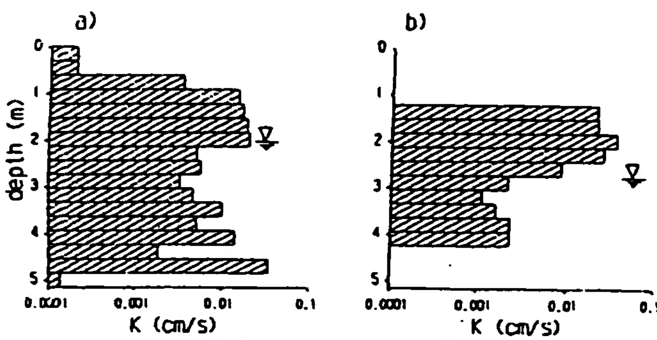



**Fig. 5. Permeameter-derived hydraulic conductivity profiles below the tile bed areas: (a) Cambridge site, core 36, and (b) Muskoka site, core 32 (K at 10°C). Core locations are shown on Figures 3 and 4.**

### *Ground-Water Flow*

At the Cambridge site, the water-table hydraulic gradient was northeasterly which was consistent with the plume-indicated direction of ground-water flow (Figure 1). The plume shape (Figure 3) demonstrated that flow within the aquifer was predominantly horizontal except beneath the tile bed where the plume had descended to near the bottom of the aquifer indicating a significant vertically downward component of flow. The Darcy equation was used to estimate the horizontal ground-water flow velocity in the downgradient area between piezometer bundles 39 and 40 (Figure 3). In this zone, the horizontal hydraulic gradient was about 0.0014 (Figure 1) and the permeameter-estimated hydraulic conductivity was about $3 \times 10^{-2}$ cm/s. Assuming an effective porosity equal to 0.35, which is reasonable for this type of sand aquifer, the calculated flow velocity was about 40 m/a. In the area of the tile bed, lower flow velocities were expected because the horizontal hydraulic gradient was less, and K was generally lower. In this area, the Darcy equation was not used to estimate velocity because the hydraulic gradient was too low to measure with the required precision. The bromide tracer test, discussed in the next section, provided an indication of flow velocities in the immediate area of the tile field.

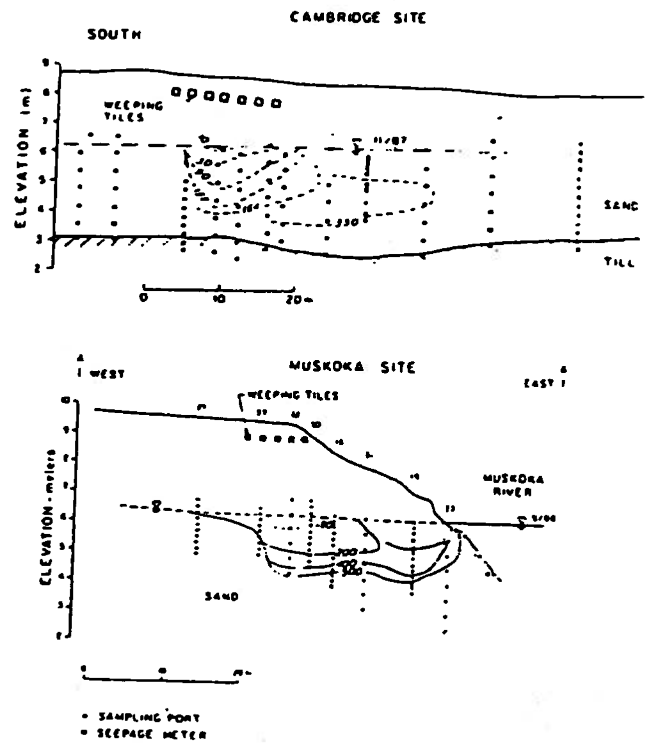



**Fig. 6. Ground-water residence time (days) along the plume centerlines, inferred from bromide tracer tests (dashed lines) and from breakthrough of 300 μmhos/cm EC front (solid lines).**

**Table 1. Aquifer Solids Properties**

| | Gravel (>2 mm) wt. % | Sand (>0.053 mm) wt. % | Silt wt. % | Clay (<0.002 mm) wt. % | Carbonate ($CaCO_3$ equiv.) wt. % | *foc* | CEC meq/100 g |
|---|---|---|---|---|---|---|---|
| Cambridge site Aquifer (N = 9) | 0 | 95 | 5 | 0 | 21 | 0.0005[1] | 5.0 |
| Muskoka site Aquifer (N = 7) Riverbed (N = 3) sediments | 0 | 87 | 13 | 0 | 0.4 | 0.0003[2] 0.017[3] | 2.2 |

[1] Sample size (N) = 7.
[2] N = 3.
[3] 0.02 to 0.3 m below riverbed at seepage meter 5 location.

### Bromide Tracer Tests

At the Cambridge site, the injected slug of NaBr resulted in maximum $Br^-$ concentrations of 360 mg/l in the septic tank effluent and 89 mg/l in the ground-water zone below the tile field. These values were much higher than background $Br^-$ concentrations (<0.5 mg/l). The test showed that the average septic tank residence time was about 2 days and that the effluent residence time in the 2-m-thick unsaturated zone and capillary fringe was on the order of 10 days. Figure 6 shows the position of peak $Br^-$ concentrations in the ground-water zone at times of 10 to 350 days after injection. During the 350-day period, the slug migrated to a maximum depth of about 2.5 m below the water table and moved horizontally away from the tile bed a distance of 23 m downgradient. These data suggested a maximum vertical ground-water velocity of 1.7 cm/day in the area under the center of the tile bed and a horizontal velocity of 7 cm/day (24 m/a) in the area immediately downgradient of the tile field. Thus, the bromide tracer test and calculations using the Darcy equation suggested that the horizontal flow velocity in the surficial aquifer at Cambridge was on the order of 20 to 40 m/a.

At the Muskoka site, the bromide tracer test was less successful because there were fewer monitoring points and these were sampled less frequently. It appears that the $Br^-$ tracer experienced a longer residence time, on the order of several weeks to months, in the 3-m-thick unsaturated zone at this site. The longer residency at Muskoka was likely the result of lower effluent dose rates (only two persons using the system) and higher evapotranspirative losses during the test period. The tracer test was begun in late June at Muskoka while it was begun in early May at Cambridge. Elevated $Br^-$ concentrations were observed under the tile bed, however, at a depth of 0.5 m below the water table, 90 days after injection. Although the bromide test was unsuccessful, an alternative method provided an estimate for ground-water velocity. The velocity along most of the septic system plume was determined by observing the rate of advance of the zone of elevated EC (Figure 6). This was possible because ground water impacted by the tile effluent was easily identified by its contrasting electrical conductance. A horizontal flow velocity of about 20 m/a was indicated by this method.

At the Muskoka site, there is a large water-table gradient of about 0.01 in a direction toward the river (Figure 2). Using the permeameter-indicated average K value of $1.3 \times 10^{-3}$ cm/s and assuming a porosity of 0.35, the Darcy equation gave a horizontal flow velocity of 12 m/a. Thus, calculations using the Darcy equation and observations of plume front arrival suggested that horizontal flow rate was on the order of 10 to 20 m/a at the Muskoka site.

### Plume Character

Table 2 summarizes the major-ion geochemistry of the background ground water, the tile effluent, and the plume core ground water at both sites. Figure 1 shows vertically averaged $Na^+$ distribution in the surficial aquifer at Cambridge, while Figure 3 shows $Na^+$ distribution along the plume centerline (section A-A'). At the Cambridge site, plume core water was characterized by high $Na^+$ (71 to 91 mg/l), $Cl^-$ (21 to 29 mg/l), and $NO_3^-$-N (21 to 48 mg/l). However, because background ground water at this site also had high $Cl^-$ (10 to 29 mg/l) and $NO_3^-$-N (17 to 34 mg/l), $Na^+$ was found to be the best indicator of impact from the septic system because it occurred in the plume at 10 to 20 times the background level. The high background $NO_3^-$ and $Cl^-$ values were the result of agricultural practices such as manure and chemical fertilizer application.

A long, narrow plume more than 130 m long and about 10 m wide was defined at Cambridge (Figure 1). Sodium concentrations that were more than 50 percent of the effluent value occurred 130 m downgradient from the tile field. This demonstrated the relatively rapid advection of the plume and the ineffectiveness of dispersion, laterally and vertically, to significantly attenuate the plume. Figure 3 shows that in the area of the tile field, the septic system plume was about 2.5 m thick and occupied the entire thickness of the aquifer except for the bottom 0.5 m. Downgradient from the tile field, the plume became overlain by an

**Table 2. Major Ion Geochemistry**

| | *Cambridge site* | | | *Muskoka Site* | | |
|---|---|---|---|---|---|---|
| | *Upgrad.* *N = 3* | *Tile effluent*[1] *N = 4* | *Plume*[2] *core* *N = 7* | *Upgrad.* *N = 2* | *Tile effluent*[1] *N = 5* | *Plume*[3] *core* *N = 8* |
| Na, mg/l | 4 | 98 | 86 | 2 | 90 | 45 |
| K, mg/l | 2 | 12 | 11 | 1 | 21 | 14 |
| Ca, mg/l | 88 | 40 | 90 | 8 | 14 | 44 |
| Mg, mg/l | 16 | 14 | 17 | 1 | 3 | 3 |
| Alk, mg/l | 204 | 365 | 276 | 7 | 316 | 12 |
| $SO_4$, mg/l | 59 | 27 | 63 | 6 | 42 | 32 |
| Cl, mg/l | 17 | 45 | 24 | 4 | 55 | 38 |
| $NO_3^-$-N, mg/l | 27 | 1 | 33 | 3 | 0.1 | 39 |
| $NH_4$-N, mg/l | <0.1 | 30 | 0.1 | <0.1 | 59 | 0.5 |
| $PO_4$-P, mg/l | <0.01 | 8 | 4 | <0.01 | 13 | 0.01 |
| DOC, mg/l | 3 | 37 | 4 | 1.6 | 50 | 3.4 |
| pH | 7.3 | 7.9 | 7.0 | 5.7 | 7.6 | 5.1 |
| EC, μmhos/cm | 770 | 1020 | 940 | 75 | 950 | 550 |

[1] Average composition of samples from weeping tile access tube.
[2] Average composition of samples from piezometer bundles; 33, 3.0 m and 4.2 m depths; 34, 2.4 m, 3.0 m, and 4.8 m depths; and 35, 2.1 m and 3.8 m depths. Locations shown on Figure 3.
[3] Average composition of samples from piezometer bundles; 32, 3.5 m depth; and 30, 2.6 m depth. Locations shown on Figure 4.

increasing thickness of nonimpacted ground water recharged from the overlying farm field. The boundary between the plume and the overlying nonimpacted zone was abrupt which reflects the low vertical transverse dispersion capability of the aquifer material.

Figure 3 also shows DO, $NO_3^-$-N, $PO_4^{3-}$-P, DOC, and pH distribution along section A-A'. The plume core was characterized by depressed levels of DO (<1 mg/l) compared to background values of 3 to 7 mg/l, but detectable amounts of DO (>0.1 mg/l) persisted at all locations. $NO_3^-$-N levels were two to four times in excess of the Ontario drinking-water limit of 10 mg/l (MOE, 1983). $PO_4^{3-}$ was observed in the plume only in the area beneath the tile field. The plume core showed slightly depressed pH levels (7.0) compared to background values (7.3).

Major ion geochemistry along the plume core at Muskoka (Figure 4) was similar to that at Cambridge for some species, with comparable levels of $Na^+$, $Cl^-$, and $NO_3^-$-N measured. The plume descended to a depth 2 m below the water table in the tile area and became overlain downgradient by nonimpacted ground water recharged through the overlying lawn area. The upper and lower plume boundaries were again sharp, demonstrating the low vertical dispersion capability of the aquifer and again demonstrating the ineffectiveness of dilution in attenuating plume solute concentrations. Depressed levels of DO were measured along the plume core with concentrations similar to those at Cambridge observed. The plume core again exhibited depressed levels of pH (5.1) compared to background values (5.7). There was no detectable $PO_4^{3-}$ (>0.02 mg/l $PO_4^{3-}$-P) in the groundwater zone at Muskoka.

Nitrogen in septic system effluent occurs about 80 percent as inorganic nitrogen, with $NH_4^+$-N dominating (Walker et al., 1973a; Andreoli et al., 1979). Inorganic nitrogen commonly occurs at concentrations ranging from 30-111 mg/l in septic system effluent (Walker et al., 1973a; Magdoff et al., 1974; Viraraghaven and Warnock, 1975; Andreoli et al., 1979). Thus, the total inorganic nitrogen concentrations of the effluent at Cambridge and Muskoka ($NH_4^+$-N + $NO_3^-$-N from Table 2; 31 and 59 mg/l, respectively) appear typical of most septic system effluents. It is difficult to find literature values for nitrogen content in septic system-impacted ground waters from sandy aerobic environments, however, several workers report $NO_3^-$-N concentrations in the subsurface below tile lines that are similar to our groundwater values (Walker et al., 1973b; Andreoli et al., 1979; 10-50 mg/l $NO_3^-$-N). $NO_3^-$-N values in septic system-impacted ground waters are commonly above the drinking-water limit of 10 mg/l (Childs et al., 1974; Rea and Upchurch, 1980). Thus, our $NO_3^-$ results and plume distances are probably not unusual.

## Geochemical Processes

A comparison of tile effluent and plume core chemistry (Table 2) shows that at both sites plume water contained much less DOC (1.6-3 vs 37-50 mg/l), much higher $NO_3^-$-N (33-39 vs 0.1-1 mg/l), and much lower $NH_4^+$-N (0.1-0.5 vs 30-59 mg/l) than the tile effluent. This suggested that: (1) organic carbon in the effluent was being consumed by an aerobic biodegradation reaction such as;

$$CH_2O + O_2 \rightarrow CO_2 + H_2O \qquad (1)$$

where the simple carbohydrate $CH_2O$ is used conceptually here to represent organic matter, and that (2) ammonium in the effluent was being oxidized via microbial nitrification.

$$NH_4^+ + 2O_2 \rightarrow NO_3^- + 2H^+ + H_2O \qquad (2)$$

The low DOC levels and high $NO_3^-$ levels observed in even the shallowest water-table zone below the tile fields indicates that these processes were completed largely during the residency of the effluent in the unsaturated zone. Likewise, Walker et al. (1973a) observed significant nitrification of $NH_4^+$ within 6 cm below the gravel pack/soil interface and after effluent residency of only a few hours, in sandy aerobic unsaturated zones below several septic systems in Wisconsin. However, the continued depletion of DO and DOC in the zone below the water table (Figures 3 and 4) suggests that these processes continued to some extent below the water table. Since ammonium-N found in the tile effluent (30-59 mg/l) was approximately equivalent to nitrate-N found in the plumes (33-39 mg/l), the tile effluent was the probable source of nearly all of the nitrate in the plumes. A similar conclusion was reached by Walker et al. (1973a) regarding the source of pore-water $NO_3^-$ in the unsaturated zones below the septic systems in Wisconsin. The relatively long residency (>7 days) of the effluent in the aerobic unsaturated zones at both Cambridge and Muskoka allowed reactions (1) and (2) to be completed largely within the vadose zones at these sites. At other locations where vadose zone residency is shorter due to a shallow water-table condition or due to a higher effluent infiltration rate, these reactions may not be complete within the vadose zone. This would tend to cause higher $NH_4^+$, lower $NO_3^-$, higher DOC, and lower DO, as well as other differences in plume chemistry.

Acidity produced by $NH_4^+$ nitrification [equation (2)] has resulted in depressed pH levels in the cores of both plumes (Figures 3 and 4; Table 2). At Cambridge, the pH depression was minor compared to background pH (7.0 versus 7.3) due to carbonate buffering, but at Muskoka, it was more significant (5.1 versus 5.7) due to low buffering capacity of the carbonate-poor aquifer sands there (Table 1). At Muskoka, pH levels in the plume may be sufficiently low to cause enhanced mobility of some constituents such as metals. This topic will be investigated in a subsequent study.

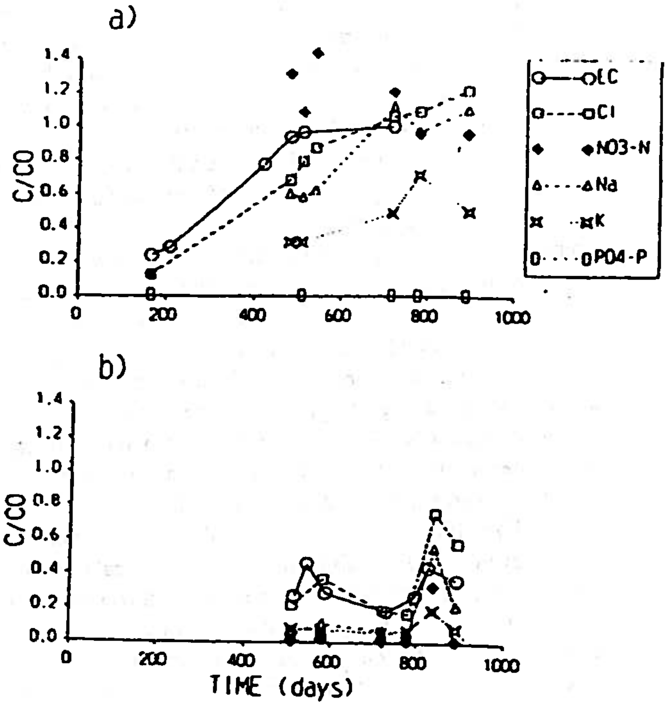



**Fig. 7. Normalized major ion breakthrough at the Muskoka site: (a) 17 m downgradient from the tile field (piezometer 19, 2.4 m depth), and (b) 24 m downgradient (seepage meter 5). ($t_0$ = April 1987, $x_0$ = center of tile field, Co from plume core values given in Table 2, except effluent value used for $PO_4^{3-}$-P).**

At Muskoka, plume development and migration rate has been documented because ground-water monitoring was begun six months after the beginning of system operation. Figure 7 shows normalized breakthrough of the major ions at piezometer 19-9 located in the plume core 17 m downgradient of the tile bed, and at seepage meter 5 located on the riverbed in the center of the plume discharge area (Figure 2). Source concentrations for normalization (Co) were taken as the plume core average values given in Table 2. At piezometer 19-9, $NO_3^-$ and $Cl^-$ appeared most mobile, arriving after about 300 days of system operation, at about the same time that elevated EC levels were observed. $Na^+$ arrival was slightly retarded compared to $Cl^-$, indicating probable minor adsorption of $Na^+$. $Na^+$ remained relatively mobile, however, and had a migration velocity at least 75 percent that of $Cl^-$. $K^+$ was more highly attenuated and retarded than $Na^+$. Table 1 indicates that the Muskoka aquifer sand has a low cation exchange capacity (2 meq/100 g).

The condition of $NO_3^-$ breakthrough in the riverbed seepage meter is of particular interest at 500 to 600 days of system operation. $NO_3^-$-N levels remained <0.5 mg/l in the seepage meter while $Cl^-$ C/Co values rose to 0.3 (Figure 7b). This was observed even though $NO_3^-$ occurred at uniformly high concentrations and was shown to be very mobile throughout the remainder of the plume (Figure 7a). It is likely that vigorous denitrification occurred in the riverbed sediments as a result of the development of anaerobic conditions there. In such an environment, $NO_3^-$ can be converted to nitrogen gas by a reaction such as,

$$\frac{4}{5} NO_3^- + CH_2O \rightarrow \frac{2}{5} N_2 + HCO_3^- + \frac{1}{5} H^+ + \frac{2}{5} H_2O \quad \ldots (3)$$

where $CH_2O$ represents organic matter. Such a phenomenon is strongly suggested by Figure 4 and Table 3 which show that $NO_3^-$-N decreased from about 20 mg/l to less than 0.5 mg/l in the last meter or so of the flow path before discharging into the river. Starr (1988) demonstrated the role of labile organic carbon in promoting the development of anaerobic conditions and denitrification in sandy aquifers. Increased availability of organic carbon in the riverbed sediments is suggested from Table 1, which shows an foc level, for the riverbed sediment, 60 times higher than that of the aquifer sand. Table 3 indicates that the most vigorous zone of denitrification was at a depth of 0.5 m below the riverbed. Nitrate continued to be attenuated in the riverbed sediment at Muskoka; however, some variability in the attenuation capacity, perhaps seasonally or water-level-related, was indicated by the temporary breakthough of $NO_3^-$ during the July 1989 sampling (T = 840 days, Figure 7b).

Phosphate occurred in the effluent at both sites at concentrations of about 10 mg/l $PO_4^{3-}$-P (Table 2), but was highly attenuated in the subsurface. Such a condition is commonly observed for $PO_4^{3-}$ in subtile sediments (Sawhney and Starr, 1977; Jones and Lee, 1979). At Cambridge, $PO_4^{3-}$-P levels of > 1 mg/l were mostly confined to the aquifer area immediately below the tile bed, while at Muskoka, no detectable $PO_4^{3-}$-P (>0.02 mg/l) was observed in the ground-water zone. Although the phosphate attenuation processes have not as yet been rigorously evaluated, at Cambridge it was noted that very little phosphate attenuation occurred in the unsaturated zone while significant attenuation (>5 mg/l to <0.02 mg/l) occurred after several meters of flow within the ground-water zone. The high-phosphate ground waters have been shown to be oversaturated with respect to hydroxylapatite [$Ca_5(OH)(PO_4)_3$] (Wilhelm, 1990), and the zone of phosphate attenuation is also a zone

**Table 3. $NO_3^-$, $Cl^-$, and $NH_4^+$ Concentrations in Riverbed Sediments, Muskoka Site; Ground-Water Samples Obtained from Seepage Meter 5 and from Drive-Point Piezometers Installed Below Seepage Meter 5**

| *Depth below riverbed (m)* | *Sampling date* | *$NO_3^-$-N mg/l* | *$Cl^-$ mg/l* | *$NH_4^+$-N mg/l* | *Sampling date* | *$NO_3^-$-N mg/l* | *$Cl^-$ mg/l* | *$NH_4^+$-N mg/l* |
|---|---|---|---|---|---|---|---|---|
| 0.0 (SM5) | 12/88 | 0.6 | 14 | 0.3 | 10/89 | <0.05 | 22 | <0.05 |
| 0.25 | 4/89 | 3.1 | 15 | <0.05 | 10/89 | <0.05 | 36 | 0.24 |
| 0.50 | 4/89 | 13 | 7 | <0.05 | 10/89 | 8.6 | 30 | <0.05 |
| river water | 9/88 | 0.1 | 1.9 | <0.05 | | | | |

of increasing pH (7.0 to 7.3) and increasing $Ca^{2+}$ concentration (80 to 100 mg/l) which are factors that would decrease hydroxylapatite solubility. Phosphate attenuation at Cambridge may thus be controlled by hydroxylapatite precipitation. At the Muskoka site, $PO_4^{3-}$ mobility was less likely controlled by hydroxylapatite solubility because $Ca^{2+}$ concentrations in the plume were lower (44 mg/l), and pH was much lower (5.1). Phosphate at Muskoka may be attenuated by the precipitation of other sparsely soluble phosphate minerals such as strengite ($FePO_4 \cdot 2H_2O$), or varisite ($AlPO_4 \cdot 2H_2O$) or may be controlled by sorption.

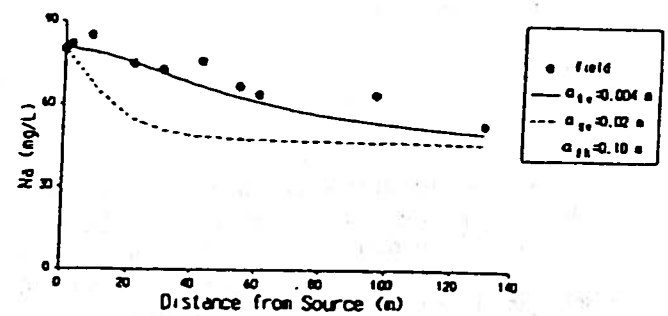



**Fig. 8. Simulated steady-state $Na^+$ distribution along the plume core at Cambridge using a 3D analytical model ($\alpha_L$ = 1 m, $\alpha_{th}$ = 0.01 m, $\alpha_{tv}$ = 0.004 m except where noted, v = 30 m/a, $D^*$ = 0.02 $m^2$/a).**

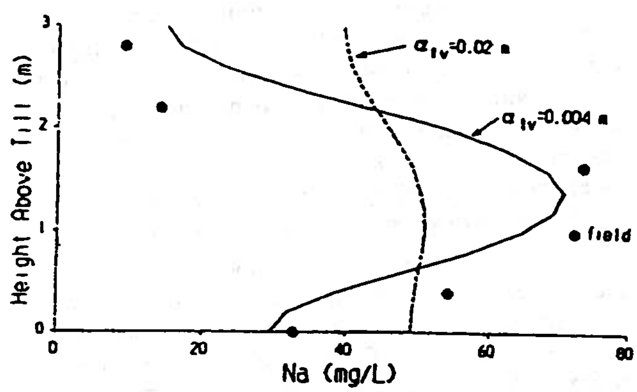



**Fig. 9. Simulated steady-state vertical $Na^+$ distribution 30 m downgradient from the tile field (piezometer bundle 39) at the Cambridge site using a 3D analytical model ($\alpha_L$ = 1 m, $\alpha_{th}$ = 0.01 m, v = 30 m/a, $D^*$ = 0.02 $m^2$/a).**

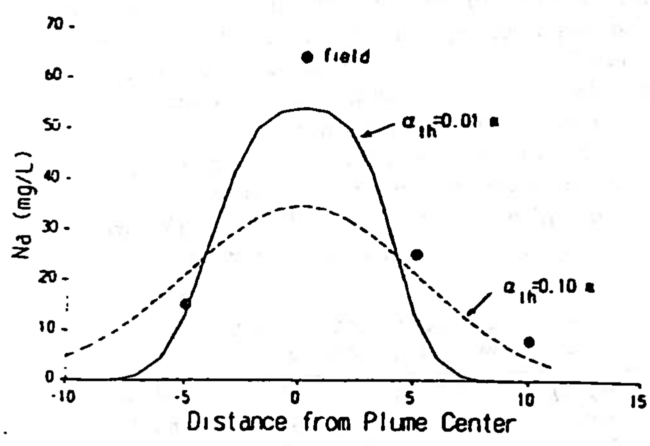



**Fig. 10. Simulated steady-state $Na^+$ distribution transverse (horizontally) to the plume core at Cambridge, 95 m downgradient from the tile field (v = 30 m/a, $\alpha_L$ = 1 m, $\alpha_{tv}$ = 0.004 m, $D^*$ = 0.02 $m^2$/a).**

## Aquifer Dispersion

At Cambridge, a three-dimensional analytical model (Sudicky, 1985) was employed to obtain preliminary estimates of the aquifer dispersion parameters within the saturated zone. For the simulations, a vertical patch source 1.75 m thick by 8 m wide in the direction perpendicular to flow, was positioned in the middle of the saturated part of the aquifer at the downgradient edge of the tile field. Aquifer thickness was set at 3 m. A uniform horizontal flowfield with average linear ground-water velocity of 30 m/a was assigned to the aquifer; otherwise, no boundary fluxes were specified. To simulate $Na^+$ distribution, ground water exiting the source was assigned a solute concentration of 80 mg/l. $Na^+$ was assumed to be nonreactive. Figure 8 shows that $Na^+$ concentrations along the plume core were well represented by the model when low dispersion parameters typical of values determined from recent natural-gradient tracer experiments in sands (Sudicky et al., 1983; Freyberg, 1986; Garabedian, 1987; Moltyaner and Killey, 1988a, b) are used [i.e., longitudinal dispersivity ($\alpha_L$) = 1 m, vertical transverse dispersivity ($\alpha_{tv}$) = 0.004 m, and horizontal transverse dispersivity ($\alpha_{th}$) = 0.01 m]. Use of larger values of $\alpha_{tv}$ or $\alpha_{th}$ resulted in $Na^+$ concentrations along the plume core that were less than those observed (Figure 8). An effective diffusion coefficient ($D^*$) of 0.02 $m^2$/a was used for all simulations. The closeness with which observed $Na^+$ concentrations could be simulated using a constant source concentration of 80 mg/l suggested that $Na^+$ loading from the septic system had been consistent during the operation of the system. This was also suggested by $Na^+$ concentrations observed in the plume core below the tile bed where, from 19 sampling points representing effluent discharged over about a six-month period, all $Na^+$ values were between 71 and 91 mg/l.

Figure 9, which shows simulated vertical $Na^+$ profiles 30 m downgradient from the tile field (piezometer bundle 39), demonstrates that a very low $\alpha_{tv}$ value of about 0.004 m or less was necessary to maintain the sharp $Na^+$ gradient along the plume upper boundary. Figure 10 demonstrates that a low value of $\alpha_{th}$ on the order of a few centimeters or less was necessary to maintain the narrow plume width (10 m) at downgradient locations. These simulations thus confirmed our observation that the aquifer at Cambridge had a low capacity for dispersion. Although no simulations of the Muskoka plume were attempted, the sharp plume boundaries suggest that dispersion there was also low.

## Conclusions and Implications

Modeling results indicated that the vertical transverse dispersivity of the Cambridge aquifer was on the order of 0.4 cm or less and that horizontal transverse dispersivity was less than 10 cm. Dispersion in the Muskoka aquifer also appeared to be low. The weak transverse dispersion process observed in the two plumes is significant in that it is consistent with very detailed tracer tests of the type recently performed at Borden and Twin Lakes, Ontario, and at Cape Cod, Massachusetts.

It is of interest to examine the implication of these dispersion values with respect to the impact of septic systems on ground water. Although no field estimates for longitudinal dispersion were obtained from this study, sensitivity analyses conducted using the previously discussed analytical model demonstrated that this parameter is likely to be unimportant in diluting plumes in low dispersion sands, even over long distances. For example, for a Cambridge-type plume 300 m in length, the portion of the plume influenced by longitudinal dispersion would be only the frontal 50 m or so when $\alpha_L$ was equal to 1 m. For the remainder of the plume length, dilution of nonreactive solutes would occur by transverse dispersion only. This conclusion is consistent with model sensitivity studies conducted by Frind and Hokkanen (1987). Using the analytical model and again considering the Cambridge example, the portion of the steady-state plume that would have contaminants at concentrations above drinking-water limits was estimated. For a plume with a source concentration of 33 mg/l $NO_3^-$-N, and where $NO_3^-$ was absent in background ground water, the steady-state plume length in which $NO_3^-$-N would be in excess of the drinking-water limit of 10 mg/l is 170 m when $\alpha_{th}$ and $\alpha_{tv}$ values are 0.1 m and 0.004 m, respectively. However, the plume length in which $NO_3^-$-N would be in excess of one-fourth of the drinking-water limit (2.5 mg/l) is much longer, about 2 km. In some jurisdictions (i.e., Province of Ontario) this is the allowable impact from a single septic system. These "above the limit" plume lengths are estimated using the upper range of the dispersion parameters indicated by the field data at Cambridge. Even longer plumes result if smaller dispersion parameters are considered for sandy aquifers. Such examples demonstrate that for many unconfined sand aquifers, the typical minimum permissible distance-to-wells regulations that exist in most parts of the United States and Canada for septic systems (25-35 m) should not be expected to be adequately protective of well-water quality, except in those circumstances where it can be shown that significant mobile contaminants such as $NO_3^-$ are attenuated by processes other than dispersion (i.e., by biodegradation). An example of such attenuation is provided by the Muskoka site, where the $NO_3^-$ plume is terminated by denitrification.

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**ATTACHMENT B – POSTER FROM 1995 NORTH AMERICAN LAKE MANAGEMENT SOCIETY CONFERENCE**

In early November, 1995, information was presented at the annual North American Lake Management (NALMS) Society conference held in Toronto which indicated that while phosphorus movement in sandy calcareous soils occurs at a rate of about 1 m per year, the upper horizon of soils of Precambrian Shield origin have a high capability for retaining phosphorus due to their naturally high iron content and low pH. The following nôtes reproduced from a poster session (**Phosphate Minerals in the Vadose Zone at Septic System Sites** by L. Zanini, W. D. Robertson, D. J. Ptacek, and S. L. Schiff), a presentation (**Septic System Phosphorus in Precambrian Shield Country** by S. L. Schiff, W. D. Robertson, L. Zanini, J. Wood and R. Elgood), and a presentation and poster session (**Laboratory Studies on the Development of a Reactive Mixture to Remove Phosphates from Septic System Effluent** by M. J. Baker, D. W. Blowes, W. D. Robertson, and C. J. Ptacek) at the NALMS conference.

**Phosphate Minerals in the Vadose Zone at Septic System Sites** by L. Zanini, W. D. Robertson, C. J. Ptacek and S. L. Schiff (Poster Presentation).

Phosphorus as a common nutrient derived from sewage wastes has limited mobility in the unsaturated and saturated zones at septic system sites. Its high retardation values are attributable to the retention capacity of soil through which the system drains. Common reactions which may enhance or limit phosphorus mobility are adsorption/desorption, precipitation/demineralization. Soil core samples were collected from three septic systems (Cambridge, Longton and Muskoka) to determine the mechanisms and extent of phosphorus attenuation. Total phosphorus evaluations combined with micropore and pore water chemical analysis provided an understanding of phosphorus retardation in unsaturated sands.

Total phosphorus concentrations were determined by digesting core sediments with 1 N NaCl. The extractions were performed on the cores every 5 cm to 10 cm of depth and compared with the core log data. At both the Cambridge and Langton sites, high phosphorus concentrations were seen immediately below the tile bed of the septic system. In contrast, the fine sand layer at Muskoka below the tile bed had a strong attenuation function.

Microprobe results: Sediments from the high phosphorus areas of concern were analyzed by a JXA - 8600 Microprobe. Back scattered electron (BSE) imaging provided evidence of secondary phosphate phosphorus mineralization. Much of the new particulate exists either as a "coating" on the grains of the soil matrix, or as individual mineral grains. It is believed that grains on the porus media act as a nucleus for phosphate phosphorus mineral growth. Microbe analyses determined the five most common oxides within the phosphate phosphorus grain. Results indicate that iron phosphate is precipitating at Cambridge whereas aluminum phosphates prevail at Longton and Muskoka.

**Mass balance**: Based on average loading rates, measured phosphate phosphorus effluent concentrations, and the area of the tile bed, mass balance calculations were performed to determine the amount of phosphate phosphorus that should be retained in the unsaturated zone. Some assumptions for these calculations were (i) that the tile field is fully functional allowing uniform infiltration of effluent; (ii) that phosphorus measured in the plumes was not included. Average values for the high phosphate phosphorus concentrated zones measured directly beneath the tile beds (and in the fine sand layer in Muskoka) were compared to calculated amounts of phosphate phosphorus concentrated in a similar interval of unsaturated sand.

**Pore water**: Pore water was collected near the original location of the core and analyzed for total inorganic ions. Secondary phosphate phosphorus minerals precipitate iron, aluminum and calcium. Iron concentrations are highest in the unsaturated zone pore waters at Cambridge whereas aluminum dominates at both Langton and Muskoka. The resulting inorganic ion chemistry was analyzed using the geochemical model PHREEZE; stregnite (iron phosphate) precipitates at Cambridge and varisite (aluminum phosphate) precipitates at both Langton and Muskoka.

**Conclusions:** High phosphorus attenuation at the Cambridge, Langton and Muskoka sites was observed in the narrow soil zones just beneath the tile bed or entrapped in porous media of low permeability. Microprobe analysis and geochemical modelling of pore water chemistry provides evidence of secondary phosphate phosphorus precipitation. Phosphate phosphorus precipitates with elements derived from the porous media which may be controlled by the pH and redox reactions of the unsaturated systems. Further research is required to determine specific geochemical environments which retain phosphate phosphorus.

The following are characteristics of the three sites:

| | | |
|---|---|---|
| Cambridge | – | small family dwelling |
| | – | 4 people |
| | – | tile bed has been operating for 18 years |
| Langton | – | school site (rural) |
| | – | 200 people during school year |
| | – | tile bed has been operational for 44 years |
| Muskoka | – | small family dwelling |
| | – | 2 people |
| | – | tile bed has been operational for 7 years. |

**Septic System Phosphorus in Precambrian Shield Country** by S. Schiff, W. Robertson, L. Zanini, J. Wood and R. Elgood (Presentation)

A tile field system treating sewage wastes from a single family residence near Cambridge has been monitored for a number of years. The tile field has been in operation for 17 – 18 years. The data indicate the phosphate phosphorus is mobile; in this regard, concentrations of 1.6 mg/L (as phosphate phosphorus) were found 20 m downgradient from the tile bed. If the tile bed had been sited on the shoreline of a lake in similar soil conditions, it would have loaded phosphate phosphorus to the lake. pH of the groundwater is about 7.

At the Langton site which is a decommissioned school, the phosphate phosphorus plume extended about 100 m from the tile bed. The greater length of plume (than at the Cambridge site) was due to the difference in ages of operation between the systems. The pH of the groundwater was about 7.

In Muskoka, very different results were noted.

At one site on the Muskoka River, the groundwater zone pH is low (5.2) and as low as 3 – 4. The soils are non-carbonate minerals and have little buffering capacity. After two years of operation, 100% of the phosphate phosphorus was attenuated within the unsaturated zone (phosphorus concentration in the effluent from the septic tank is about 10.0 mg/L). The system is now about eight years old, and there is still no phosphate phosphorus at the water table. The reason for the attenuation is that the soils underlying the tile bed are on iron rich sand, and phosphate phosphorus is being precipitated. Phosphate phosphorus concentrations at every few cm of depth indicate a spike or highest concentrations immediately beneath the tile beds; the size of this spike indicates that nearly 100% of the phosphate phosphorus loading generated over the eight year period of operation was attenuated and caught within the layer.

At a second site on the shoreline of Harp Lake, the soils were a heterogenous coarse sand boulder complex and not as uniform as at the Muskoka River site. In general, phosphate phosphorus concentrations were much lower than at the Muskoka River site; however, sporadically high concentrations occurred, including the odd high values at the lakeshore. The small amount of phosphorus reaching the lake may have done so through preferential flow routes such as macropores. At a second cottage site on Harp Lake, the high phosphate phosphorus spike immediately below the tile bed accounted for 100% of the phosphorus loaded to the system.

**Laboratory Studies on the Development of a Reactive Mixture to Remove Phosphates from Septic System Effluent** by M. J. Baker, D. W. Blowes, W. D. Robertson and C. J. Ptacek. (Presentation and Poster Session)

Phosphorus is recognized as being the most important contributor to the enhanced eutrophication of freshwater lakes. Shallow, groundwater flow systems, which may eventually discharge to sensitive surface waters, can become contaminated with phosphate phosphorus from domestic septic systems. Phosphorus as ortho-phosphate ($PO_4^{-3}$) is a non-conservative solute. In groundwater, soil water interactions can retard its migration. Phosphate phosphorus is mobile where the natural retardation potential is insufficient or has been depleted due to long term loading. **<u>In situ</u>** geochemical barriers may be successful at limiting the movement of phosphorus in groundwater. Laboratory experiments are being conducted to test potential reactive mixtures before they are applied in the field.

**The technology**: Limiting phosphate phosphorus migration involves the installation of a passive system that treats groundwater **in situ**. A geochemical barrier can be installed in two ways (Figure 2a and 2b). In both examples, an existing phosphate phosphorus plume can be treated **in situ** by a permeable reactive wall which intercepts the plume and treats the groundwater before it discharges to the lake.

**Phosphate phosphorus attenuation mechanism**: The objective of the reactive mixture is to provide a geochemical environmental in which the aqueous phosphate phosphorus is preferentially partitioned to the solid phase by specific adsorption onto metal-oxide surfaces.

**Laboratory studies**: Solid mixtures containing sources of iron, aluminum, and calcium are being evaluated for their ability to remove phosphates from tile bed effluent. Batch experiments indicate that treatment efficiency varies and that >90% phosphate phosphorus approval can be attained in short reaction times (<1 – 10 hours). Column (20 cm in length) experiments have been used to evaluate the mixtures under dynamic or continuous flow conditions and cumulative phosphorus loadings. An average phosphate phosphorus removal efficiency of 93% over 1,000 pore volumes or 2.5 years of treatment has been achieved.

**Conclusions:** (i) Reactive mixtures have been developed to remove phosphate phosphorus from septic system effluent using inexpensive readily available materials. (ii) Batch experiments show that the effectiveness of the mixtures can vary and that 100% removal can be achieved in <1.0 hour. (iii) A column experiment (5% metal oxide/45% limestone/50 sand) has continued to remove 90% of the influent phosphate phosphorus after 2.5 years of treatment. (iv) The shape of the column breakthrough curve along with geochemical modelling suggest that both adsorption and precipitation are controlling phosphate phosphorus in the column. (v) Current field experiments include removing phosphate phosphorus from a groundwater septic plume and municipal waste water.

**ATTACHMENT C – REVIEW OF PHOSPHATE MOBILITY AND PERSISTENCE IN 10 SEPTIC SYSTEM PLUMES (W.D. ROBERTSON, S.L. SCHIFF, AND C.J. PTACEK, NOVEMBER – DECEMBER, 1998)**

# Review of Phosphate Mobility and Persistence in 10 Septic System Plumes

by W.D. Robertson[a], S.L. Schiff[a], and C.J. Ptacek[b]

## Abstract

Phosphate distribution was reviewed in 10 mature, highly monitored septic system ground water plumes in central Canada. It was shown that six plumes (primarily those on calcareous sands) of enriched P concentrations (0.5 to 5 mg/L P) exceeding 10 m in length are present. In each case, phosphate migration velocity is highly retarded (retardation factor, 20 to 100) compared to the ground water velocity, but migration rates remain sufficiently fast (~1 m/a) to be of concern when considering long-term operation and the normal setback distance of septic systems from adjacent surface water bodies (~ 15 m). Much smaller scale phosphate plumes (< 3 m in length) are present at the acidic sites on noncalcareous sands and on silt- and clay-rich sediments.

At all of the sites, ground water concentrations are lower than effluent values by amounts ranging from 23 to 99%, suggesting that P accumulation has occurred in the vadose zone. This was confirmed by sediment analyses at four of the sites which, in each case, showed that zones of P enrichment were present within 1 m of the infiltration pipes (Wood 1993; Zanini et al. 1998). Also, observed phosphate concentrations are generally consistent with values expected based on the solubility constraints of the minerals vivianite in reducing zones (including the septic tank), and strengite and variscite in oxidizing zones, providing further evidence that mineral precipitation reactions play a role in limiting P concentrations. Strengite and variscite have the potential to limit P to low concentrations (<0.1 mg/L) under acidic conditions, but oxidation of sewage effluent leads to acidic conditions only in noncalcareous terrain or beneath old septic systems where calcium carbonate has been depleted. Overall, phosphate plume migration velocities in ground water appear to be controlled by sorption processes, but the phosphate concentrations that are present in the plumes appear to be strongly controlled by mineral precipitation reactions that occur in close proximity to the infiltration pipes.

## Introduction

Concentrations of phosphorus normally found in sewage effluent (~5 to 20 mg/L) are far in excess of the much lower concentrations (~0.03 mg/L) that have been observed to stimulate algae growth in aquatic environments (Dillon and Rigler 1974; Schindler 1977). As a consequence, phosphorus is often the contaminant of greatest concern when septic systems are located close to sensitive surface water bodies. The mobility of phosphate in the subsurface remains uncertain, however, as a result of the considerable reactivity of this constituent. Phosphate is strongly adsorbed by most sediments (e.g., Parfitt et al. 1975; Rajan 1975; Isenbeck-Schröter et al. 1993) and is capable of combining with a number of metal cations, particularly iron, aluminum, manganese, and calcium, to form a wide range of minerals that can be stable in low-temperature aqueous environments (Nriagu and Dell 1974; Stumm and Morgan 1981).

In this study, the behavior of phosphate is reviewed at 10 septic system sites in central Canada that have been investigated in considerable detail during the past decade. These include systems servicing two seasonal-use cottages (Killarney and Harp sites), three single-family households (Cambridge, Muskoka, and Paradise sites), two seasonal-use campgrounds (Long Point and Camp Henry sites), a tourist resort (Delawana site), and a public school (Langton site). All are mature septic systems that have been in operation for at least six years, and in most cases detailed ground water monitoring networks consisting of up to 500 monitoring points each have been installed so that flowpaths emanating from the septic systems have been identified with confidence. A variety of silt and sand sediments are represented and geochemical conditions in the plumes vary over a wide range of pH and Eh values. Previous investigations of $PO_4$ behavior in septic system plumes (Childs et al. 1974; Reneau and Pettry 1976; Sawhney and Starr 1977; Beek et al. 1977; Cogger et al. 1988; Whelan 1988; Walter et al. 1996) have, in most cases, been restricted to one or two sites. The objective here is to consolidate the site-specific results into a more comprehensive assessment of phosphorus transport in the sediments below conventional septic tank infiltration beds.

## Methods

At the sites on sand, the monitoring networks consist primarily of multiple piezometer bundles of the type described by Cherry et al. (1983). At the sites on silty sediments, individual piezometers were installed in manually augered holes or drive point-type piezometers were installed with a percussion hammer.

Plume configurations were determined by sampling indicator parameters such as electrical conductance (EC) and $Cl^-$ (Killarney, Long Point, Camp Henry, Harp, and Delawana sites), $Na^+$ (Cambridge site) or $NO_3^-$ (Muskoka and Langton sites). In most

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cases, more detailed sampling was then concentrated along the plume centerlines and generally included sampling for nutrients ($NO_3^-$, $NH_4^+$, and $PO_4$), major ions ($Ca^{2+}$, $Mg^{2+}$, $K^+$, $Na^+$, $Cl^-$, $SO_4^{2-}$, and $HCO_3^-$), trace metals (Fe, Mn, and Al), and dissolved organic carbon (DOC).

$PO_4$, in most cases, was analyzed colorimetrically on field-filtered and acidified (pH <2) samples using a COBAS FARA® infrared spectrometer which provided a lower detection limit of 0.01 to 0.02 mg/L P. Generally only orthophosphate ($PO_4$) concentrations were measured since this is the chemically reactive form. Although this may underestimate total phosphorus concentrations in high DOC waters such as the raw waste water, in most of the plume zones this analysis is expected to represent the bulk of the phosphorus present in solution. To confirm this, six samples were retrieved from the Langton plume (January 1998) from zones encompassing a range of P values (0.1 to 2.7 mg/L) and were analyzed for both $PO_4$ and total dissolved P. In these samples, total P averaged only 8% higher than $PO_4$ concentrations. Reneau and Pettry (1976) and Harman et al. (1996) have also noted that $PO_4$ and total dissolved P concentrations are generally equivalent in aquifer zones impacted by septic systems. Fe was analyzed by inductively coupled plasma spectrometry (ICP), which provided a detection limit of 0.001 to 0.02 mg/L. Low-level Al analyses were done by ICP–mass spectrometry with a detection limit of 0.005 to 0.01 mg/L.

Concurrent with the present study, characterization of P solids that were accumulated in the vadose zone was undertaken at three of the sites (Cambridge, Langton, and Muskoka). Profiles of sediment P content were determined by acid leaching and the identified P solids were characterized using back-scattered electron imaging techniques and microprobe compositional analyses (Zanini et al. 1998).

At four of the sites (Cambridge, Long Point 1, Langton, and Muskoka), bromide tracer tests were used to determine the residence time of the effluent in the vadose zone and to establish ground water velocity. The tests were conducted by injecting 1 to 4 kg of NaBr into the plumbing systems and then monitoring $Br^-$ concentrations in the ground water zone. At the Muskoka, Long Point 2, and Killarney sites, ground water velocities were measured by monitoring breakthrough of conservative constituents as the frontal parts of the plumes advanced. At the remainder of the sites, velocities were estimated using the Darcy equation in conjunction with the observed hydraulic gradients and hydraulic conductivity values estimated from grain size analyses and piezometer response tests.

Data acquisition methods at most of the sites have been described in more detail in previous reports (see following section).

## Results

All of the study sites are located in the Province of Ontario and most have been described in detail in previous publications as follows: Paradise (Alpay 1993), Cambridge (Robertson et al. 1991; Robertson 1995; Wilhelm et al. 1996), Camp Henry (Ptacek 1998), Long Point 2 (Aravena and Robertson 1998), Long Point 1 (Robertson and Cherry 1992), Langton (Harman et al. 1996), Harp 1 (Wood 1993; site A1), Muskoka (Robertson et al. 1991), and Killarney (Robertson and Blowes 1995). Table 1 summarizes the important features of each site including the waste water source, septic system age, sediment type, ground water velocity, and the characteristics of the phosphate plumes including plume lengths, phosphate concentrations, and degree of phosphate retardation (R) relative to the ground water velocity. The plume $PO_4$ concentrations presented in Table 1 are recent average values from the proximal core zones (immediately below the tile bed or within a few meters downgradient). Based on the concentrations of the more conservative constituents ($Na^+$, $Cl^-$), these plume zones are considered to be unaffected by dilution, unless otherwise noted in Table 2. The calculated average values were generally derived from the most recent data available at each site. At most of the sites, the phosphate plume lengths are arbitrarily defined and include all points exceeding the 0.1 mg/L P concentration level. Although this value represents only 1 to 2% of the average effluent value, it remains more than an order of magnitude higher than background phosphate concentrations in most of these ground waters (<0.01 mg/L P) and thus normally provides a good indication of septic system impact.

**Table 1**
**Ten Septic System Plumes in Central Canada of Sufficient Age to Develop P Zones**

| Site | Septic System | | | P Plume | | | | Sediment Type |
|---|---|---|---|---|---|---|---|---|
| | Age (a) | Source | G.W. Vel. (m/a) | $PO_4$-P[a] (mg/L) | Length[b] (m) | R[c] | pH | |
| **Oxidizing-Calcareous** | | | | | | | | |
| Paradise | 25 | house | 0.4 | 0.3 | 0.3 | 30 | 8.0 | silt |
| Cambridge | 20 | house | 20 | 4.9 | 25 | 20 | 7.0 | m. sand |
| Camp Henry | 16 | campground | 10–60 | 1.1 | 15 | 20 | 7.0 | c. sand |
| Long Point 2 | 6 | campground | 40 | 4.8 | >10 | <20 | 6.9 | m. sand |
| Long Point 1 | 18 | campground | 40 | 2.8 | 6–16 | 100 | 6.6 | m. sand |
| Langton | 44 | school | 100 | 1.3 | 70 | 40 | 6.6 | m. sand |
| **Oxidizing – Noncalcareous** | | | | | | | | |
| Harp 1 | 29 | cottage | 400 | 0.03 | 15 | – | 5.9 | sand till |
| Muskoka | 10 | house | 20 | 0.05 | 3 | 50 | 4.5 | f.-c. sand |
| **Reducing – Noncalcareous** | | | | | | | | |
| Delawana | 10 | resort | 70 | 0.61 | >25 | <40 | 6.6 | m. sand |
| Killarney | 10 | cottage | 3 | 0.95 | 1 | >50 | 6.1 | silt, f. sand |

[a] Average of recent data from the proximal plume core.
[b] Plume front defined by 0.1 mg/L P concentration level.
[c] Phosphate retardation factor (R) = ground water velocity/phosphate velocity

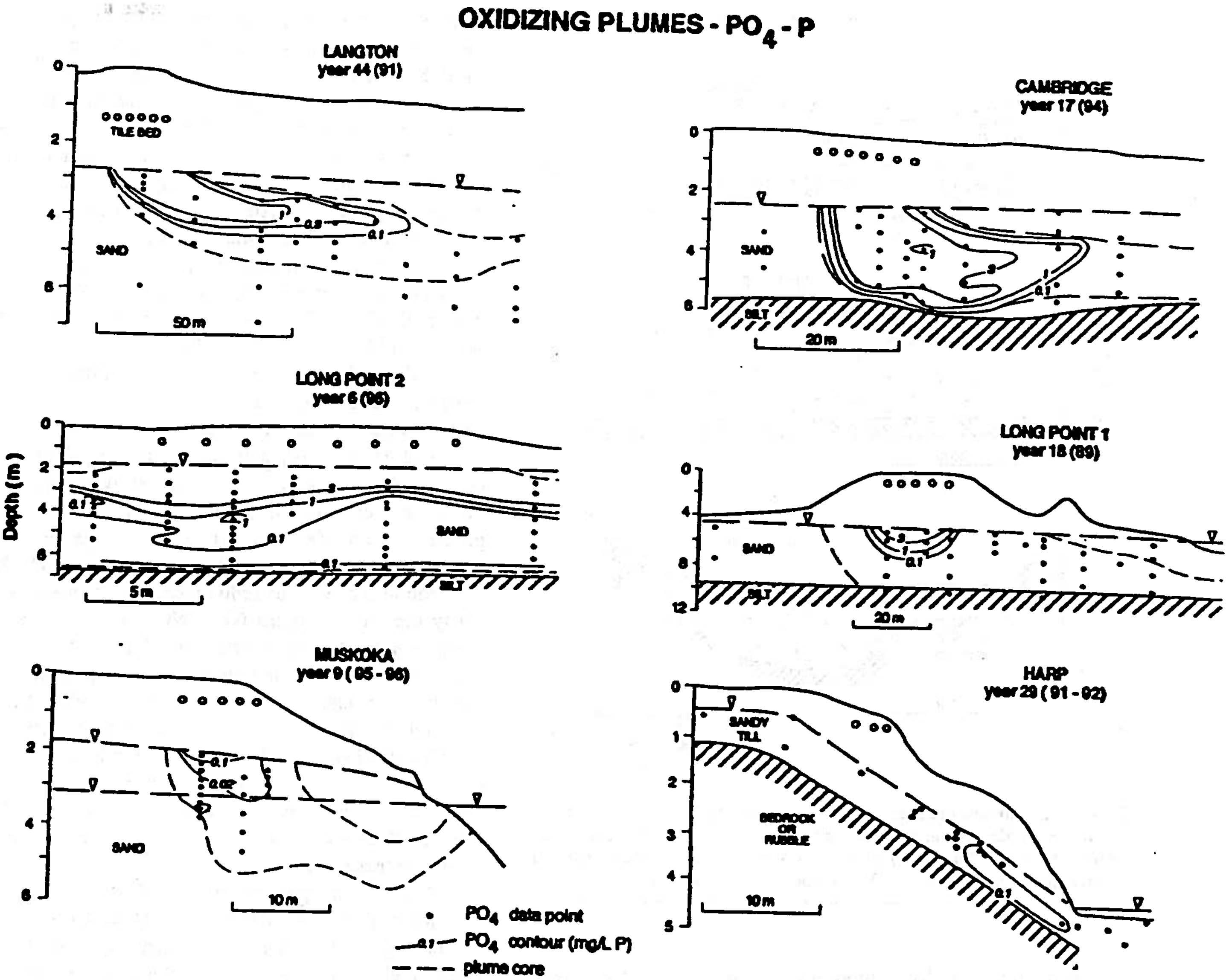


**Figure 1. Phosphate zones in six oxidizing septic system plumes in central Canada. Dashed lines denote plume boundaries defined by mobile solutes ($Cl^-$, $Na^+$, $NO_3^-$). At Muskoka both average and seasonal high water table positions are shown.**

**Table 2**
**Phosphate Attenuation in the Vadose Zone; Comparison of Effluent and Proximal Plume Core $PO_4$ Concentrations**

| Site | $PO_4$-P (mg/L) Effluent[a] | n[b] | Plume[c] | n | C/Co | Plume pH |
|---|---|---|---|---|---|---|
| **Oxidizing—Calcareous** | | | | | | |
| Cambridge | 6.4 | 21 | 4.9 | 26 | 0.77 | 7.0 |
| Camp Henry | 11.8 | 1 | 1.1 | 9 | 0.09 | 7.0 |
| Long Point 2 | 7.1 | 1 | 4.8 | 16 | 0.68 | 6.9 |
| Long Point 1 | 6.2 | 12 | 2.8 | 13 | 0.45 | 6.6 |
| Langton | 8.2 | 6 | 1.3 | 10 | 0.16 | 6.6 |
| **Oxidizing—Noncalcareous** | | | | | | |
| Delawana | 1.2 | 3 | 0.30 | 15 | 0.25 | 6.3 |
| Harp 1 | 8.9 | 2 | 0.03 | 3 | 0.003[d] | 5.9 |
| Muskoka | 12.1 | 10 | 0.05 | 27 | 0.004 | 4.5 |
| **Reducing** | | | | | | |
| Camp Henry | 11.8 | 1 | 0.19 | 7 | 0.02 | 7.2 |
| Delawana | 1.2 | 3 | 0.61 | 10 | 0.51 | 6.6 |
| Killarney | 7.5 | 8 | 0.95 | 12 | 0.13[d] | 6.1 |

[a] Average effluent $PO_4$ concentration except total dissolved phosphorus at Harp and Henry sites.
[b] n = number of analyses
[c] Average of recent data
[d] Conservative constituents ($Na^+$, $Cl^-$) indicate that proximal plume core is affected by dilution at these sites (see Tables 3 and 4)

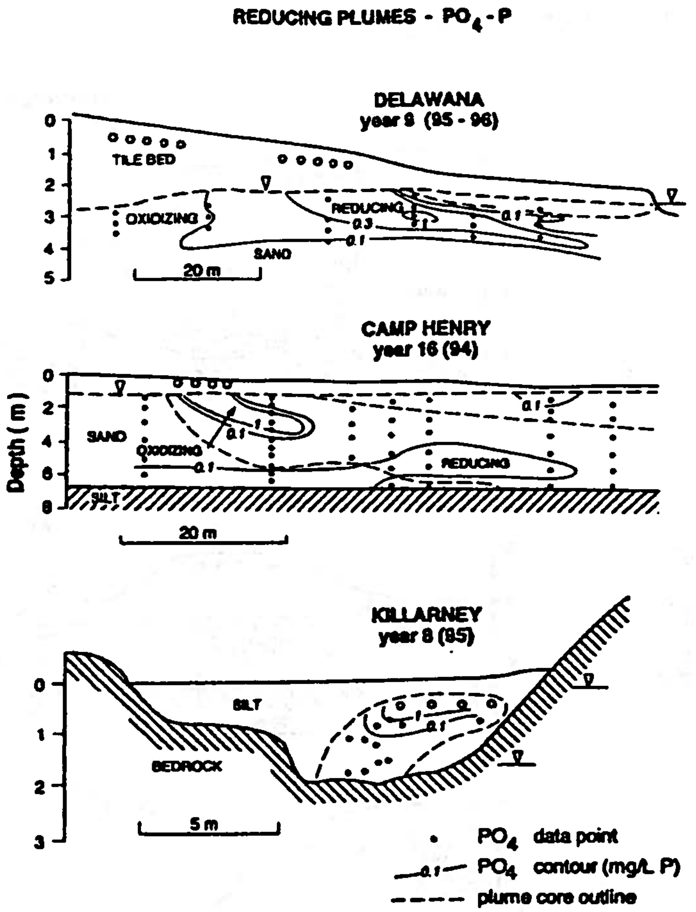


**Figure 2. Phosphate zones in three reducing septic system plumes in central Canada. (Note: At Delawana and Camp Henry sites, reducing zones are characterized by Eh <200 mV and are $NH_4^+$ rich; adjacent oxidizing zones have Eh > 300 mv and are $NO_3^-$ rich).**

Figure 1 shows the distribution of phosphate at the six oxidizing plume sites where mappable phosphate zones are present (Cambridge, Langton, Harp, Muskoka, Long Point 1, and Long Point 2). Figure 2 shows the three reducing plumes where mappable phosphate zones are present (Camp Henry, Delawana, and Killarney). At two of the latter sites (Camp Henry and Delawana) oxidizing zones are also present as noted (Figure 2). The sections shown in Figures 1 and 2 follow the plume centerlines as established from the distribution of the conservative plume solutes. The Paradise site is not included in Figures 1 or 2 because above background P concentrations at this site were limited to a single data point located within 0.3 m of the infiltration pipe; thus a mappable P zone was not present.

Because of the mature age of these septic systems (6 to 44 years; Table 1), it is likely, for the sites located on sands, that most of the sediment phosphate sorption capacity has already been consumed in the near-tile area. This suspicion was confirmed at several of the sites (Cambridge, Langton) where long-term monitoring over periods of four to nine years revealed that proximal plume core P concentrations remained steady and showed a consistent degree of attenuation compared to effluent values (Robertson 1995). If the attenuation was only the result of simple adsorption, concentrations should slowly increase over time as sorption sites are gradually consumed, which was not the case. At the Long Point 2 tile bed, high phosphate concentrations (~ 3 mg/L P) reached the water table at a 2 m depth after only two months of operation, providing addition evidence of the relatively rapid use of vadose zone sorption capac ity at these sites compared to their long histories of usage. Thu observed differences in phosphate concentrations between th effluent and the shallow water table zones, after several years c operation, are considered to represent attenuation by processe other than fast reversible adsorption, presumably solid phase fo mation reactions (irreversible adsorption and/or mineral precipita tion) that are occurring in the vadose zone. Table 2 compares di ferences in effluent and proximal plume core phosphat concentrations. Normalized plume concentrations range from C/C values of 0.77 at Cambridge (6.4 vs. 4.9 mg/L P) to 0.004 a Muskoka (12.1 vs. 0.05 mg /L P).

Table 1 distinguishes oxidizing (Cambridge, Long Poin Langton, Muskoka, and Harp) and reducing (Delawana an Killarney) sites with the indicated redox condition pertainin specifically to the phosphate zones. Oxidizing plumes are identifie either by the presence of detectable dissolved oxygen (DO) (> 0. mg/L) or by elevated Eh values (> 200 mV), while in the reducin plume zones no detectable dissolved oxygen is present (< 0. mg/L) and /or Eh values are depressed (<200 mV). Other param eters can also be used as redox indicators in these plumes, partic ularly the redox couples $NH_4^+$-$NO_3^-$ and $Fe^{2+}$-$Fe^{3+}$. Oxidizin conditions facilitate the conversion of $NH_4^+$ to $NO_3^-$ and cause iro to be present in the ferric oxidation state ($Fe^{3+}$). At pH ranges of to 9, $Fe^{3+}$ concentrations are limited to low values (<0.1 mg/L) b the solubility of ferric oxyhydroxide minerals such as ferrihydrit ($Fe(OH)_3$), whereas $Fe^{2+}$ iron can have substantially higher solu bility. Fe >0.1 mg/L is indicative of iron occurring as $Fe^{2+}$ in mo of these water samples. Thus, the presence of >2 mg/L $NH_4^+$-N c > 0.1 mg/L Fe were found to be good indicators of reducing co ditions in these plumes.

Table 3 lists representative field DO values for the oxidizin sites where data are available. Although the Delawana site is liste as reducing in Table 2, this site also contains zones that are oxidizin in areas where the vadose zone is thicker (2 to 3 m). An oxidizin zone, characterized by Eh values > 300 mV, is identified on Figu 2 and is in contrast to the reducing zone where Eh values are low (< 200 mV). The Long Point 1 plume also contains a reducing zon but at a location farther downgradient. This zone is characteriz by an absence of detectable dissolved oxygen (<0.1 mg/L) and t abrupt attenuation of nitrate due to denitrification (Robertson an Cherry 1992). However, this zone does not overlap with the pro imal phosphate zone which remains aerobic throughout (DO=0 to 3 mg/L). Thus, the Long Point 1 site is considered oxidizing f the purposes of this discussion.

At the Camp Henry site, a phosphate-rich plume core zone to 3 mg/L P) extends about 10 m downgradient from the tile b (Figure 2), but then P concentrations decline abruptly to valu below detection (<0.01 mg/L) farther downgradient. A seco zone is present, however, at the bottom of the aquifer, where mo erate P concentrations (0.1 to 0.8 mg/L) occur, but where conditio are more reducing (Eh < 200 mV) than in the shallower aquifer. also has enriched Fe concentrations (2 to 15 mg/L). This reduc zone may be an artifact of an earlier period of heavier loading higher water table conditions at the site, or it may be derived fro other sewage sources that were released nearby in the pa Alternatively, it may occur naturally as a consequence of the re tively high organic carbon content of the nearby wetland sed ments or underlying clay unit (Ptacek 1998).

## Table 3

**Oxidizing Plumes; Comparison of Effluent (EFF) and Proximal Plume Chemistry**

Presented are recent data considered representative of average conditions. Mineral saturation indices (SI) determined using the chemical speciation model PHREEQE (Parkhurst et al. 1985). Sampling point numbers are piezometer numbers.

| Sample Point | Delawana | | Camp Henry | | Long Point 2 | | Langton | | Cambridge | | Muskoka | | Paradise 2 | | Harp - 1 | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | EFF | 4-1 | EFF | 7-4 | EFF | 101A-10 | EFF | 31-1 | EFF | 33-1 | EFF | 32-3 | EFF | PL2-2 | EFF | 35 |
| x (m)[a] | 0 | 2 | 0 | 7 | 0 | 2 | 0 | 2 | 0 | 2 | 0 | 3 | 0 | 1.3 | 0 | 8 |
| Date | 10/95 | 11/95 | 5/95 | 6/94 | 10/95 | 10/95 | 10/91 | 10/91 | 10/95 | 10/95 | 5/95 | 5/95 | 8/93 | 10/92 | 91-92 | 5/92 |
| Na (mg/L) | 31 | 29 | 43 | 11 | 23 | 27 | 114 | 101 | 75 | 71 | 54 | 51 | 55 | 30 | 35 | 9 |
| K | 7 | 3 | 21 | 9 | 16 | 17 | 49 | 62 | 7 | 8 | 12 | 15 | 24 | 3 | 22 | 1 |
| Ca | 36 | 72 | 84 | 106 | 112 | 199 | 12 | 249 | 33 | 74 | 10 | 34 | 37 | 110 | 10 | 10 |
| Mg | 6 | 7 | 13 | 9 | 17 | 20 | 26 | 30 | 15 | 13 | 3 | 4 | 12 | 32 | 4 | 3 |
| AIK(as $CaCO_3$) | 147 | 150 | | 520 | 350 | 230 | 452 | 210 | 261 | 270 | 202 | <1 | 402 | 401 | 310 | 14 |
| $SO_4$ | 35 | 35 | 34 | 47 | 38 | 44 | 44 | 66 | 42 | 45 | 17 | 47 | 11 | 23 | 1 | 21 |
| Cl | 24 | 24 | 57 | 112 | 42 | 57 | 228 | 209 | 21 | 26 | 41 | 36 | 53 | 20 | 50 | 11 |
| $SiO_2$ | 5 | 10 | 10 | | | | 8 | 57 | 21 | 20 | 15 | 20 | 10 | 10 | | |
| $PO_4$-P | 1.8 | 0.04 | 11.8[b] | 1.4 | 7.1 | 4.2 | 8.5 | 1.5 | 7.7 | 4.6 | 6.7 | 0.08 | 7.6 | <0.01 | 10.3[b] | 0.03[b] |
| $NO_3$-N | 0.05 | 13 | 0.05 | 18 | 0.6 | 69 | <0.05 | 131 | 0.9 | 14 | <0.05 | 37 | <0.05 | <0.05 | 0.1 | 3.2 |
| $NH_4$-N | 17 | 0.2 | 98 | 0.4 | 57 | <0.1 | 135 | 0.06 | 19 | <0.05 | 34 | 0.05 | 58 | 0.1 | 69 | 0.08 |
| Fe | 1.2 | 0.03 | 0.60 | 0.004 | 0.12 | <0.02 | 0.2 | <0.02 | 0.12 | 0.02 | 0.14 | 0.03 | 0.37 | <0.02 | 25 | 0.11 |
| Mn | 0.06 | 0.11 | 0.48 | 1.0 | | | 0.14 | 2.2 | 0.05 | <0.01 | 0.02 | 0.37 | 0.11 | 0.36 | | |
| Al | <0.10 | 0.16 | 0.10 | <0.01 | | | | | 0.047 | 0.01 | 0.066 | 3.5 | | | 0.18 | |
| DOC | 9 | 5 | 32 | 13 | | | 28 | 0.7 | 40 | 4 | 81 | 3 | 26 | 5 | 65 | 4 |
| DO | | | | | | 0.2 | | 0.3 | | 8 | | 3 | | | | |
| pe[d] | 0.0 | 10 | 0.0 | 10 | 0.0 | 10 | 0.0 | 10 | 0.0 | 10 | 0.0 | 10 | 0.0 | 10 | 0.0 | 10 |
| pH | 7.15 | 6.25 | | 7.00 | 7.5 | 6.9 | 7.35 | 6.55 | 7.1 | 6.9 | 6.57 | 4.3 | 6.9 | 7.75 | 7.5[c] | 5.9 |
| Calcite SI | −0.6 | −1.2 | | −0.3 | 0.4 | −0.1 | 0.4 | −0.4 | −0.5 | −0.4 | −1.7 | −5.7 | −0.5 | 0.7 | 0.0 | −3.5 |
| Siderite | −0.3 | −6.1 | | −6.3 | −0.8 | <−6.9 | −0.6 | −6.6 | −1.2 | −6.8 | −1.8 | −11 | −0.7 | <−7.8 | 1.6 | −6.2 |
| $FeOH_3\alpha$ | −1.6 | 0.7 | | 0.8 | −1.7 | <1.1 | −1.9 | 0.8 | −2.8 | 1.1 | −4.4 | −1.3 | −2.9 | <1.7 | 0.7 | 0.9 |
| Gibbsite | <1.1 | 1.4 | | | | | | | 0.8 | −0.1 | 0.6 | −1.0 | | | 1.4 | |
| Jurbanite[e] | <−4.0 | −2.1 | | | | | | | −4.1 | −4.6 | −3.5 | −0.2 | | | −6.0 | |
| Hydroxyapatite[f] | 0.8 | −7.9 | | 0.2 | 5.6 | 3.4 | 5.5 | 0.3 | 2.0 | 1.4 | −4.0 | −22 | 1.1 | −1.9 | 4.5 | −15 |
| Vivianite[g] | 1.4 | −20 | | −18 | −0.1 | <18 | 0.2 | −18 | −0.8 | −15 | −2.2 | −30 | 0.1 | <−27 | 7.2 | −17 |
| Strengite[h] | −2.2 | −0.3 | | <1.0 | −2.5 | <1.1 | −2.3 | <0.8 | −2.7 | 1.3 | −3.6 | 0 | −2.5 | −2.3 | 0.2 | 0.1 |
| Variscite[i] | <0.0 | −0.3 | | | | | | | 0.4 | −0.3 | 1.2 | −0.1 | | | 0.3 | |

[a] Flowpath length between infiltration pipes and sampling point.
[b] Total dissolved P.
[c] Lab measurement.
[d] Assumed values.
[e] Jurbanite, Al OH $SO_4 \rightarrow Al^{3+} + OH^- + SO_4^{2-}$, log $K_{sp} = -17.8$ (Appelo and Postma 1994)
[f] Hydroxyapatite, $Ca_5(PO_4)_3OH + 4H^+ \rightarrow 5\ Ca^{2+} + 3HPO_4^{2-} + H_2O$, log $K_{sp} = -3.421$ (PHREEQE database)
[g] Vivianite, $Fe_3(PO_4)_2 \bullet 8H_2O \rightarrow 3Fe^{2+} + 2PO_4^{3-} + 8H_2O$, log $K_{sp} = -36.0$ (PHREEQE database)
[h] Strengite, $FePO_4 \bullet 2H_2O \rightarrow Fe^{3+} + PO_4^{3-} + 2H_2O$, log Ksp = −26 (Stumm and Morgan 1981)
[i] Variscite, $AlPO_4 \bullet 2H_2O \rightarrow Al^{3+} + PO_4^{3-} + 2H_2O$, log Ksp = −21 (Stumm and Morgan 1981)

increases in iron concentrations in solutions already at equilibrium with vivianite (i.e., the effluent) would be expected to cause additional precipitation of vivianite. Thus, the decreases in phosphate concentrations observed in all three reducing plumes ($C/C_0$ = 0.02 to 0.51, Table 2) may be attributable to additional vivianite precipitation. Nriagu and Dell (1974) identified authogenic vivianite grains in anaerobic Lake Erie sediments and established that the sediment pore water, containing Fe of 1 to 8 mg/L and $PO_4$-P of 0.03 to 0.3 mg/L was near equilibrium with respect to vivianite. Because all three reducing plumes have Fe (1 to 20 mg/L) and $PO_4$-P concentrations (0.1 to 3 mg/L) as high or higher than those in the Lake Erie sediments and have the same pH range (6.5 to 7.5), it is reasonable to expect that vivianite also limits phosphate concentrations in reducing septic system plumes. Other workers also implicate vivianite as the main control on phosphate in reducing lake sediments (e.g., Emerson and Widmer 1978; Wersin et al. 1991). A more detailed geochemical modeling assessment of vivianite equilibrium at the Camp Henry site is presented by Ptacek (1998).

The highest phosphate concentrations (4 to 5 mg/L P) occur in the plumes that are oxidizing and are on calcareous sand at near neutral pH (Cambridge, Long Point 2; Tables 1 to 3; Figure 5a). A common feature of both plumes is that they exhibit a high degree of supersaturation with respect to hydroxyapatite (SI 1.4 to 3.4; Table 3) and the degree of supersaturation persists along the flowpaths. Jenkins et al. (1971) suggest that the solubility of freshly precipitated amorphous hydroxyapatite and impure phases may be several orders of magnitude higher than that of the crystalline form which is used in the PHREEQE database ($K_{sp}$ = -3.4 at 25°C). Alternatively, hydroxyapatite precipitation might be kinetically limited, but Zanini et al. (1998) found little evidence of secondary solids containing Ca and P at these sites. Moore and Reddy (1994) suggest that a more soluble calcium phosphate mineral, beta tricalcium phosphate ($\beta$-$Ca_3(PO_4)_2$), may control phosphate concentrations in Lake Okeechobee, Florida, sediments. The field data thus suggest that crystalline hydroxyapatite or other Ca and P solids are relatively ineffective at controlling phosphate concentrations in these plumes.

The solubility curves for variscite and strengite shown on Figure 5 imply that low phosphate concentrations should occur in acidic plumes that may be present in noncalcareous terrain. In addition, adsorption of $PO_4$ onto oxide surfaces is enhanced at lower pH (Goldberg and Sposito 1984). The strengite solubility curve includes the assumption that in oxidizing environments, ferric iron concentrations are controlled by the solubility of amorphous $Fe(OH)_3$ (e.g., Whittemore and Langmuir 1975) and for variscite, that Al concentrations are primarily controlled by the solubility of gibbsite ($Al(OH)_3$) (van Grinsven et al. 1992; Appelo and Postma 1994). However, at very acidic values, it has been suggested that aluminum concentrations in some waters will be controlled by the solubility of the mineral jurbanite ($Al\,OH\,SO_4 \cdot 5H_2O$) (Nordstrom 1982; Appelo and Postma 1994). With respect to the Muskoka plume water chemistry (Table 3), chemical equilibrium modeling suggests that jurbanite rather than gibbsite controls Al concentrations at pH values <4.9, as is indicated in Figure 4b. Thus, the variscite solubility curve (Figure 5) assumes Al concentrations are controlled by gibbsite equilibrium at pH >4.9 but by jurbanite at lower pH values. A comparison of plume Al concentrations and the gibbsite/jurbanite solubility curve (Figure 4b) suggests that these assumptions may be reasonable for both reducing and oxidizing plumes. Substantially elevated Al values (0.2 to 5 mg/L) are

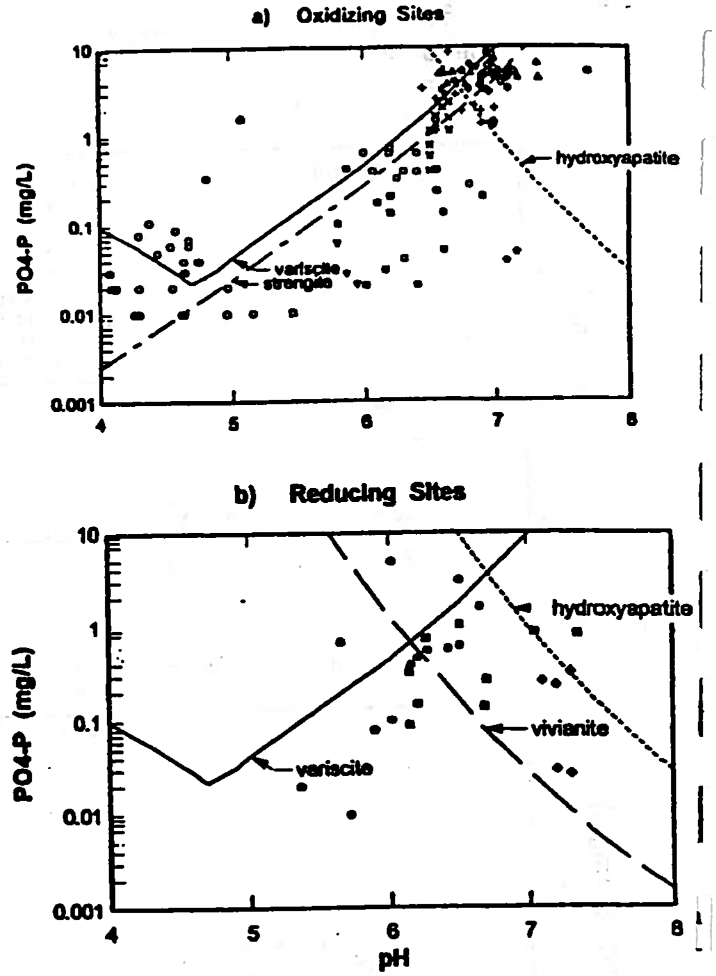


**Figure 5. Phosphate concentrations in proximal plume core zones compared to solubility curves for hydroxyapatite, strengite, variscite and vivianite for (a) oxidizing and (b) reducing plumes. Included are recent data from proximal plume core zones. Hydroxyapatite curve calculated using Cambridge plume water composition ($Ca^{2+}$ = 74 mg/L); vivianite curve calculated using Delawana reducing plume water composition (Fe = 10 mg/L); strengite curve calculated using Cambridge plume water composition assuming equilibrium with amorphous $Fe(OH)_3$; variscite curve calculated using Muskoka plume water composition assuming equilibrium with gibbsite at pH > 4.9 and jurbanite at pH < 4.9 (re = reducing zone, ox = oxidizing zone, c = calcareous site, nc = noncalcareous site).**

observed as pH drops below 5.5. For $Fe^{3+}$, however, the assumption of concentrations limited by amorphous $Fe(OH)_3$ solubility appears less valid in these plumes. At the Muskoka site, for example, low pH plume water is substantially undersaturated with respect to amorphous $Fe(OH)_3$ (Figure 4c). Fe at this site is perhaps limited by some other less soluble mineral such as goethite. In general, the data are consistent with $PO_4$ concentrations controlled by variscite or strengite solubility, including the important result that lower phosphate concentrations occur at lower pH values.

Few data points substantially exceed variscite solubility. Two data points from Muskoka that do exceed the variscite solubility curve (1.55 and 0.34 mg/L $PO_4$-P; Figure 5a) were collected in 1997 from the same sampling point (piezometer 32-1), which is the shallowest point in the monitoring network (1 m below the infiltration pipes). The first sample (1.55 mg/L $PO_4$-P) was obtained at the end of the spring snow melt period (April 22), while the second sample was retrieved one month later (May 20). The much lower P value in the second sample (0.34 mg/L) suggested that the pore water at this location was not at a condition of equilibrium with respect to phosphate during this period, possibly because of the close proximity of the sampling point to the infiltration pipes and because of the rapid infiltration rates that normally accompany snow melt.

## Implications

If the minerals indicated in Figure 5 (strengite, variscite, hydroxyapatite, and vivianite) are valid controls on phosphate in septic system plumes, then it can be seen that the highest phosphate concentrations should be present in the neutral pH plumes that occur in calcareous terrain, whereas much lower concentrations should be present in noncalcareous terrain when pH values drop below about six. The hydroxyapatite control may be ineffective as a consequence of kinetic limitations or as result of the greater solubility of impure or amorphous phases of this mineral. Generally, the field data support a pH dependency and the highest phosphate concentrations (~ 5 mg/L P) occur in the two oxidizing plumes at pH ~ 7.0 (Cambridge, Long Point 2). The reducing plumes at neutral pH (Delawana, pH 6.6; Camp Henry, pH 7.2) have substantially lower phosphate concentrations (0.1 to 1 mg/L P), possibly as a result of the effectiveness of the vivianite control in reducing environments. Thus, at neutral to slightly alkaline pH values, oxidizing plumes may have higher phosphate concentrations than reducing plumes. Interestingly, this relationship is opposite to that normally reported in lake and marine sediments where higher pore water phosphate concentrations are generally associated with the deeper reducing zones compared to the more oxidizing shallower sediments (Nriagu and Dell 1974; Moore and Reddy 1994). In lake sediments this occurs because, at the redox boundary, oxidation of $Fe^{2+}$ to $Fe^{3+}$ causes precipitation of ferric oxyhydroxide minerals and the subsequent sorption of P onto fresh surfaces or the precipitation of sparingly soluble ferric phosphate minerals. It is likely that similar attenuation of septic system P occurs when reducing plume water, high in $Fe^{2+}$, discharges to oxidizing surface water environments.

This aerobic/anaerobic relationship is reversed at lower pH (Figure 5), because at pH values below about six the vivianite control is less effective unless Fe concentrations are high (>10 mg/L). However, the occurrence of acidic (pH<6) reducing plumes is unlikely because septic system effluent normally has high alkalinity (150 to 500 mg/L as $CaCO_3$; Tables 3 and 4) and near neutral pH (6.5 to 7.5; Figure 3) so that substantial oxidation of the effluent is necessary to effect a decrease in pH. This is evident at the Delawana site where pH in the reducing plume zone (6.6) is higher than in the oxidizing zone (6.3; Table 2) and has also been noted at a sewage plume on Cape Cod, Massachusetts (Walter et al. 1996), where pH in the anaerobic zone (~6.5) is substantially higher than in the aerobic zones and in the background ground water (~5.7).

In general, the field data indicate that P is "buffered" to moderate concentrations in reducing zones, whereas in oxidizing zones greater contrasts in P concentrations are exhibited. Patrick and Khalid (1974) reported similar behavior in laboratory batch tests in which P was added incrementally to a set of four soils (pH range 5.4 to 6.8) under both aerobic and anaerobic conditions. In their study, soil solution P values in the untreated soils were initially higher under anaerobic conditions, but as P was added, solution P values eventually became substantially higher under aerobic conditions.

**Table 5**
**Alkalinity of Septic Tank Effluent Compared to Alkalinity Consumption Expected from Oxidation of Effluent $NH_4^+$**
**(Data from Tables 3 and 4)**

| | Effluent Alkalinity (meq/L) | Alkalinity Consumed by $NH_4^+$ Oxidation (meq/L) |
|---|---|---|
| Long Point 1 | 9.8 | 9.7 |
| Langton | 9.0 | 19 |
| Paradise | 8.0 | 8.3 |
| Killarney | 7.2 | 16 |
| Long Point 2 | 7.0 | 8.1 |
| Harp 1 | 6.2 | 9.9 |
| Cambridge | 5.2 | 2.7 |
| Muskoka | 4.0 | 4.9 |
| Delawana | 2.9 | 2.4 |

Consideration of these mineral solubility controls can be useful in the development of alternative septic system designs for the improved attenuation of phosphorus. For example, the possibility of achieving lower phosphate concentrations at more acidic pH values invites design modifications that promote zones of decreased pH, provided that such zones do not adversely affect other treatment reactions. This can be done by recognizing that properly functioning septic systems normally generate acidity by oxidation reactions such as the nitrification of ammonium;

$$NH_4^+ + 2O_2 \rightarrow NO_3^- + 2H^+ + H_2O \qquad (1)$$

Using acid-base accounting in a manner similar to that described by Wilhelm et al. (1996), it can be seen from Table 5 that oxidation of effluent $NH_4^+$ is, by itself, sufficient to consume most or all of the alkalinity available in most of these effluents. Thus, it is likely that the thorough oxidation of most septic tank effluent will result in relatively complete consumption of available alkalinity and will cause the development of acidic conditions. At most sites, however, low pH conditions do not develop as a consequence of the buffering reactions that occur in the sediments, particularly carbonate mineral dissolution reactions. Only at the Harp and Muskoka sites, where sediment buffering capacity is limited, do pH values decline below six (Table 1). When constructing infiltration beds it should be possible, however, to specifically engineer localized low pH zones simply by using noncalcareous fill material. This may be particularly feasible when constructing raised filter beds (e.g., Converse and Tyler 1990) in which case import of fill material is normally required. Other simple mitigation measures follow. For example, at sites where phosphate loading is the primary concern, septic systems should be designed to allow for maximum oxidation if the sediments have limited buffering capacity, but in calcareous terrain it may be more prudent to provide a lesser degree of oxidation so that anaerobic, Fe-rich plumes develop, although moderate P concentrations (~1 mg/L) would likely still persist in the latter case.

## Summary

Review of phosphate distribution in 10 mature septic system plumes reveals that in six cases relatively large phosphate plumes are present (>10 m in length) where $PO_4$ concentrations (0.5 to 5 mg/L P) are about two orders of magnitude higher than normally found in uncontaminated aquatic ecosystems. At the other four sites, which are those on noncalcareous and silt- and clay-rich

sediments, high phosphate concentrations only occur within 3 m of the infiltration pipes. Although phosphate plume migration velocities are substantially retarded compared to the ground water velocities at all of the sites (R = 20 to 100), the phosphate migration velocities at the sites on calcareous sands remain sufficiently fast (~1 m/a) to be of concern with respect to the normal minimum setback distance of septic systems from surface water bodies.

Although phosphate is persistent and moderately mobile in a number of these plumes, phosphate concentrations at all sites are attenuated compared to effluent values by amounts ranging from 23 to 99%. Observed plume concentrations are generally consistent with the constraints imposed by the solubilities of the minerals vivianite, strengite, and variscite, but not hydroxyapatite. Plumes at near-neutral pH consistently exhibit a high degree of hydroxyapatite supersaturation possibly as a consequence of kinetic limitations or as a result of amorphous and impure forms of the mineral being more soluble than crystalline forms. In general, in reducing plumes, pH values appear to be maintained at near neutral values similar to that of the effluent and phosphate concentrations appear to be buffered to the ~ 1 mg/L P range, consistent with vivianite equilibrium. This represents a moderate amount of P attenuation compared to effluent concentrations. In oxidizing plumes pH values vary considerably, depending on sediment buffering capacity, with the result that phosphate concentrations also vary. In calcareous terrain, $PO_4$ concentrations (2 to 5 mg/L P) exhibit only minor attenuation compared to effluent concentrations, whereas in noncalcareous terrain almost complete attenuation of phosphate appears possible if acidic conditions develop.

The dominant mineral precipitation reactions appear to occur rapidly after the effluent enters the subsurface and are largely completed within the vadose zones at these sites. This results in the accumulation of phosphorus-rich solids within narrow depth intervals in close proximity to the infiltration pipes. At the Langton site, approximately 85% of the total sewage P remains retained within the 2 m thick vadose zone, even after 44 years of effluent loading.

Recognition of the constraints imposed by the solubilities of the secondary phosphate minerals opens up the possibility of modifying septic system design to achieve improved phosphate attenuation.

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**ATTACHMENT D – ENHANCED ATTENUATION OF SEPTIC SYSTEM PHOSPHATE IN NONCALCAREOUS SEDIMENTS (W.D. ROBERTSON, 2003)**

ground water

# Enhanced Attenuation of Septic System Phosphate in Noncalcareous Sediments

by W.D. Robertson[1]


## Abstract/

Review of phosphate behavior in four mature septic system plumes on similar textured sand has revealed a strong correlation between carbonate mineral content and phosphate concentrations. A plume on calcareous sand (Cambridge site, 27 wt % $CaCO_3$ equiv.) has proximal zone $PO_4$ concentrations (4.8 mg/L P average) that are about 75% of the septic tank effluent value, whereas three plumes on noncalcareous sand (Muskoka, L. Joseph, and Nobel sites, <1 wt % $CaCO_3$ equiv.) have proximal zone phosphate concentrations (<0.1 mg/L P) that are consistently less than 2% of the effluent values. Phosphate attenuation at the noncalcareous sites appears to be an indirect result of the development of acidic conditions (site average pH 3.5 to 5.9) and elevated Al concentrations (up to 24 mg/L), which subsequently causes the precipitation of Al-P minerals such as variscite ($AlPO_4 \cdot 2H_2O$). This is supported by scanning electron microscope analyses, which show the widespread occurrence of (Al+P)—rich secondary mineral coatings on sand grains below the infiltration beds. All of these septic systems are more than 10 years old, indicating that these attenuation reactions have substantial longevity.

A field lysimeter experiment demonstrated that this reaction sequence can be readily incorporated into engineered waste water treatment systems. We feel this important P removal mechanism has not been adequately recognized, particularly for its potential significance in reducing P loading from septic systems in lakeshore environments.


## Introduction

Phosphorus concentrations in domestic waste water (~5 to 20 mg/L) are several orders of magnitude greater than the values that are considered capable of stimulating algal growth and eutrophication in aquatic environments (~0.03 mg/L; Dillon and Rigler 1974). The role of septic systems in lakeshore phosphorus loading has remained a topic of considerable debate because of the substantial P mass that this source potentially represents and the uncertainty regarding the mobility of this constituent in subsurface transport pathways. In a recent review of $PO_4$ behavior in 10 well-characterized septic system plumes in Ontario (Robertson et al. 1998), we observed considerable variability in P mobility even between sites on similarly textured sand. Six sites on sand had distinct $PO_4$ plumes extending more than 10 m in length, but two other sites (Muskoka and Harp sites) exhibited virtually no P plume development. The present study was undertaken to better understand the cause of the dramatically greater P attenuation at the latter two sites. The aquifer sediments at the Muskoka and Harp sites are noncalcareous, and acidic conditions had developed in the plumes as a consequence of sewage oxidation reactions:

$$NH_4^+ + 2O_2 \rightarrow NO_3^- + 2H^+ + H_2O \qquad (1)$$

In the absence of carbonate mineral buffering capacity, it was suggested that the persistence of acidic conditions then caused gibbsite dissolution:

$$Al(OH)_3 + 3H^+ \rightarrow Al^{3+} + 3H_2O \qquad (2)$$

and subsequently P attenuation by the precipitation of the aluminum phosphate minerals such as variscite:

$$Al^{3+} + PO_4^{3-} + 2H_2O \rightarrow AlPO4 \cdot 2H_2O \qquad (3)$$

Supporting this was the occurrence of high Al concentrations (0.1 to 3.5 mg/L) in the acidic, P-deficient plumes and the presence of (Al+P)-rich secondary mineral coatings on the sand grains at the Muskoka site (Zanini et al. 1998). At sites where carbonate minerals are available, this reaction sequence is halted by the buffering provided by carbonate mineral dissolution:

$$CaCO_3 + H^+ \rightarrow HCO_3^- + Ca^{2+} \qquad (4)$$

We considered the evidence of enhanced P attenuation in noncalcareous terrain (plume P values of ~0.05 mg/L

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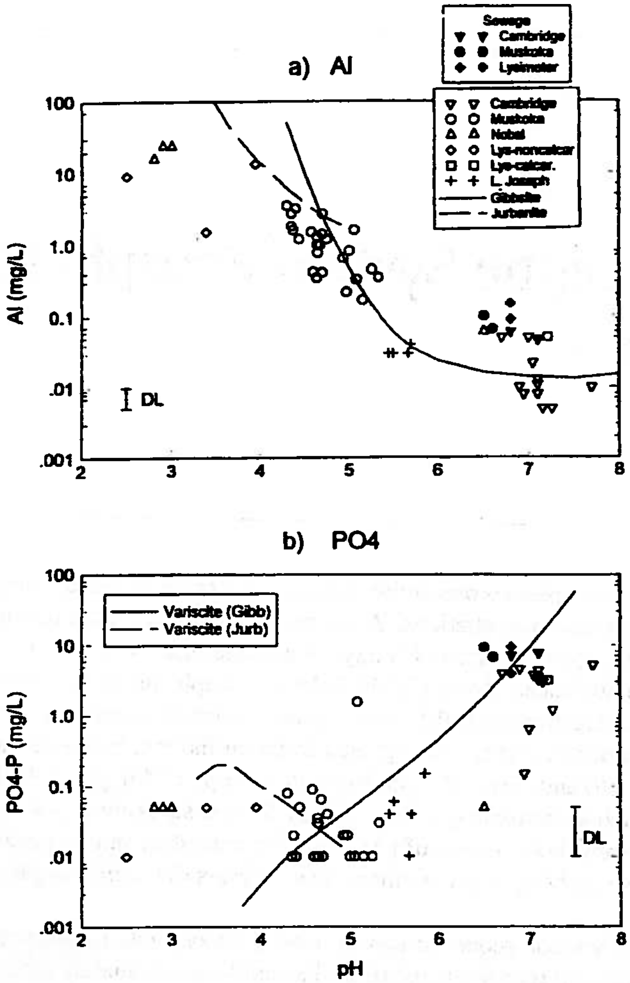


Figure 3. (a) Al and (b) $PO_4$ concentrations in the septic tank effluent, lysimeter effluent, and the four ground water plumes, compared to solubility of gibbsite, jurbanite, and variscite. Variscite curves assume equilibrium with gibbsite or jurbanite. Solubility curves calculated using Muskoka plume water composition given by Robertson et al. (1998) (Table 3) and the chemical equilibrium model PHREEQE (Parkhurst et al. 1985); gibbsite (microcrystalline), $Al(OH)_3 + 3H^+ \rightarrow Al^{3+} + 3H_2O$, $\log K_{sp} = 9.35$, PHREEQE database; jurbanite, $Al\,OH\,SO_4 \cdot 5H_2O \rightarrow Al^{3+} + OH^- + SO_4^{2-}$, $\log K_{sp} = -17.8$ (Nordstrom 1982); variscite, $AlPO_4 \cdot 2H_2O \rightarrow Al^{3+} + PO_4^{3-} + 2H_2O$, $\log K_{sp} = -21$ (Stumm and Morgan 1981).

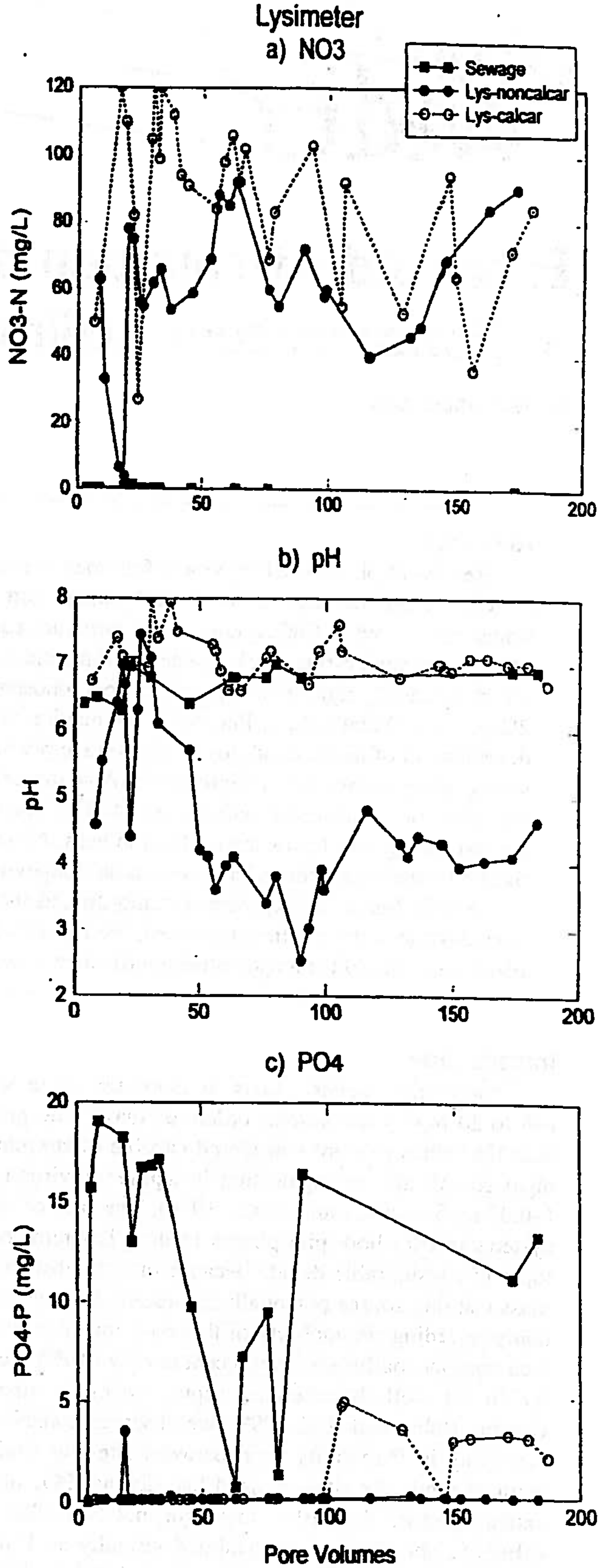


Figure 4. Breakthrough of (a) $NO_3$, (b) pH, and (c) $PO_4$ in the lysimeter effluent compared to the influent sewage.

~30 m downgradient from the tilebed at Cambridge, contrasting greatly with the other sites, where P plumes are absent (<0.05 mg/L, Nobel site), or are more localized and much lower in concentration (<0.1 mg/L, Muskoka and L. Joseph sites). Long-term monitoring of $PO_4$ breakthrough in the proximal piezometers at Cambridge and Muskoka (Figure 2) shows that $PO_4$ concentrations in the shallow water table zones below the tilebeds are not increasing over time, indicating that the P attenuation reactions are not diminishing and that they continue to occur predominantly within the vadose zone.

### pH, Alkalinity, and Cations

Although these plumes have similar concentrations of conservative constituents (Na, Cl) and nitrogen, their compositions contrast greatly with respect to $PO_4$, pH, Al, alkalinity, and to a lesser extent, Ca. Acidic conditions (average pH 3.5 to 5.9; Table 2) have developed at the sites on noncalcareous sand (Muskoka, L. Joseph, and Nobel), whereas near neutral plume pH (average 7.0) is maintained at the calcareous site (Cambridge). Acidic conditions are accompanied by distinctly higher Al concentrations (0.03 to 24

**Table 1**
**Properties of the Aquifer Sediments and the Lysimeter Media**

| | Gravel (>2 mm) | Sand (0.06–2 mm) | Silt+Clay (<0.06 mm) | $CaCO_3$ equiv. | CEC[1] | $d_{10}$ | pH |
|---|---|---|---|---|---|---|---|
| | wt % | | | | (meq/100g) | (mm) | |
| **Septic System Sites** | | | | | | | |
| Cambridge (n=10) | 0.3 | 97 | 2.8 | 27 | 5.0 | 0.14 | 7.0 |
| L. Joseph (n=4) | 1.2 | 96 | 3.3 | <0.5 | | 0.11 | 5.9 |
| Muskoka (n=7) | 0.0 | 87 | 13 | 0.6 | 2.2 | 0.08 | 4.6 |
| Nobel (n=3) | 6.4 | 92 | 1.2 | <0.5 | | 0.14 | 3.5 |
| **Lysimeter Media** | | | | | | | |
| Noncalcareous (n=1) | 0.0 | 98 | 1.9 | <0.5 | | 0.13 | 4.0 |
| Calcareous (n=1) | 0.0 | 97 | 2.7 | 1.5 | | 0.10 | 6.9 |

[1]Average cation exchange values from Robertson et al. (1991)
pH values are from plume and lysimeter effluent values given in Tables 2 and 3.
$d_{10}$ is the grain diameter at which 90% of the sediment is coarser.

**Table 2**
**Average Plume (Proximal Zone) and Septic Tank Effluent Composition at the Four Study Sites**

| Site | Age (years) | | pH | $PO_4$-P | Al | Fe | $NO_3$-N | $NH_4$-N | Na | Cl | Ca | Alk[5] |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | mg/L | | | | | | | | |
| Cambridge[1] | 20 (1997) | effluent | 7.9 | 6.3 | 0.05 | 0.12 | 1 | 30 | 78 | 45 | 40 | 365 |
| | | plume | 7.0 | 4.8 | 0.015 | 0.07 | 16 | 0.1 | 54 | 24 | 90 | 276 |
| L. Joseph[2] | >10 (1998) | effluent | 6.6 | 6.3 | NM[6] | 0.16 | 0.1 | 22 | 117 | 166 | 37 | 191 |
| | | plume | 5.9 | 0.06 | 0.033 | 0.03 | 20 | <0.1 | 127 | 143 | 60 | 135 |
| Muskoka[3] | 10 (1997) | effluent | 6.6 | 13 | 0.08 | 0.17 | 0.1 | 59 | 90 | 55 | 14 | 316 |
| | | plume | 4.6 | 0.016 | 1.3 | 0.02 | 39 | 0.5 | 45 | 38 | 44 | 12 |
| Nobel[4] | 35 (1999) | plume | 3.5 | <0.05 | 21 | NM[6] | 76 | 0.2 | 55 | 69 | NM[6] | NM[6] |

n = number of measurements
Year of measurement and reference are in parentheses.
[1]Plume (n=7) (1988, from Robertson et al. 1991) except pH, $PO_4$ (n=11) 1997; Al (n=8) 1994; effluent (n=4) (1988, from Robertson et al. 1991)
[2]Plume (n=9 to 13) 1998, except Ca, Alk, Fe, $NH_4$ (n=2); Al (n=4) 2001; effluent (n=1) 1998
[3]Plume (n=8) (1988, from Robertson et al. 1991) except pH, $PO_4$ (n=8) 1997 and Al (n=22) 1994-97; effluent (n=5) (1988, from Robertson et al. 1991) except Al (n=2) 1994-1997
[4](n=3 to 4) 1999; effluent sample unavailable
[5]$CaCO_3$ equivalent alkalinity
[6]Not measured

below the infiltration bed (when compared to the fifth piezometer located 3 m upgradient), left little doubt that the septic system plume had been encountered and that it was little affected by dilution from background ground water.

## Methods

Site characterization methods, including sediment coring, piezometer installation, and ground water sampling techniques, were generally similar at all of the field sites and have been described in detail previously (Robertson et al. 1991; Robertson et al. 1998). Piezometer bundles, consisting of 5 to 13 variable-depth sampling tubes, were installed at Cambridge, Muskoka, and L. Joseph, using 5 cm diameter steel casing with expendable drive tips and a percussion hammer. Upon removal of the drive casing, the aquifer sand collapsed around the piezometer bundles to complete the installation. At Nobel, piezometers were installed individually or in pairs, in manually excavated, 8 cm diameter auger holes. Sediment samples for aquifer characterization were retrieved from auger cuttings (L. Joseph and Nobel sites) or by using special coring apparatus (Cambridge and Muskoka sites; Robertson et al. 1991).

All ground water samples were collected using a peristaltic pump with narrow diameter (0.6 cm) polyethylene tubing. Samples were filtered (0.45 µm) inline, prior to atmospheric exposure and were either acidified to pH < 2 with HCl (metals, $PO_4$, $NO_3$, $NH_4$ analyses) immediately after collection, or were left untreated (Na, Cl analyses). Measurement of pH was completed inline, prior to atmos-

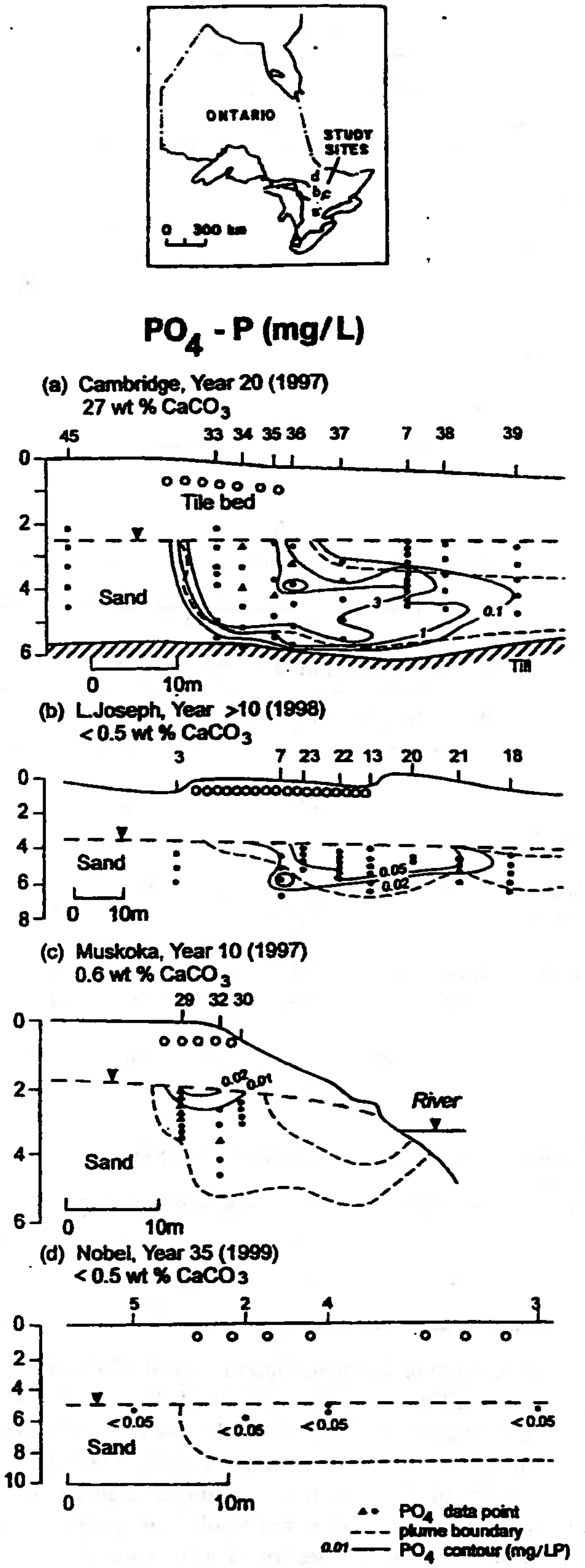


Figure 1. $PO_4$ distribution in four septic system plumes in aquifers with varying carbonate mineral contents. Plume boundaries are defined by 10 mg/L $NO_3$-N concentration (L. Joseph, Muskoka, and Nobel) or 40 mg/L Na concentration (Cambridge). Plume thickness at Nobel site is inferred.

pheric exposure, using a field portable meter with combination electrode checked against buffers of pH 4 and 7.

$PO_4$ was analyzed colorimetrically using a Cobas Fara® infrared spectrometer, which provided a detection limit of 0.01 to 0.05 mg/L P. Al analyses were completed by inductively coupled plasma mass spectrometry, which provided a detection limit of 0.005 to 0.01 mg/L.

Sediment samples from the vadose zone below the tilebed at Nobel were assessed by SEM analysis to evaluate the morphology and composition of any authigenic P solids that may have been present. Samples were prepared in a manner similar to that reported previously for the Cambridge and Muskoka sites (Zanini et al. 1998), by impregnating the sand in 2.5 cm diameter epoxy "pucks" and then preparing a polished surface for back-scattered electron imaging and quantitative wavelength dispersive spectroscopy for compositional analysis. Analyses were completed at the University of Western Ontario, using a Joel JXA 800 microprobe (Joel Corp., Boston, Massachusetts) with capability for elemental analysis (Reed 1996).

Sediment grain size properties were determined using sieves, and carbonate content was determined by measuring $CO_2$ evolved upon acidification with HCl (Chittick method; Dreimanis 1962).

### Lysimeter Experiment

A lysimeter experiment was undertaken to mimic the behavior of septic system treatment, using sand filters of varying carbonate mineral content. An aboveground, plywood-framed lysimeter, 44 cm long by 20 cm wide by 65 cm high, was constructed with a vertical panel separating it into two approximately equal compartments. Each compartment was loaded with same-sourced, noncalcareous, medium-grained filter sand, except that the sand in one compartment was mixed with 5 wt % limestone. The limestone media consisted of "crusher dust" which is a well-graded mixture of silt to 5 mm diameter crushed limestone. The lysimeter was set up at the Killarney site (Robertson and Blowes 1995), which contains a seasonal-use cottage septic system located 20 m from a softwater lake. During the nonfreezing period (May to October) from 1997 to 2000, the lysimeter was dosed, from the top, with septic tank effluent at a rate generally in the range of 0.6 to 2.6 L/day into each compartment (1.5 to 6 cm/day). Effluent was dosed at 1.2-hour intervals, using a peristaltic pump and timer and was allowed to drain through discharge tubes at the bottom of each compartment. Over the three-year duration of the test, a total of 913 L (179 pore volumes) and 815 L (173 pore volumes) were loaded to the calcareous and noncalcareous lysimeters, respectively. Sampling occurred by collecting gravity drainage from the discharge tubes. A tracer test was carried out to establish the effective water-filled pore volumes of the two compartments during operation. This was done by temporarily replacing the influent sewage with low TDS lake water (EC ~50 µS) and monitoring EC breakthrough in the lysimeter effluent. This established that the calcareous and noncalcareous compartments had effective pore volumes of 5.1 and 4.7 L, respectively, indicating a water-filled porosity of ~ 18%.

Table 3
Comparison of Septic Tank and Lysimeter Effluent Chemistry at Pore Volume 175

| | pH | $PO_4$-P | Al | Fe | $NO_3$-N | $NH_4$-N | Na | Cl | Ca | Alk |
|---|---|---|---|---|---|---|---|---|---|---|
| | | mg/L | | | | | | | | |
| Septic tank | 6.9 | 11 | 0.15 | 1.1 | <0.1 | 82 | 45 | 55 | 10 | 350 |
| Lys.— calcareous | 7.0 | 3.0 | <0.05 | <0.02 | 90 | <0.1 | 43 | 54 | 150 | 162 |
| Lys.— noncalcareous | 4.1 | 0.03 | 13.4 | 0.11 | 84 | 4.5 | 54 | 56 | 36 | <1 |

Measurements made September 2000

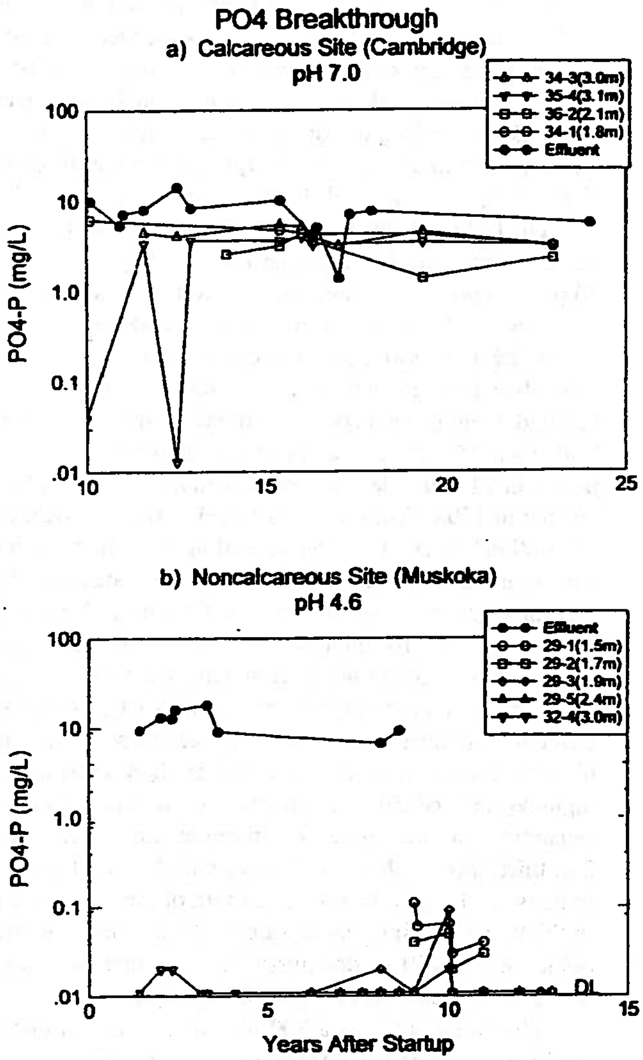


Figure 2. Long-term breakthrough of $PO_4$ in the proximal plume zones at Cambridge and Muskoka. Monitoring point locations are indicated by triangles in Figure 1. Legend indicates piezometer numbers and the distances between the piezometer intakes and the infiltration pipes.

## Results

### Aquifer and Lysimeter Characteristics

Table 1 shows that the aquifer and lysimeter grain size characteristics are roughly similar, consisting of medium-grained sand with only minor gravel (<7%) and silt+clay (<13%) contents. Carbonate mineral contents (Table 1) varied considerably, however, from 27 wt % at Cambridge to <1 wt % at the other three field sites. Mineralogy was not rigorously determined, although the sands are dominantly quartz at each of the sites. Gibbsite is expected to be present as a trace constituent derived from the weathering of Al-silicate minerals contained in the sand. At Cambridge, the carbonate content of the aquifer (27 wt %) is similar to the source till units (Wentworth and Port Stanley tills, 30 to 40 wt % $CaCO_3$ equiv.), in which calcite and dolomite are the primary carbonate minerals with dolomite dominating (Cowan 1970). The lysimeter media also had low carbonate mineral content (<0.5 wt %), but the compartment with the limestone admixture exhibited a higher carbonate mineral content (1.5 wt %), as expected, although this was less than the amount added in the initial mixture (5 wt%, or 2.3 kg). The difference is a result of the sample for carbonate content being collected at the end of the test, when some of the added limestone had presumably already been dissolved, or may be the result of some of the larger limestone particles not being represented in the sample due to its small size (~ 20 g). The latter is more likely as the amount of Ca present in the effluent (140 mg/L, Table 3) indicates that about 0.3 kg of limestone was leached during the test, which is only ~13% of the amount initially added.

### Plume Development and $PO_4$ Behavior

Figure 1 shows that these sites are hydrogeologically similar and that, except for Nobel, where thickness is uncertain, distinct ground water plumes, 2 to 3 m thick, have developed below the tile fields and are now migrating laterally within dominantly horizontal flow fields. Table 2 compares the concentrations of the significant major ions (Na, Cl, Ca, $NO_3$, $NH_4$, $PO_4$) and trace metals (Al, Fe) in the proximal plume core zones and in the septic tank effluent. Each of these plumes is well oxidized, with $NO_3$-N dominating (>95 % of ($NH_4$+$NO_3$)-N), which is a result of the relatively thick (2 to 5 m) unsaturated zones present at these sites (Figure 1). Comparison of the relatively conservative constituents (Na, Cl) in the plume zones and in the effluent (Table 1) indicate that these plumes are little affected by dilution (<50%). At Nobel, samples of the effluent were not obtained, but the relatively high Na and Cl concentrations in the four piezometers below the tilebed (average of 55 and 69 mg/L, respectively; Table 2), compared to the much lower values present in the upgradient piezometer (3 and 8 mg/L, respectively; piezometer 5), suggests that this plume is also relatively undiluted. Figure 1 also shows the distribution of $PO_4$ along the plume centerlines for the most recent comprehensive sampling episode at each site. A distinct P plume (>1 mg/L) extends versus ~5 mg/L in similar calcareous sediments) to be significant. If the development of acidic conditions and subsequent P attenuation is widespread in noncalcareous sediments, then these terrains might properly be identified as less vulnerable to P contamination from onsite sewage disposal.

This paper presents the results of further monitoring at two of the previously reported sites, one noncalcareous (Muskoka) and one calcareous (Cambridge), and results from two new field sites (L. Joseph and Nobel), both located on noncalcareous sediments. All four sites have conventional septic systems located on similarly textured, medium-grained sand aquifers, but with varying carbonate mineral contents. To assess the possibility of incorporating these P attenuation reactions into engineered sewage treatment systems, a field lysimeter experiment was undertaken comparing the treatment characteristics of twin sand filters, containing calcareous and noncalcareous sand media, treating septic tank effluent. Back-scattered electron images and microprobe compositional analyses of sand grains from below the infiltration beds assist in confirming the nature of the attenuation reactions at these sites.

## Site Descriptions

The two established field sites (Cambridge and Muskoka) have septic system plumes that have been discussed extensively in previous publications (Robertson et al. 1991; Aravena et al. 1993; Shutter et al. 1994; McAvoy et al. 1994; Shimp et al. 1994; Robertson 1995; Wilhelm et al. 1996; Zanini et al. 1998; Robertson et al. 1998; MacQuarrie et al. 2001). The two new sites (L. Joseph and Nobel) have not been previously reported. The septic systems at all four sites are of conventional design and have ground water plumes that have developed in similar hydrogeologic environments, consisting of relatively homogenous, unconfined sand aquifers, in which moderate velocity (20 to 70 m/yr), dominantly horizontal flow fields are present.

The Cambridge site has a typical gravity-fed septic system, consisting of a septic tank and 100 $m^2$ infiltration bed, that has been treating effluent from a four-person household since 1977. Metered water usage averaged 1.3 $m^3$/day during the period March to November 2000. In 1987, a ground water monitoring network of more than 500 monitoring points was installed that enabled the septic system plume to be mapped in detail. These results showed that the aquifer had limited dispersive capacity and as a result the plume remained distinct and had a core zone that was little affected by dilution for a distance of more than 100 m downgradient. A bromide tracer test revealed that the septic tank effluent resides for about 10 days in the 2 m thick unsaturated zone and then migrates horizontally in the ground water zone at a rate of 24 m/yr. Because the plume flowpath can be traced precisely at this site, it has been the focus of numerous studies (see previous), several of which have investigated the behavior of a distinct zone of $PO_4$ enrichment that is present in the plume core (Robertson 1995; Zanini et al. 1998; Robertson et al. 1998). The proximal plume zone has P concentrations (~5 mg/L) that are a substantial fraction (~75%) of the effluent amount. This P-rich zone is advancing downgradient at a retarded, but still significant, velocity of ~1 m/yr.

The Muskoka site has a study history similar to Cambridge. A detailed monitoring network, with more than 250 monitoring points, was installed at the site in 1987 to assess plume development at a then newly installed, conventional septic system servicing a two-person household. A plume, again exhibiting minimal dilution in the core zone, was observed to develop and then migrate, over a one-year period, to a surface water discharge point located 20 m downgradient. Plume characteristics are similar to Cambridge, except that acidic conditions developed as a consequence of sewage oxidation reactions (Equation 1) occurring in the absence of carbonate mineral buffering capacity in the aquifer. More importantly, the plume has remained virtually devoid of detectable $PO_4$ (<0.1 mg/L P; Robertson et al. 1991; Wilhelm et al. 1996; Robertson et al. 1998).

The L. Joseph site has a larger communal septic system servicing a seasonal use campground used by an average of 85 persons per day during peak use from May to September (Wilsack 1999). The septic system has a 1500 $m^2$ infiltration bed which is pressure dosed and has been in operation for more than 10 years, although its exact age is not known. A detailed monitoring network, similar to that of the Cambridge and Muskoka sites and consisting of 114 monitoring points in 21 multiple piezometer bundles, was installed at the site in 1998. These were installed in the area below the infiltration bed and for a distance of up to 30 m downgradient. Analyses of ground water electrical conductance (EC) and major plume constituents (Na, Cl, and $NO_3$) revealed that, in general, the plume could be easily distinguished from background ground water and that the plume core zone again showed little dilution compared to the septic tank effluent. One area of the plume appears to be underlain by higher EC water related to road salt applied seasonally to a highway located 50 m from the tilebed. This plume was separated from the septic system plume, however, by a 1 to 2 m thick layer of lower EC water with Na and Cl concentrations < 30 mg/L. The septic system plume is about 2 to 3 m thick and is migrating toward a surface water discharge point located 200 m downgradient, at a rate of ~70 m/yr based on calculations using the Darcy equation.

The Nobel site has a 300 $m^2$ infiltration bed that currently receives effluent from a service center on a major highway. This facility generates waste water from washrooms only, which is predominantly "blackwater," making it distinct from the other three sites which have more typical domestic waste water, consisting of both blackwater (toilet waste) and graywater (laundry and washwater). For a 35-year period prior to construction of the service center in 1995, the same tilebed received effluent from a 14-unit motel complex (Hughes 1999). Monitoring at this site was undertaken in 1998 and 1999 and was less detailed than at the other sites, consisting of the installation of five 1.3 cm diameter PVC piezometers, advanced into the shallow water table zone in the immediate area of the tilebed. Although a lesser number of monitoring points were used, the distinctly higher concentrations of Na, Cl, and particularly $NO_3$ encountered in the four piezometers located

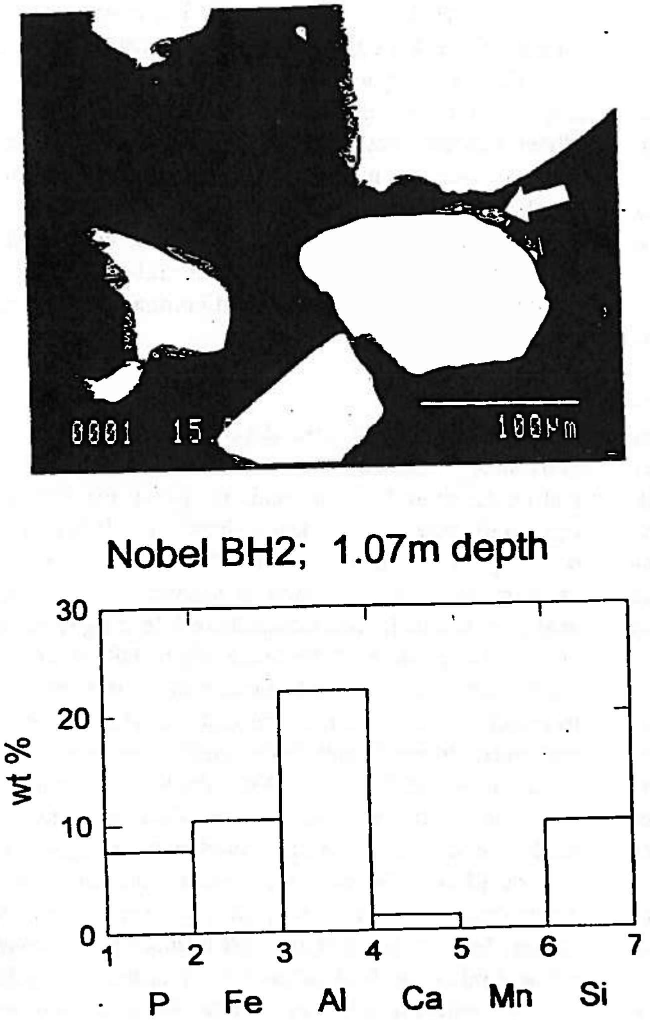


Figure 5. Back-scattered electron image and microprobe compositional analysis of authigenic coating on sand grain from the vadose zone at Nobel. Arrow indicates location of analysis, which is ~10 µm thick coating on amphibole grain with quartz inclusion.

mg/L; Table 2, Figure 3), compared to the neutral site (Cambridge, 0.005 to 0.046 mg/L; Table 2, Figure 3). Appelo and Postma (1994) report similarly high Al concentrations (1 to 10 mg/L) in ground water and surface water with pH <5 and indicate that this is the result of equilibrium with gibbsite or kaolinite. The Cambridge site has higher Ca (90 mg/L) and alkalinity (276 mg/L) compared to the L. Joseph and Muskoka sites (44 to 60 mg/L for Ca and 12 to 135 mg/L for alkalinity).

### Lysimeter Experiment

The lysimeter loading rate (1.5 to 6 cm/day) and the effective water-filled porosity value of 18% indicated a hydraulic retention time of 2 to 8 days in the lysimeter media, which was of similar duration to the effluent residence time in the unsaturated zone at Cambridge (~10 days, Robertson et al. 1991). Effluent $NO_3$ concentrations (Figure 4, Table 3) showed that substantial conversion of effluent $NH_4$ to $NO_3$ occurred (> 50% after pore volume 20), demonstrating that effluent oxidation was active. Effluent from the calcareous lysimeter remained near neutral in pH

Table 4
Saturation Indices for Selected Carbonate, Iron, Aluminum, and Phosphate Minerals in the Proximal Plume Zones and Lysimeter Effluent, Determined Using the Chemical Speciation Model PHREEQE (Parkhurst et al. 1985)*

| | Cambridge[1] | Muskoka[2] | L. Joseph[3] | Lysimeter[4] | |
|---|---|---|---|---|---|
| | | | | Calcareous | Noncalcareous |
| Calcite | −0.4 | −5.7 | −1.9 | −0.3 | −6.0 |
| Siderite | −6.8 | −11 | −5.9 | <−7.2 | −6.7 |
| $Fe(OH)_3$ | 1.1 | −1.3 | 0.4 | < 1.2 | 1.9 |
| Gibbsite | −0.1 | −1.0 | 0.08 | < 0.6 | −1.1 |
| Jurbanite | −4.6 | −0.2 | −1.9 | <−4.6 | −0.4 |
| Hydroxyapatite | 1.4 | −22 | −11 | 2.7 | −25 |
| Vivianite | −15 | −30 | −19 | <−19 | −17 |
| Strengite | 1.3 | 0.0 | −0.2 | < 0.9 | −1.0 |
| Variscite | −0.3 | −0.1 | −0.7 | < 0.05 | −0.4 |
| pH | 7.0 | 4.6 | 5.9 | 7.0 | 4.1 |

*Nobel plume water not included because of incomplete chemistry
[1]From Robertson et al. (1998); piezometer 33-1
[2]From Robertson et al. (1998); piezometer 32-3
[3]Using average plume chemistry given in Table 2 with assumed PE of +10 and $SO_4$ of 122 mg/L (piezometer CB20-4.5)
[4]Using effluent chemistry given in Table 3 with assumed PE of +10 and $SO_4$ values of 15 and 19 mg/L for the calcareous and noncalcareous effluent, respectively (values measured at pore volume 175)

(6.6 to 8.0), similar to the septic tank effluent (6.3 to 6.9), whereas distinctly lower pH values were measured in the noncalcareous lysimeter effluent, particularly after pore volume 50 (pH 2.5 to 4.8; Figure 4). Initially, $PO_4$ concentrations remained near or below detection (<0.05 mg/L P) in effluent from both lysimeters, but then at pore volume 105, $PO_4$ broke through abruptly in the calcareous lysimeter (Figure 4) and, excluding the single sampling event at pore volume 146, remained elevated thereafter (~3 mg/L P). This behavior contrasted sharply with the noncalcareous lysimeter, where, excluding the single sampling event at pore volume 20, $PO_4$ remained low (<0.1 mg/L P) throughout the test.

## Discussion

Overall, the geochemical characteristics of these plumes can be considered the normal result of sewage oxidation reactions (Equation 1) occurring in sediments with varying carbonate mineral buffering capacity. In calcareous terrain, acidity generated by the oxidation reactions stimulates carbonate mineral dissolution (Equation 4), resulting in near-neutral pH and Ca enrichment in the plume water. Where the sediments are carbonate deficient and do not have readily available buffering capacity, acidic conditions persist, resulting in gibbsite dissolution (Equation 2) and subsequent Al enrichment. Gibbsite was not specifically identified in this study but is assumed to be present as a result of weathering of Al-silicate minerals. Mineral saturation indices (Table 4) confirm that the Cambridge plume water and the calcareous lysimeter effluent are close to equilibrium with calcite (SI −0.3 to −0.4), whereas the noncalcareous sites are undersaturated (SI −1.9 to −6.0). Figure 3 and Table 4 show that, in general, these plumes remain close to equilibrium with gibbsite (SI 0.08 to −1.1), with the result that significantly increased concentrations of Al (> 0.1 mg/L) occur as pH drops below ~5.5. The exception is the more acidic waters in the Muskoka plume and the noncalcareous lysimeter (pH 4.1 to 4.6), which are moderately undersaturated with respect to gibbsite (SI −1.0 to −1.1). This may be the result of Al being controlled by other less soluble minerals such as jurbanite ($Al\,OH\,SO_4 \cdot 5H_2O$) at these pH levels, as has been suggested by Nordstrom (1982) and Appelo and Postma (1994). Both the Muskoka plume water and the noncalcareous lysimeter effluent are close to equilibrium with jurbanite (SI −0.2 to −0.4). A consequence of gibbsite dissolution and subsequent Al enrichment at low pH is that $PO_4$ can then be maintained at low levels (< 0.1 mg/L P at pH <5) as a result of variscite precipitation (Equation 3). All of the sites are close to equilibrium with variscite (SI −0.1 to −0.7). Table 4 suggests that strengite precipitation could also play a role in limiting $PO_4$ concentrations at the acidic sites (SI −1.0 to 0.0), but other phosphate minerals such as vivianite and hydroxyapatite are undersaturated (SI −11 to −30). Plume characteristics at all of these sites appear consistent with the aforementioned geochemical evolution. Long-term monitoring of $PO_4$ breakthrough at Muskoka and Cambridge (Figure 2) provides additional evidence that P attenuation is largely the result of mineral precipitation reactions, rather than other processes such as adsorption. If the latter process were dominant, P concentrations should slowly increase over time as sorption sites are used up, which is clearly not the case, even though these monitoring points are located only 1.5 to 3 m below the infiltration pipes and monitoring has been ongoing for more than 10 years. The cation exchange capacity of these sands are relatively low (2.2 to 5.0 meq/100g average at Muskoka and Cambridge, respectively; Table 1), further limiting the possibility that contrasts in sorption characteristics account for the differences in $PO_4$ behavior at these sites.

Back-scattered electron images and microprobe analyses of the vadose zone sands at Nobel provide additional evidence that the reaction chemistry suggested previously is the dominant P attenuation mechanism. Figure 5 shows the widespread occurrence of secondary mineral coatings, ~10 μm thick, on most sand grains at this depth (1.1 m). Analysis of the coating indicated in Figure 5 showed that it was P-rich (7.6 wt %) and had Al as the dominant cation (22.2 wt %), whereas lower amounts of Fe (10.4 wt %) and Ca (1.4 wt %) were present. Analyses of other coatings, shown in Figure 5, revealed similar compositions (5.5 ±0.8 wt % P, 16.5 ±3.5 wt % Al, 7.0 ±2.8 wt % Fe, and 1.4 ±0.7 wt % Ca; n=4). The molar ratio of Al:P in these coatings (~3.4:1) is, however, greater than that for variscite (1:1), suggesting that other aluminum minerals (e.g., jurbanite or gibbsite) may also be present. The considerable thickness of these coatings (5 to 20 μm) is indicative of authigenic mineral formation. Secondary coatings were also observed on sand grains from both of the lysimeters (0.35 m depth, not shown); however, these were thinner (~1 to 4 μm) making microprobe analysis more difficult. Nonetheless, semiquantitative analyses were obtained (5.2 ±0.8 wt % P, 21.5 ±4.1 wt % Al, 8.5 ±3.8 wt % Fe, 3.5 ±2.1 wt % Ca, n=6) which indicated a composition similar to the Nobel coatings. Al was again the dominant cation, suggesting that a reaction sequence similar to the attenuation process at Nobel was also occurring in the lysimeters. In contrast, scanning electron microscope analyses at the calcareous Cambridge site (Zanini et al. 1998) showed secondary P-rich coatings of similar morphology, but with Fe as the dominant cation (20 wt %) and only relatively minor Al (1 wt %), indicating a distinctly different attenuation mechanism.

## Implications

The apparent widespread occurrence of acidic conditions in septic system plumes on noncalcareous sediments and the resulting $PO_4$ attenuation suggest that noncalcareous terrain may be much less vulnerable to P loading from septic systems than is similar calcareous terrain. We are unaware that this distinction is currently considered, for example, in calculations of lakeshore P loading when septic systems are present (Dillon et al. 1986; Dillon and Molot 1996). Furthermore, it has been suggested that P sequestered in association with Al is stable under a wider range of pH and Eh conditions than if redox sensitive cations such as Fe were involved (Smith et al. 2001). The lysimeter experiment has demonstrated that these attenuation reactions can be readily incorporated into engineered treatment systems such as sand filters. We caution, however, that in most cases acidic conditions will develop only if thorough oxidization occurs, because most septic tank effluent has near-neutral pH and relatively high alkalinity. A properly functioning tilebed is thus required. In addition, the amount of acidity generated during oxidation will depend on waste water composition. Some waste waters, such as high alkalinity graywater with low ammonium content, may have insufficient acid-generating capacity to fully consume its inherent alkalinity (Robertson et al. 1998). At these sites, acidic conditions may not develop even if the effluent is well oxidized and the sediments are noncalcareous.

**Editor's Note:** The use of brand names in peer-reviewed papers is for identification purposes only and does not constitute endorsement by the authors, their employers, or the National Ground Water Association.

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**ATTACHMENT E – SCOPE NEWSLETTER – SPECIAL ISSUE: FATE OF PHOSPHORUS IN SEPTIC TANKS (JANUARY 2006)**

# SCOPE NEWSLETTER

NUMBER 63 January 2006

## Special issue : fate of phosphorus in septic tanks (autonomous waste water treatment systems)

*This is a review of a number of papers covering research knowledge and needs regarding nutrient contamination from septic tanks and other decentralised sewage treatment systems.*

### The Scope Newsletter

The SCOPE Newsletter is produced by the Centre Européen d'Etudes des Polyphosphates, the phosphate industry's research association and a sector group of CEFIC (the European Chemical Industry Council).

The SCOPE Newsletter seeks to promote the sustainable use of phosphates through recovery and recycling and a better understanding of the role of phosphates in the environment.

The SCOPE Newsletter is open to input from its readers and we welcome all comments or information. Contributions from readers are invited on all subjects concerning phosphates, detergents, sewage treatment and the environment. You are invited to submit scientific papers for review.

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## Septic tanks

### Review of research into nutrient release from autonomous sewage treatment systems

An overview of existing knowledge regarding nitrogen and phosphorus releases from septic tanks and autonomous sewage treatment systems shows the significant differences between the behaviours of these two nutrients. Nitrogen is only retained in septic tanks to a small extent, and once tank effluent is infiltrated into soil will tend to be converted to nitrates which are then very mobile and move with underground waters. Phosphorus, on the other hand, is significantly retained in septic tanks (up to 48%) and then precipitated or adsorbed in soil, so that significant contamination rarely moves more than a few metres from septic tank infiltration.

The authors assess available research regarding the operation of different types of autonomous or decentralised sewage systems, including conventional septic tank / soil absorption systems, but also innovative new systems such as grey/toilet water separate management systems, soil based and wetland systems.

#### Nitrogen contamination

Raw human sewage contains 2 – 8 kg total N/year. Traditional septic tanks are estimated to achieve 40% reductions in sludge volume, 60% reduction in biological oxygen demand (BOD), 70% retention of suspended solids, and 48% (Pell and Nyberg 1989) – 57% (Tetra tech 2002) which will in time need to be pumped for disposal. Settling and periodic pumping however is estimated to only remove 5 – 15% in inflow total nitrogen.

Soil adsorption can then remove a further 20% of total nitrogen, as ammonium, but this may be reversible when aerobic conditions occur and the ammonium is converted to soluble and water transportable nitrate.

Nitrogen from septic tank outflows can thus move to groundwater in a matter of days, and may tend to move mainly through shallow aquifers, posing an immediate risk to shallow wells and to surface waters.

### Phosphorus from septic tanks

Phosphorus in septic tanks and in their outflow behaves completely differently from nitrogen. Firstly, a significant proportion of inflow phosphorus in septic tanks is effectively removed by settling and subsequent pumping of septic tanks (48% - 57%, see above).

Phosphorus in septic tank outflow is 85% soluble orthophosphate, with some organic and inorganic particulate phosphorus attached to suspended solids. The latter will be retained in soil. The soluble orthophosphate can be retained in soils both by precipitation to mineral phases by ions present either in the septic tank effluent or in the soil (iron, aluminium, calcium …), or can be adsorbed to soil colloids (formation of a strong chemical bond between orthophosphate and clay minerals).

Typical mass balance studies have shown that 65% - 95% of the septic tank effluent phosphorus is found in soils within a few metres of the outflow point, even after years of septic tank operation. The "plume" of phosphorus concentrations downstream of septic tank outflow is estimated by several studies to develop 10x – 100x more slowly than the general plume of contamination.

**Aulenbach et al. 1981**, estimated 85% overall removal of phosphorus from sewage in septic tank systems (including soil retention, and assuming 5% of systems failing) around Lake George, New York State.

### Previous research

Several authors, many cited by Gold in this review, or elsewhere, have previously confirmed that the risks of phosphorus contamination of wells or surface waters from septic tank outflow are very limited.

* **Johnson & Atwater 1988** used 3m long experimental channels of different soil materials to test removal of different components of raw sewage, showing 96-99% removal of soluble phosphate with different soil types (3 loamy sands, 3 sands), whereas certain of the soil types tested removed only ¼ of the inflow inorganic nitrogen.

* **Robertson (2000, see SCOPE Newsletter n° 44)**, in 2-year field experiment using a lysimeter containing natural sandy soils, showed that septic tank effluent soluble phosphate levels were brought down below the detection limit (< 0.05 mgP/l). Only around 0.2% of soil iron had been used, forming stable coatings on the soil particles, suggesting that the system would remain effective for many years.

* **Harman et al 1996 and Robertson & Harmann 1999** studied the effluent plumes of 3 septic tank systems which had served a 200-pupil school (Langton) for nearly 50 years and a seasonal 200-person campsite for 5 and for 25 years (2 outflows), in Ontario, Canada. They reported that even after these long operational periods for large septic systems, around 85% of phosphate was being retained in the first 30 cm of soil around the outflows (vadose layer). Phosphate above background levels was detectable up to 75m away from the older system in a situation with mobile groundwater, but not beyond. They concluded that over long periods of use of septic tanks, long-term migration of phosphorus in the ground water zone may occur.

* **Zanini, Robertson et al. 1998** reported results from monitoring of the Langton school plume (as above) and of three domestic septic tank systems also in Ontario: Cambridge operational for around 20 years, Muskoka ten years, Harp Lake 30 years. They again found high phosphorus removal within the first 10-30 cm of soil around septic tank outflow infiltration pipes. Based on soil iron contents, they

estimated that it would take around 35 years to saturate the first 25 cm around a septic tank outflow, coherent with the 85% phosphorus retention in the 30 cm vadose zone observed for the Langton school site above.

* **Robertson et al. 1991** had reported analyses of the plumes of the Cambridge and Muskoka domestic septic tanks cited above. The plume at the Cambridge site was 130m long and 10m wide, for soluble contaminants such as nitrate and sodium ions, but phosphate was observed only immediately beneath the infiltration field. The plume at Muskoka was 20m long, reaching nearby surface water, but again phosphate was not detectable in the ground water below the infiltration field, nor at any significant horizontal distance away from the infiltration zone.

* **Robertson 1995** reported further monitoring results from the Cambridge domestic septic tank site, Ontario, Canada, operational for around 20 years, indicating a pattern of slow but pervasive expansion, with a migration velocity of about 1m/year. This represents a reatrding factor of around 20, probably as a result of soil particle sorption of phosphate. Phosphate levels then stabilise at around 1 mgP/l in the plume. Analysis of dilution factors led to the conclusion that around 25% of septic tank effluent P continued to be attenuated in the valdose zone, whilst throughout the rest of the plume soil capacity for phosphate sorption is progressively saturated thereby allowing slow extension of the phosphate plume. It is suggested that the attenuation in the valdose zone is probably the result of mineral precipitation, most probably of calcium phosphates. Comparisons with work at other sites suggest that higher attenuation values are obtained at lower pH levels (acidic waste water or soil conditions).The extension rate of the phosphate plume at the Cambridge site meant that it has already reached piezometers situated 20m from the tank infiltration bed, the separation distance locally required between septic tank infiltrations and sensitive surface waters, indicating that this distance is inadequate where the soil offers poor P retention.

* **Robertson & Blowes 1995**, studied a septic tanks system serving a seasonal cottage for four years after installation, at Sudbury, Ontario. In this situation, on poorly buffered silt earth, an acid contamination plume developed in the ground, but with limited phosphate mobility (retardation factor > 10) and no phosphate migration significantly beyond the infiltration bed gravel layer over the study period.

* **Robertson et al. 1998** looked at 10 mature septic tank systems in Ontario, including the 6 cited above, plus in addition another campsite (Camp Henry, 18 year old system), a resort (Delawana, 10 years) and 2 further houses (Paradise, 25 years and Killarney, 10 years). They concluded that phosphate migration is 20 – 100 times slower than the extension of the plume for other soluble contaminants, such as nitrates, but may reach around 1 m/year. Six phosphate plumes of over 10m were identified in sandy soils, but phosphate plumes <3m long on acidic silt or clay rich soils. Ground water phosphate concentrations immediately below the septic tank outflows were significantly lower than septic tank effluent levels, suggesting 23-99% phosphorus retention in the vadose zone within 1m of outflow pipes.

* **Ptacek 1998** studied the plume from the Camp Henry campsite septic tank (see above), Ontario, situated on sand alongside the coast. He found phosphate concentrations higher than background (but low at < 0.02 mgP/l) up to 60m away from the septic tank in part of the soil ground water (non-surface groundwater with low oxygen levels). This shows that septic tank outflows can contribute phosphate to surface waters where septic tanks are relatively close to surface waters (< 100m) and in sand substrate (rather than soil) over an impermeable layer.

* **Jones and Lee 1979** stated for Wisconsin, USA, found no detectable phosphate contamination at 15 sampling points situated 10 – 100 m distant from a septic tank tile field, 4 years after starting its operation, concluding "No evidence for phosphate transport fro septic tank effluent was found in any of the monitoring wells".

* **Gilliom and Parmont 1983**, for eight 20-40 year old septic systems close to Pine Lake, Puget Sound, Washington, concluded: "movement of more than 1% of effluent P to the lake was rare" (despite

movement of diluted effluent commonly occurring). Chen 1988, New York concluded that all of 17 septic systems examined showed "good removal of orthophosphate".

* **Wieskel and Howes 1992** looked at nutrients from four different 10-75 year old septic tank systems situated close to Buttermilk Bay, Massachusetts, and concluded that approx. 61% of septic tank effluent nitrogen would reach the Bay water (10 – 100m down gradient from the septic tanks), but that only approx 0.3% of the effluent phosphorus would reach the Bay.

* **Reneau and Pettry 1976**, studied phosphorus movement in sandy loam coastal plain sols in Virginia, from two septic systems aged 4 and 15 years. They detected no soluble phosphorus in a slowly moving water table below the septic tank outflows (seasonally perched water table) and orthophosphate concentrations < 0.2 µg/l at points 3m distant from the outflows.

* **Reneau 1979**, in the Virginia coastal plain, studied transfer of contamination from 10 domestic septic tank systems (all > 12 years old) to an agricultural tile drain situated 11 – 19 metres from the tank outlflows. Also, sampling wells were drilled 1.5 – 17 m away from three of the septic tank systems. Variation in the soil phosphorus abatement capacity was found, with 99% of phosphorus being removed within 8m for two of the septic tank outflows, but only at 30m for the third. Mean phosphorus concentrations were lower in the sampling wells 13 – 17m away from the septic tank outflows than in the surface water receiving the tile drain outfall, and phosphorus was not detectable in the tile drain outflow (lower concentration than in the receiving water).

* **Reneau, Hagedorn and Degen 1989**, reviewing available literature, concluded that "the limited movement of P away from on site wastewater disposal systems (OSWDS) is well-documented" and that "Most field studies indicate that P contamination is limited shallow groundwaters adjacent to OSWDS and that P sorption continues under saturated conditions". The risk of phosphorus movement to surface waters is thus minimal.

* **Viraraghaven & Warnock 1976**, in Ottowa, Canada, analysed contaminants in groundwater samples immediately below a septic tank drainfield for a system which had been operating for three years. Most samples (14 out of 18) showed phosphate concentrations lower than the background groundwater, but some were 3-4 x higher.

* **Sawhney and Starr 1977**, used sampling tubles installed 15 – 120 cm below and 20 – 120 cm horizontally distant from a secptic tank outflow trench system. They concluded that soil 15-30 cm below the trench was continuing to remove most of the outflow phosphate after 6 years of septic tank operation, and that 60 cm of soil should "effectively remove phosphorus from septic system drainfields for a number of years and should allow only minimal additions to the groundwater". They also showed through alternate operation of 2 outflow trenches from the septic tank that the soil" regenerated" its phosphorus removal capacity: this is conform to laboratory experiments which show that soil phosphorus fixing capacity is increased by wetting – drying cycles.

* **Chen 1988** analysed contamination in groundwater samples at various distances from 17 different septic tanks systems situated near the shores of lakes in Northern and Eastern New York State. Of 45 sampling points, situated 0 – 3m below the surface and up to 100m distant from the septic tank outflows, only 4 showed phosphate concentrations > 0.1 mgP/l and the ten points > 40m distant all showed concentrations < 0.04 mgP/l. The author noted that several sites showed groundwater contamination near enough the lake edge for transfer to surface water to be possible and indicates that problems of nutrient and coliform bacteria transfer from septic tanks where their outflow is situated in rocky or sandy substrate over an impermeable layer.

* **Alhajjar et al. 1989**, compared nitrogen and phosphorus contamination of ground water for two sets of respectively 8 and 9 domestic septic tank systems, with households using in one case phosphate-based and in the other carbonate-based laundry detergents. They concluded that there was zero probability of more than 5% of phosphate reaching ground water in all cases with mean

phosphate transfer < 0.1 mgP/l in all cases. However, they found total nitrogen concentrations reaching groundwater of 39 and 69 mgN/l for the phosphate- and carbonate-detergent households respectively, concluding that the use of phosphate-based detergents led to substantially lower levels of nitrogen contamination. They conclude that the use of carbonate-built (P-free) detergent "exacerbates nitrogen leachate to ground water" with human health and environmental implications.

* **Alhajjar et al. 1990**, compared phosphate and nitrogen removal in lab-scale soil-filled columns simulating mound, new conventional and mature conventional septic tank soil drainfields, fed with septic tank effluents from households using phosphate-built or phosphate-free (zeolite built) laundry detergents. The columns fed with P-detergent effluent showed higher outflow phosphate levels, but on the other hand lower outflow nitrogen levels. The authors concluded that P-built detergents used in households served by septic tanks reduce nitrogen leaching to groundwater by a factor of 1.8 (new systems) to 2.1 (mature systems), and "slightly" increase phosphate leaching compared to households using zeolite based detergents. They suggest that this may be the result of precipitation of struvite (magnesium ammonium phosphate) or similar minerals because of higher available phosphate in the drainfield soil.

* **Woods, 1993**, studied the fate of phosphorus in a context where soil absorption of phosphorus was susceptible to be problematic, around Harp Lake, Ontario (180 km North-East of Toronto, see Zanini, et al. 1998 above): a thin heterogeneous till soil over acidic Precambrian shield bedrock. For one typical domestic septic tank dating from 1962, most of the phosphorus from 30 years use was found in the 14 cm soil layer below the tile-bed outflow. Phosphorus in the aquatic sediments at this and four other septic tank sites around Harp Lake showed mean phosphorus concentrations in the zone contaminated by the outflows with means 0.5 – 13x and maximums 0.3 – 38x background levels (see p155). The author concludes that septic tank phosphorus could be reaching the lake in 3 out of 5 cases, but in only in one case were mean contaminated zone phosphorus concentrations >20 µgP/l.

## Conclusions

It thus appears clear that phosphorus contamination from septic tanks is limited, because much of the phosphorus is retained in the septic tanks, and because that released in the outflow is then retained in soil, often in the soil immediately around the discharge infiltration, thus resulting in only a very low proportion (<1%) of phosphorus in septic tank inflow being susceptible to reach surface waters. There may however be concern where septic tanks are situated close to (< 10m) surface waters or water courses in areas of calcareous sandy soil.

*"Research needs in decentralized wastewater treatment and management: a risk-based approach to nutrient contamination"*
http://www.ndwrcdp.org/userfiles/RESEARCH_NEEDS_PROCEEDINGS_CD.PDF

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## See also:

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**ATTACHMENT F – A REVIEW OF THE COMPONENTS, COEFFICIENTS AND TECHNICAL ASSUMPTIONS OF ONTARIO'S LAKESHORE CAPACITY MODEL (A.M. PATTERSON, P.J. DILLON, J.J. HUTCHINSON, M.N. FUTTER, B.J. CLARK, R.B. MILLS, R.A. REID, AND W.A. SCHEIDER, LAKE AND RESERVOIR MANAGEMENT 22[1]: 7-18, 2006**

*Lake and Reservoir Management* 22(1):7-18, 2006


# A Review of the Components, Coefficients and Technical Assumptions of Ontario's Lakeshore Capacity Model

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## Abstract

Paterson, A.M., P.J. Dillon, N.J. Hutchinson, M.N. Futter, B.J. Clark, R.B. Mills, R.A. Reid and W.A. Scheider. 2006. A review of the components, coefficients, and technical assumptions Ontario's Lakeshore Capacity Model. Lake and Reserv. Manage. Vol. 22(1):7-18.

Phosphorus is the nutrient that most often limits the primary productivity of inland lakes on the Precambrian Shield. Recognizing the need to develop quantitative relationships to assess the impact of shoreline development on phosphorus concentrations in lakes, the Lakeshore Capacity Model (LCM) was developed by the Ontario Ministry of the Environment, Canada. The LCM is a steady-state mass-balance model that uses empirical relationships to predict the ice-free total phosphorus concentration of a lake. The model, calibrated and tested on lakes on the Precambrian Shield, has subsequently formed the basis for management decisions in the public and private sectors. Over the past two decades the coefficients, input parameters and assumptions of the LCM have been modified and updated to reflect an improved scientific understanding of the relative importance of sources and losses of phosphorus in lakes and watersheds. Here we present a comprehensive review of the components, coefficients and assumptions of the most recent version of the LCM (v. 3.0), providing a standard reference for all users of the model.

Key Words: phosphorus, shoreline development, mass balance, nutrient model, Ontario

Phosphorus is the nutrient that most often limits the primary productivity and algal biomass of inland lakes on the Precambrian Shield (Schindler *et al.* 1971). Commonly measured as total phosphorus (TP), its concentration is strongly related to the aesthetic appearance of a lake and thus is an important measure of water quality. For example, increases in TP are correlated to decreases in water transparency (Dillon and Rigler 1975) resulting from an increase in algal biomass or biovolume (Nicholls and Dillon 1978, Dillon *et al.* 1988, Molot and Dillon 1991). Increased biomass may result in long-term decreases in deep-water oxygen concentrations with consequences for coldwater fish habitat (Molot *et al.* 1992, Clark *et al.* 2002).

Phosphorus enters surface water from atmospheric precipitation, stream and overland flow, and groundwater, with lake concentrations regulated by local geology, land-use, lake morphometry, soil type and depth, and human activity (Dillon *et al.* 1993). At sites without urban drainage or point sources such as sewage treatment plants, domestic waste from septic systems may represent the most important anthropogenic source of phosphorus to recreational lakes on the Canadian Shield (Dillon *et al.* 1993). With increases in shoreline development in recreational areas, quantitative, as opposed to qualitative, relationships are required to assess the impact of anthropogenic disturbances on lake trophic status.

A conceptual model proposed by Dillon and Rigler (1975) quantified linkages between natural sources of phosphorus, human inputs from shoreline development, water balance, lake morphometry and the ice-free TP concentration of a lake. The model, referred to as Ontario's Lakeshore Capacity Model (LCM), was calibrated on Shield lakes in central Ontario (Dillon *et al.* 1986, Hutchinson *et al.* 1991, Dillon *et al.* 1994) and has since formed the basis of many management decisions in the public and private sectors. Model predictions may subsequently be used to predict other measures of trophic status, including chlorophyll *a*, water transparency and deep-water oxygen concentrations (Dillon and Rigler 1974, Nicholls and Dillon 1978, Molot and Dillon 1991, Molot *et al.* 1992).

Over time, the coefficients and input parameters of the LCM have been modified as new information has become available. These changes reflect an improved scientific understanding of the relative importance of sources and losses of phosphorus in lakes and watersheds (Dillon *et al.* 1993, Dillon and Molot 1996). Furthermore, assumptions regarding the measurement of anthropogenic sources of phosphorus, including per capita water use, have been re-evaluated and modified. Here we present a review of the most recent version (v. 3.0) of the LCM, with the intent of providing a standard reference for all users of the model. We provide an update of the components, export coefficients and technical assumptions that govern the use of the model for predicting the ice-free TP concentration of inland lakes on the Canadian Shield.

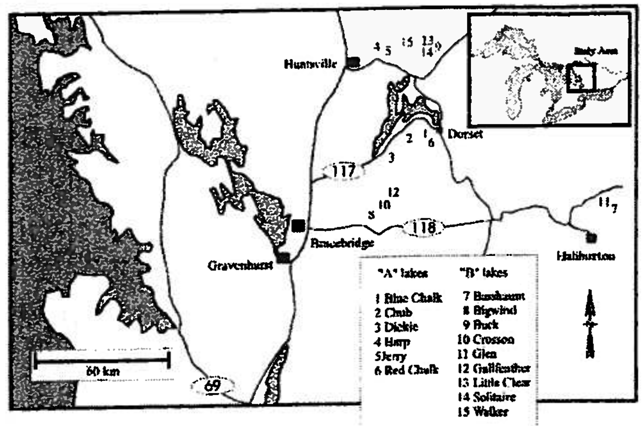


**Figure 1.**-Map of the Lakeshore Capacity Study area in south-central Ontario. Shown are the locations of the 'A' (calibration) and 'B' (test) lakes (see Methods).

## Methods

The methods used to calibrate the LCM, including the calculation of nutrient budgets, morphometric and hydrological data, are presented in detail in Scheider *et al.* (1983), Locke and Scott (1986), Dillon *et al.* (1986), and Dillon *et al.* (1994). In summary, the model was developed using six lakes with hydrologically calibrated watersheds in south-central Ontario. These intensively studied lakes were selected to span a gradient of development pressure, varying from no-to-moderate shoreline development in their watersheds (Fig. 1). Model coefficients were derived for the intensively studied lakes, and assumptions regarding human phosphorus contributions were made. Predictions of TP were subsequently calibrated against detailed monitoring data from the six lakes (Dillon *et al.* 1986, Dillon *et al.* 1994) and other lake sets in central Ontario (Dillon *et al.* 1986, Hutchinson *et al.* 1991).

During the calibration phase, water samples were collected from all inflow and outflow streams of the intensively studied lakes. Samples were collected from nine additional streams in the region to explore the relationship between watershed geology and TP export (Dillon *et al.* 1986). Meteorological data were collected from 1976 to 1980 at nine stations within 50 km of Dorset, Ontario. Stream stage or level was continuously monitored for the calibration of all lake inflows and outflows and converted to discharge using stage-discharge curves developed for each stage control structure (*i.e.*, weir and/or flume). Details of control structure design, stage-discharge relationships, gauging and sampling frequency, and calculation of discharge from ungauged portions of the subwatersheds are provided by Scheider *et al.* (1983).

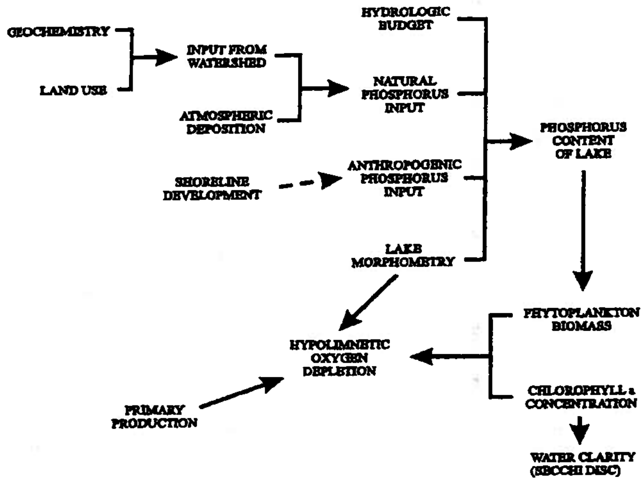


**Figure 2.**-Conceptual model used in the Lakeshore Capacity Model of the relationships linking phosphorus and the trophic status of lakes.

Chemical samples were collected at weekly intervals for the intensively studied lakes and monthly for an independent set of test lakes (Fig. 1). Analytical methods for the chemical analysis of precipitation, lake and streamwater samples are outlined further in MOE (1983).

For the purposes of recalibration, meteorological and hydrological data were updated to 2001 and 1998, respectively, using the above methods. To validate version 3.0 of the LCM, predicted values were compared to monitoring data provided by the District Municipality of Muskoka for a 61-lake watershed in central Ontario. Measured values in the test set were available for 21 lakes and were based on 2-7 years of deep-station water samples collected at spring overturn. For model comparison, spring-overturn measurements were converted to ice-free values using the equation provided in Clark and Hutchinson (1992; $[TP]_{ICE\text{-}FREE} = 0.80*[TP]_{SPRING\text{-}OVERTURN} + 2.04$).

# Results and Discussion

## *1.0 Predicting the Total Phosphorus Concentration of Lakes*

Ontario's LCM is a steady-state mass balance model that uses empirical relationships to calculate a phosphorus budget for a lake based on natural and anthropogenic phosphorus, hydrologic and morphometric data, assumptions or estimates (Fig. 2). The model output, the TP concentration of a lake during the ice-free season, is predicted using the following equation:

$$[TP]_{ICE\text{-}FREE} = L_T * (1-R_p) * (0.965 * q_s)^{-1} \qquad (1)$$

where, $L_T$ = is the total areal loading rate (*i.e.*, the total annual supply from atmospheric deposition, the watershed, and anthropogenic sources divided by the lake surface area), $R_p$ = the retention coefficient, and $q_s$ = the areal water load (*i.e.*, lake outflow discharge or volume (Q) * $A_o^{-1}$).

To understand how this equation is derived, we describe each of the relevant components below (section 2.0). In section 2.1, we begin with a discussion of the background or 'natural' inputs of phosphorus from atmospheric deposition, forested watersheds and wetlands. This is followed by a description of the anthropogenic sources of phosphorus to lakes, which include contributions from shoreline development (section 2.2), agriculture and urban centres (section 2.3). Finally, in sections 2.4 and 2.5, we discuss the hydrological components and loss processes of phosphorus considered by the model. In addition to the technical components, we illustrate how many lakes within a watershed or chain may be modelled concurrently (section 3.0), and provide a validation of the predictive ability of the model (section 4.0).

## *2.0 Components of the Lakeshore Capacity Model (LCM)*

### 2.1 Undeveloped or "Natural" Phosphorus Inputs

All lakes receive inputs of phosphorus from natural sources through atmospheric deposition, runoff from streams and overland flow from the watershed. Groundwater may also contribute to the natural phosphorus load, although in many Shield lakes this contribution is negligible (Dillon *et al.* 1993) or reported as surface water derived from the interception of shallow subsurface groundwater flow. Considered alone, the natural phosphorus supply (*i.e.*, the supply excluding development or land-use change in the watershed) allows one to estimate the 'undeveloped' phosphorus concentration for a lake, an important management benchmark for evaluating the impact of present and future development on trophic status (Hutchinson *et al.* 1991, Hutchinson 2002).

In catchments with small drainage ratios (watershed area: lake area) and minimal shoreline development, atmospheric precipitation may be a significant source of phosphorus, contributing more than half the total load in some cases (Dillon *et al.* 1993). The LCM currently uses a value of 16.7 mg TP·$m^{-2}$·$yr^{-1}$ for atmospheric deposition, calculated as a 17-year mean (1984/85 to 2000/01) from three meteorological stations in central Ontario (Fig. 3). This value is lower than previously reported figures for Ontario: 75.0 mg·$m^{-2}$·$yr^{-1}$ reported by Dillon and Rigler (1975); 35.3 mg·$m^{-2}$·$yr^{-1}$ reported by Dillon *et al.* (1986); and 20.7 mg·$m^{-2}$·$yr^{-1}$ reported by Dillon *et al.* 1993. This lower value is due to the removal of a fourth station with local sources of contamination and improvements in the analytical detection limit and not the result of a trend in phosphorus deposition over time (Fig. 3).

Phosphorus may be transported in dissolved organic and/or inorganic forms, as colloidal phosphorus, or in particulate forms ranging from sub-micron-sized particles to leaf-litter and other large debris (Dillon *et al.* 1986). The TP export coefficient may be calculated using a variety of empirical relationships, depending on the availability of meteorological and hydrological data. For example, Dillon *et al.* (1991) developed a multiple regression model to predict phosphorus export from 32 forested watersheds in central Ontario. While 88% of the variation in TP export was explained by this equation, the model required the input of eight independent variables including hydrological data (*e.g.*, springflow) that are not readily available from maps, air photos, or weather records.

In most regions, therefore, this model is not easily adapted for general use. However, a simpler version of this relationship recognizes the importance of wetlands in controlling the export of phosphorus to Precambrian lakes (Dillon *et al.* 1991, Dillon and Molot 1997). A linear regression of phosphorus export and % wetland area (expressed as % peat) was developed from 20 watersheds in central Ontario (following Dillon and Molot 1997; Fig. 4). Phosphorus export was calculated as a long-term mean from more than two decades of empirical measurement (Ontario Ministry of the Environment, Dorset Environmental Science Centre, unpublished data). Equation (2) may be used to predict the phosphorus supply from forested watersheds on the Precambrian Shield:

$$TP\ (kg \cdot yr^{-1}) = \text{catchment area}\ (km^2) * (0.47 * \%\ \text{wetland area} + 3.82) \quad (2)$$

When specific data exist for sub-watersheds, calculations should be made separately and then summed.

In addition to wetland coverage, the earliest version of the model recognized that the export coefficient may also vary significantly with land-use (Dillon and Rigler 1975). Based on experimental work and a literature survey of 69 watersheds, Dillon *et al.* (1986) determined that forested watersheds with ≥15% cleared land had a mean TP export value of 9.8 mg $TP \cdot m^{-2} \cdot yr^{-1}$, compared to 5.5 mg $TP \cdot m^{-2} \cdot yr^{-1}$ for forested watersheds where <15% of the land was cleared. Similarly, a doubling of the coefficient from 10 to 20 mg $TP \cdot m^{-2} \cdot yr^{-1}$ was used to estimate phosphorus export from forested and pasture lands, respectively, for a small watershed on sedimentary geology in southern Ontario (Winter and Duthie 2000). For forested watersheds on the Precambrian shield with ≥15% cleared or pastured land, and where the % wetland area is known, the LCM uses the following equation to calculate phosphorus export:

$$TP\ (kg \cdot yr^{-1}) = \text{catchment area}\ (km^2) * (0.47 * \%\ \text{wetland area}^2 + 3.82) * (1.8) \quad (3)$$

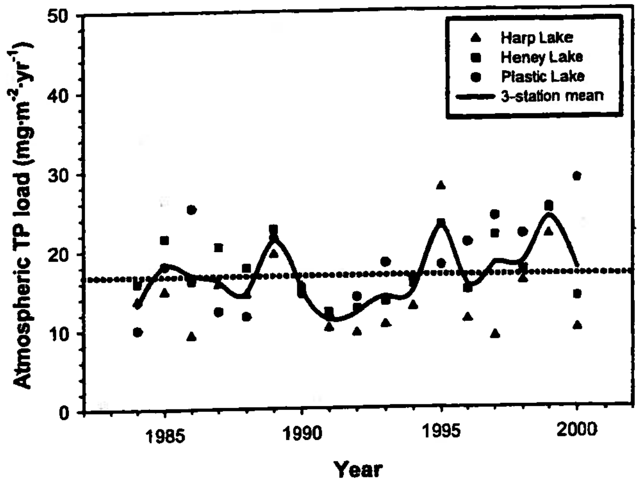


**Figure 3.**-Plot of long-term atmospheric TP loads from three meteorological stations in south-central Ontario, Canada. Shown are values from each station (Harp Lake, Heney Lake, and Plastic Lake) and a three-station mean. The dotted line represents the 17-year mean (16.7 $mg \cdot m^{-2} \cdot yr^{-1}$).

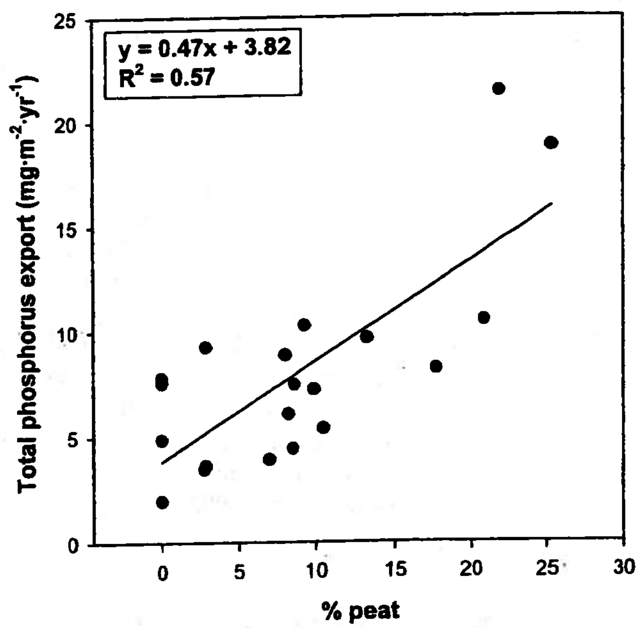


**Figure 4.**-Relationship between mean annual export of total phosphorus and % wetland (expressed as % peat) for 20 watersheds in south-central Ontario, Canada.

Equation (3) assumes that export from wetlands is similar for forested and cleared watersheds. A constant is applied to reflect the mean difference in export between forested and forest-pasture watersheds on the Canadian Shield (*i.e.*, $(9.8)*(5.5)^{-1}$; please see Dillon *et al.* 1986 for derivation of export coefficients).

Equations (2) and (3) rely heavily on accurate determinations of wetland areas. If the % wetland area is unknown for a watershed, it can be estimated from 1:50,000 topographic maps or an analysis of aerial photographs. Alternatively, new GIS-based software (*e.g.*, Ontario Flow Assessment Techniques; Chang *et al.* 2002), or existing datasets (*e.g.*, Rapid Assessment Technique; Hogg *et al.* 2002) may be used to delineate watershed and wetland areas. In all other cases, a default value of 15% is used by the LCM, based on the median value of percent wetland area calculated for 846 watersheds in the District of Muskoka, Ontario (O'Connor and Dillon, unpublished data). However, the default should be applied with caution given the wide range of wetland areas observed in Canadian Shield watersheds and the high sensitivity of the model to this input parameter (Mills *et al.* 2004). Equations (2) and (3) can lead to an under-prediction of TP export from small watersheds when wetlands comprise <3.5% of the catchment area. In such cases, the export coefficients of 5.5 and 9.8 mg $TP{\cdot}m^{-2}{\cdot}yr^{-1}$ should be used for less than or more than 15% cleared land, respectively, for forested watersheds on the Precambrian shield (Dillon *et al.* 1986).

## 2.2 Anthropogenic Phosphorus Inputs – Shoreline Development

*2.2.1 Per capita phosphorus load.* While the contribution to the total phosphorus load from anthropogenic sources comes from a variety of sources, domestic sewage (*i.e.*, septic systems) represents the primary potential source to most recreational lakes on the Canadian Shield (Dillon *et al.* 1986, Dillon *et al.* 1993). The total load from septic systems is a function of the annual phosphorus supply contributed per capita and the demand (*i.e.*, # of individuals) on the system each year. Historically the former has been reported as 0.80 $kg{\cdot}capita^{-1}{\cdot}yr^{-1}$, based on an average phosphorus contribution of measured TP in septic tanks of 13.2 $mg{\cdot}L^{-1}$ and a daily water use of 164 L per capita (*i.e.*, 13.2 $mg{\cdot}L^{-1}$ * 164 $L{\cdot}capita^{-1}{\cdot}day^{-1}$ * 365 $days{\cdot}yr^{-1}$ = 0.79 $kg{\cdot}capita^{-1}{\cdot}yr^{-1}$; Dillon *et al.* 1986). While a significant range of values was observed for effluent concentrations (5-21.8 $mg{\cdot}L^{-1}$), the resultant loading value compared well with other studies across North America and in Europe in the 1960s and 1970s (summarized in Dillon *et al.* 1986).

Recent studies suggest that previous estimates of effluent concentrations do not reflect reductions in the phosphate content of detergents. An examination of 174 measurements from seasonal and permanent residences in Lanark and Rideau Counties, Ontario, revealed a mean TP concentration of 8.2 $mg{\cdot}L^{-1}$ (range = 4.3-13.3 $mg{\cdot}L^{-1}$; Hutchinson 2002). Based on these surveys, we have selected 9 $mg{\cdot}L^{-1}$ as an estimate of septic effluent concentrations following the reduction of detergent phosphates.

Water use is a second component of the per capita phosphorus load that requires re-evaluation. The number reported by Dillon *et al.* (1986; 164 $L{\cdot}capita^{-1}{\cdot}day^{-1}$) is based on a literature review of ten studies published in the 1970s. In the early 1980s, the Ontario Ministry of the Environment (MOE 1982) reported water use at 275 $L{\cdot}capita^{-1}{\cdot}day^{-1}$ for houses, apartments and cottages in Ontario. An increase in water use since the 1970s in cottaged areas is difficult to validate given the scarcity of long-term monitoring data; however, an increased use of appliances such as dishwashers and washing machines may have resulted in increased water use in recreational areas, reflected in the higher figures reported above. In the absence of accurate monitoring data, an estimate of 200 $L{\cdot}capita^{-1}{\cdot}day^{-1}$ is now used to reflect lower water use in cottaged areas than small municipalities, but an increased use of appliances since the 1970s. Assuming an effluent concentration of 9 $mg{\cdot}L^{-1}$ and a daily per capita water usage of 200 $L{\cdot}capita^{-1}{\cdot}day^{-1}$, the per capita phosphorus contribution is now estimated as 0.66 $kg{\cdot}capita^{-1}{\cdot}year^{-1}$:

$$9\ mg{\cdot}L^{-1} * 200\ L{\cdot}capita^{-1}{\cdot}day^{-1} * 365\ days{\cdot}yr^{-1} = 0.66\ kg{\cdot}capita^{-1}{\cdot}yr^{-1} \quad (4)$$

*2.2.2 Usage figures.* Usage or occupancy (measured in capita $years{\cdot}yr^{-1}$) considers both the number of days a residence is occupied each year and the mean number of people in each residence. Multiplied by the total number of dwellings and the per capita phosphorus load, usage provides an estimate of the total potential anthropogenic TP supply from shoreline development. Occupancy varies depending on the type of development considered (*e.g.*, cottage, permanent home, resort) and accessibility factors, such as the distance from major urban centres and highway intersections. Occupancy measurement is hindered by demographic changes on a lake-to-lake basis, or temporally within watersheds. Long-term changes in the ratio of seasonal to permanent habitations, for example, can have significant impacts on the total anthropogenic contribution through time (Hutchinson 2002).

The LCM employs seven development types based on the original Lakeshore Capacity Study Land Use Model (Downing 1983): seasonal, extended seasonal, permanent, resorts (serviced, housekeeping cabins, or meal plan), trailer parks, youth camps, and campgrounds/tent trailers/RV parks (Table 1). Based on 1139 surveys obtained in the late 1970s, Downing (1983) determined usage values for seasonal (including summer, weekend, and long-weekend use), extended seasonal, and permanent dwellings (Table 1). Extended seasonal exists as a compromise between seasonal and permanent residences and should be used for non-permanent developments that have reliable year-round access.

Estimating the number of present-day developments within each category is a considerable task. Preferable sources of such information include surveys, records from lake resident associations, recent topographic maps, aerial photos, municipal lot records or some combination of the above. Of particular concern is estimating the relative contributions from each development category for future development scenarios. The Ontario Ministry of the Environment applies a "hybrid" usage factor for future developments based on the existing ratio of seasonal to permanent dwellings, with the assumption that use patterns will be unchanged in the future. A more conservative approach is to consider all new developments as either extended seasonal, or permanent, reflecting possible worst-case scenarios for future development pressure.

Usage figures for resorts, trailer parks, youth camps and campgrounds may also be derived. Based on a use period of ~140 days from Victoria Day (May 24) to Thanksgiving weekend (2nd week in October), and a mean occupancy of 3.07 people·day$^{-1}$ (*i.e.*, average occupancy of cottage developments), the usage figure for resort units can be calculated as:

$$140 \text{ days} * 3.07 \text{ people}\cdot\text{day}^{-1} = (430) * (365)^{-1} = 1.18 \text{ capita years}\cdot\text{yr}^{-1} \quad (5)$$

If staff members are included, resort usage per unit can be closely estimated using the extended seasonal value of 1.27 capita years·yr$^{-1}$. Trailer parks, which include permanent mobile homes and trailers commonly used on a seasonal basis, have a usage equivalent to seasonal cottages (0.69 capita years·yr$^{-1}$). A phosphorus load of 125 grams·capita$^{-1}$·yr$^{-1}$ is included for youth camps, based on an estimated use period of 70 days·yr$^{-1}$ (July, August, and open and closing dates) and a daily contribution per person of 1.8 grams [(660 grams·capita$^{-1}$·yr$^{-1}$) * (1 year)(365 days)$^{-1}$]. Finally, for recreational campgrounds with septic systems servicing pump outs, comfort and wash stations, a usage of 0.37 capita years·yr$^{-1}$·site$^{-1}$ is assumed (Downing 1983).

*2.2.3 Other considerations–attenuation of septic system phosphorus in watershed soils.* Attenuation of septic system phosphorus in watershed soils has been the subject of considerable scientific discussion over the past two decades. While management agencies traditionally assume that all phosphorus from septic systems is mobile over the long-term, others argue that trophic status models should consider a retention coefficient to account for reduced mobility of phosphorus in some environments. For example, Hutchinson (2002) found that the application of a retention coefficient of 74% in watersheds with significant soils (*i.e.*, native, mineral-rich acidic soils of >1 m in thickness), improved the positive bias of predicted over measured total phosphorus (72%; Dillon *et al.* 1994). This bias may be explained in part, however, by the coefficients used to estimate the "natural" phosphorus load in the Hutchinson (2002) model, which were higher than in version 3.0 of the LCM.

**Table 1.**-Usage values for shoreline properties (modified from Downing 1983).

| Development Type | Usage (capita years·yr$^{-1}$) |
|---|---|
| Seasonal residence | 0.69 |
| Extended seasonal residence[1] | 1.27 |
| Permanent residence | 2.56 |
| Resorts (serviced, housekeeping cabins, or meal plan)[2] | 1.18 |
| Trailer Parks | 0.69 |
| Youth Camps | 125 grams·capita$^{-1}$·yr$^{1}$ |
| Campgrounds/tent trailers/ RV parks[3] | 0.37 |

[1]Extended seasonal included residences that are non-permanent, but have reliable year-round access;

[2]If staff members are included, usage per resort unit is estimated using the extended seasonal value of 1.27;

[3]Includes recreational campgrounds with septic systems servicing pump outs, comfort and wash stations.

The attenuation of septic system phosphorus is controlled by both adsorption to charged surfaces and precipitation as insoluble minerals (Robertson *et al.* 1998, Robertson 2003), with the former limited by the availability of charged particle surfaces in soils (Zanini *et al.* 1998). In a study of phosphate mobility and persistence in ten septic systems plumes, Robertson *et al.* (1998) determined that phosphorus-enriched groundwater plumes migrated in calcareous soils at a sufficient rate to be of concern over the long-term. In contrast, much smaller phosphorus plumes were present at four acidic sites situated on non-calcareous sands, with proximal-zone phosphate concentrations consistently measured at <2% of effluent concentration and limited plume mobility (Robertson 2003).

Robertson (2003) cautions, however, that the acidic conditions required to promote attenuation may only develop if thorough oxidation occurs, and thus phosphorus mobility may vary with changes to the condition of the septic system tilebed, the water load to septic systems, the position of the water table and the characteristics of the tile field soils. In addition, the amount of acidity generated will depend on the composition of the waste water. For example, graywater with low ammonium content may not provide the acid-generating capacity required for mineralization processes. Finally, migration rates of phosphorus plumes in soils are slow (commonly reported at <1 m/yr), and a significant lag-time may occur before the cumulative effects of shoreline development are realized in lakes (Dillon *et al.* 1986, Dillon and Molot 1996). Consequently, the Ontario Ministry of the Environment recommends a cautious approach, adhering to the "precautionary principle."

**Table 2.**-Morphometric characteristics of lakes in the 61-lake test watershed. See Appendix 1 for watershed schematic.

| Characteristic | Minimum | Maximum | Mean | Mean (Calibration lakes) |
|---|---|---|---|---|
| Lake area ($A_o$) (ha) | 5 | 1380 | 97 | **58** |
| Catchment area ($A_d$) (ha) | 15 | 9470 | 976 | **474** |
| Wetland area (%) | 0.0 | 43.3 | 7.4 | **5.4** |
| Mean annual runoff (m) | 0.446 | 0.589 | 0.547 | **0.511** |

Note. Although all lakes were modeled, predicted values were compared to monitoring data from the following lakes: Arrowhead, Bay, Bella, Bing, Buck, Camp, Clark, Foote, Fox, Golden City, Jessop, Loon, Oudaze, Perch, Rebecca, Ripple, Solitaire, Tasso, Vernon-Main, Waseosa (Appendix 1).

**Table 3.**-Input coefficients that have changed in version 3.0 of Ontario's Lakeshore Capacity Model.

| Coefficients | Units | LCM v. 2.1[1] | LCM v. 3.0 |
|---|---|---|---|
| Background or 'natural' | | | |
| i) Precipitation | $mg \cdot m^{-2} \cdot yr^{-1}$ | 20.7[2] | 16.7 |
| ii) Watershed/wetland equation | | | |
| - slope | | 0.57[1] | 0.47 |
| - intercept | | 3.08[1] | 3.82 |
| Anthropogenic | | | |
| i) Per capita load | $capita\ yrs \cdot yr^{-1}$ | 0.80[3] | 0.66 |
| ii) % septic load, 0-100 m | % | 100[3] | 100[4] |
| % septic load, 100-200 m | % | 100[3] | 66[4] |
| % septic load, 200-300 m | % | 100[3] | 33[4] |
| % septic load, > 300 m | % | 0[3] | 0 |

[1]LCM v. 2.1 is an unpublished version of the model that has been used by Ontario Ministry of the Environment staff over the past several years. Version 2.1 represented a significant improvement over earlier versions of the model, as it considered the importance of wetlands as a source of phosphorus to lakes.

[2]Dillon *et al.* 1993;

[3]Dillon *et al.* 1986;

[4]Hutchinson 2002.

The balance between recent science and the need for regulatory precaution can be accommodated by use of a three-step 'graduated' approach that recognizes that phosphorus attenuation may occur in some watersheds and probably increases with distance from the lake shoreline. First, in watersheds (or portions of watersheds) with shallow (generally <3 m) or absent soils, and with exposed or fractured bedrock, the existing assumption of zero retention is applied. This assumption has been used for many lakes (Hutchinson *et al.* 1991, Dillon *et al.* 1994, Hutchinson 2002, see model validation below), with the majority of model predictions falling within the coefficient of variation of measured values. Second, at sites where deeper (generally >3 m), non-calcareous native soils are present, the modeller may use the coefficients outlined in Table 3. Here, the degree of attenuation increases with distance from the shoreline, with an assumption of zero export at distances of >300 m (Hutchinson 2002). Third, in cases where site-specific characteristics demonstrate that retention of septic system phosphorus may occur over the long-term, attenuation factors may be developed for consideration by local planning authorities and plugged into the model.

*2.2.4 Other considerations–contributions from the clearing of shoreline lots.* Shoreline development may also contribute phosphorus to a lake when forests are removed for lot development. The LCM now includes a constant loading value for the cleared portion of each lot, regardless of the development type. A survey of more than 1000 cottage lots in the Muskoka-Haliburton region of Ontario determined that the mean forest disturbance, by proportion, was ~15% of the total lot area (mean lot size = 3789 $m^2$). Thus, for Shield lakes the export from individual lots may be calculated using the export coefficient for forest-pasture (*i.e.*, forested areas with >15% cleared land), multiplied by the mean size of surveyed lots (Euler 1983):

$$\text{Export from lots: } 9.8\ \text{mg·m}^{2}\text{·yr}^{1} * 3789\ \text{m}^{2}\text{·lot}^{-1} = 0.04\ \text{kg TP·lot}^{-1}\text{·yr}^{1} \qquad (6)$$

Finally, an allowance is made for developments that use holding tanks that pump and remove waste from the watershed. A correction factor is included whereby sites with holding tanks are excluded from the total load. However, this correction is made independent of the lot-load presented above.

In summary, the potential anthropogenic supply from shoreline development (kg·yr$^{-1}$) can be calculated as follows:

$$J_A = P(1-R_S)\sum_{i=5}^{5}\sum_{j=1}^{4} n_{ij}x_i d_j + \sum_{j=1}^{4} n_{.j}d_j Lot + O \qquad (7)$$

where, $P$ = per capita TP load, $R_S$ = the proportion of development with holding tanks and wastes removed from the watershed, $n_{ij}$ = the number of development units ($i$) at distance ($j$) from the shoreline, $x_i$ = usage based on development type (*e.g.*, seasonal), $d_j$ = TP attenuation factor based on distance from the shoreline (see explanation below), $Lot$ = lot load, and $O$ = additional TP loads that have been measured (*e.g.*, inputs from STPs, contributions from golf courses). Note that $J_A$ is calculated in kg but converted to mg for subsequent stages of model development.

### 2.3 Anthropogenic Phosphorus Inputs–Agriculture and Urbanization

In addition to cleared or pastured land, the LCM provides unique export coefficients for land subjected to intensive crop agriculture and/or urbanization. In a survey of 198 watersheds draining cropland in North America, Chambers and Dale (1997) report a mean phosphorus export coefficient of 30 mg·m$^{-2}$·yr$^{-1}$. Winter and Duthie (2000) lowered their estimate to 25 mg·m$^{-2}$·yr$^{1}$ for a small watershed in southern Ontario based on a large proportion of non-row crops, which experience generally lower levels of soil erosion than row crops. In the Lake Simcoe watershed in central Ontario, Winter *et al.* (2002) report phosphorus export from watersheds draining cropland within the range of values reported by Chambers and Dale (1997; 12-39 mg·m$^{-2}$·yr$^{1}$). The LCM currently uses a value of 30 mg·m$^{-2}$·yr$^{1}$,or 0.3 kg·ha$^{-1}$·yr$^{1}$, for the portion of the watershed with intensive agriculture, based on the mean value presented by Chambers and Dale (1997) and subsequent studies in southern and central Ontario (Winter and Duthie 2000, Winter *et al.* 2002).

Vacation areas in Ontario include urban hubs on recreational lakes, and phosphorus from these sources should be considered. In a review of 78 studies, Dodd *et al.* (1992) report export values ranging from 45 to 150 mg·m$^{-2}$·yr$^{1}$ for watersheds draining urban areas. In southern Ontario, Winter and Duthie (2000) suggested that a lower value of 50 mg·m$^{-2}$·yr$^{1}$ was appropriate for a watershed draining low- to mid-density residential development. To reflect the range of values reported from other studies in Ontario, but recognizing that high-density residential developments are rare for most regions of the Canadian Shield, the LCM now uses a similar value of 50 mg·m$^{-2}$·yr$^{1}$,or 0.5 kg·ha$^{-1}$·yr$^{1}$, for the export of phosphorus from urban areas draining directly to lakes. However, given that export coefficients for agriculture and urban centres are derived primarily from a review of published values, we suggest that region or site-specific values based on empirical measurements be substituted where available.

### 2.4 Hydrological Characteristics

In addition to the nutrient budget and physical characteristics of the lake and watershed, the model incorporates hydrological information into its calculation of lake TP concentration. Lake outflow discharge or volume (Q, in m$^{3}$·yr$^{-1}$) is used by the model to estimate the areal water loading rate ($q_s$), which is used in the final calculation of TP and expressed as the outflow discharge divided by the lake area (*i.e.*, Q·$A_o^{-1}$).

For watersheds with unknown lake outflow volumes, the following equation is used to calculate Q, requiring estimates of catchment area ($A_d$), lake area ($A_o$), and mean annual runoff:

$$Q = (A_d + A_o) * \text{mean annual runoff} \qquad (8)$$

Runoff can be calculated using several methods, depending on the availability of meteorological data and knowledge of the physical and geological characteristics of the watershed (Dillon *et al.* 1986). If changes in lake storage and groundwater flux are assumed to be zero, then runoff will equal the difference between precipitation and evapotranspiration, provided independent measurements of these variables are available. In the absence of more detailed information, average long-term runoff values have been published for all of Canada and provide the most convenient and reliable source of runoff data. For example, LCM version 3.0 uses runoff estimates from the Canada Department of Fisheries and Environment (1978) that have been updated and collated into a look-up table. These values are interpolated at a spatial grid-size of one minute by one minute.

### 2.5 Loss Processes

Phosphorus is lost from the water column of lakes by two major processes: sedimentation and outflow, with the former being the most difficult to quantify (Dillon and Evans 1993). Sedimentation can be measured using sediment traps, estimating bulk sedimentation rates in sediment cores, or using a mass-balance approach as the difference between total inputs and outputs (*i.e.*, the fraction of the total input not lost to outflow, or the retention coefficient - $R_p$). Due to possible sediment resuspension, mineralization, over/under trapping and the relative short period of deployment, sediment traps are considered to be the least reliable method. Retention, however, showed close agreement when estimated from sediment cores and the mass balance approach in five of seven calibration lakes in central Ontario, although $R_p$ was consistently higher when estimated using sediment chronologies (Dillon and Evans 1993).

The retention coefficient may be expressed by the relationship between the settling velocity of phosphorus (v, in $m \cdot yr^{-1}$) and the areal water load ($q_s$). For oligotrophic lakes in the Precambrian Shield, the retention of TP (loss to lake sediments) can be estimated with the following equation:

$$R_p = v * (v + q_s)^{-1} \tag{9}$$

Based on empirical calibration by Kirchner and Dillon (1975), v can be estimated as 12.4 $m \cdot yr^{-1}$ for dimictic, oligotrophic lakes on the Precambrian Shield with oxic hypolimnia. In lakes that experience prolonged periods of anoxia during the ice-free season, the settling velocity should be reduced to 7.2 $m \cdot yr^{-1}$, reflecting either a release of TP from lake sediments or less efficient removal of TP from the water column under anoxic conditions (Dillon *et al.* 1994). This correction results in a significant improvement of the fit of measured and predicted TP in anoxic lakes (Dillon *et al.* 1994).

Equation 9 does not consider the influence of partial anoxia on phosphorus retention. For example, Reckhow (1977) proposed predictive models for lakes with hypolimnia that are oxic, partially anoxic or anoxic during the summer stratification period. Similarly, predictive models can be used to estimate the degree of anoxia occurring in lake hypolimnia (*e.g.*, the anoxic factor; Nürnberg 1995a, 1995b). Coupled with predictions of the release rate of phosphorus from lake sediments and lake morphometry (Nürnberg 1988, Nürnberg and LaZerte 2004), an internal phosphorus load can be estimated. However, these steps did not improve the predictive ability of the LCM for anoxic lakes (*e.g.*, 13 lakes in the test set were anoxic). Similarly, an examination of the Reckhow (1977) and Ontario Ministry of the Environment models showed comparable results for an anoxic lake in central Ontario (Dillon *et al.* 1986).

Losses to outflow can be measured directly as the amount of discharge (outflow volume/unit time) and the nutrient concentration. In a uniformly mixed system, the concentration of TP in the lake should equal that of the outflow. However, long-term empirical data from many lakes in central Ontario (Dillon *et al.* 1986) have shown that this is not correct. $[TP]_{OUTFLOW}$ is more closely related to $[TP]_{LAKE}$ by the following equation:

$$[TP]_{OUTFLOW} = 0.956 * [TP]_{LAKE\ (Ice\text{-}free)} \tag{10}$$

## *3.0 Connecting Lakes in a Watershed*

To accurately predict phosphorus concentrations in lakes, modeling must occur at a watershed-scale (*e.g.*, Appendix 1). In practice this means the nutrient budget of downstream lakes must account for phosphorus loads from upstream lakes (Dillon and Rigler 1975). As with previous versions of the LCM, version 3.0 assumes this transfer is complete, with no net retention occurring in connecting streams or wetlands (Dillon *et al.* 1986). To calculate the upstream load, the ice-free phosphorus concentration at the outflow of the upstream lake ($[TP]_{ICE\text{-}FREE}$ * 0.956) is multiplied by its outflow discharge (Q). For example, for a lake with an outflow $[TP]_{ICE\text{-}FREE}$ of 5.0 $\mu g \cdot L^{-1}$ and a discharge of 1,500,000 $m^3 \cdot yr^{-1}$, the net contribution to the downstream phosphorus load would be 8.25 $kg \cdot yr^{-1}$. In cases with multiple lakes upstream (*e.g.*, Lake Vernon – Main Basin, Appendix 1), the contribution of each much be calculated separately and added to the TP load of the downstream lake.

## *4.0 Validation of LCM Version 3.0*

To validate LCM version 3.0, we presented model results from a 61-lake watershed on the Canadian Shield in central Ontario. The study lakes span gradients of measured [TP], morphometry, and landscape position typical of lakes in the southern Canadian Shield (Appendix 1, Table 2). To determine if the predictive ability of version 3.0 improved, we compared the model results from v. 3.0 to those from a previous, unpublished version of the LCM (v. 2.1; see Table 3 for summary of coefficients for v. 2.1 and v. 3.0). The significance of each model was tested using a Wilcoxon signed-ranks test, a non-parametric equivalent to a paired t-test. A p-value of $<0.05$ indicates a significant difference between predicted and measured values.

In general, both models show close agreement between predicted and measured [TP] for the 21 lakes with at least two years of monitoring data (Fig. 5). However, version 2.1 shows a significant positive bias in the prediction of [TP] (Wilcoxon signed-ranks test: Wilcoxon W = 183, p = 0.02, n =21). This bias was not apparent using version 3.0, where no significant difference was found between predicted and measured values (Wilcoxon W = 94, p = 0.47, n = 21).

Recent improvements in the predictive ability of the model are explained primarily by the lower coefficient values used to estimate the 'natural' phosphorus load to lakes (Table 3). In the case of inputs from the watershed, these changes reflect the gradual decline in stream loads that have been observed for watersheds in central Ontario (Dorset Environmental Science Centre, unpublished data).

### *5.0 Conclusions*

Following its development and calibration in the 1970s and 1980s, versions of Ontario's LCM have been used widely in the public and private sectors to assess the impacts of shoreline development on lake trophic status. Over the past two decades, the coefficients and assumptions of the model have evolved as new information has become available. Consequently, multiple versions of the model are now in use in Ontario and elsewhere, leading to uncertainty in the interpretation and implementation of model results. This paper provides a technical review of the most recent version of the LCM (version 3.0), providing a single reference for users of the model.

As with all models, however, the LCM has limitations and assumptions. The LCM was calibrated on small, headwater lakes on the Canadian Shield. Although the coefficients have been tested on a wider distribution of lakes within central Ontario and elsewhere (*e.g.*, Hutchinson *et al.* 1991, M.N. Futter *et al.* unpublished, this study), caution is needed when applying the model to lakes in other regions. In particular, estimates of phosphorus loads from atmospheric deposition, overland flow and wetlands may vary spatially. Furthermore, variations in runoff across regions will have a significant impact on the hydrological processes that govern the flow of water and nutrients through watersheds. Finally, these parameters may vary temporally, as illustrated in this study. Thus, we recommend that model coefficients be calibrated regionally to reflect local conditions and, with monitoring, be updated periodically to evaluate long-term changes in sources and losses of phosphorus.

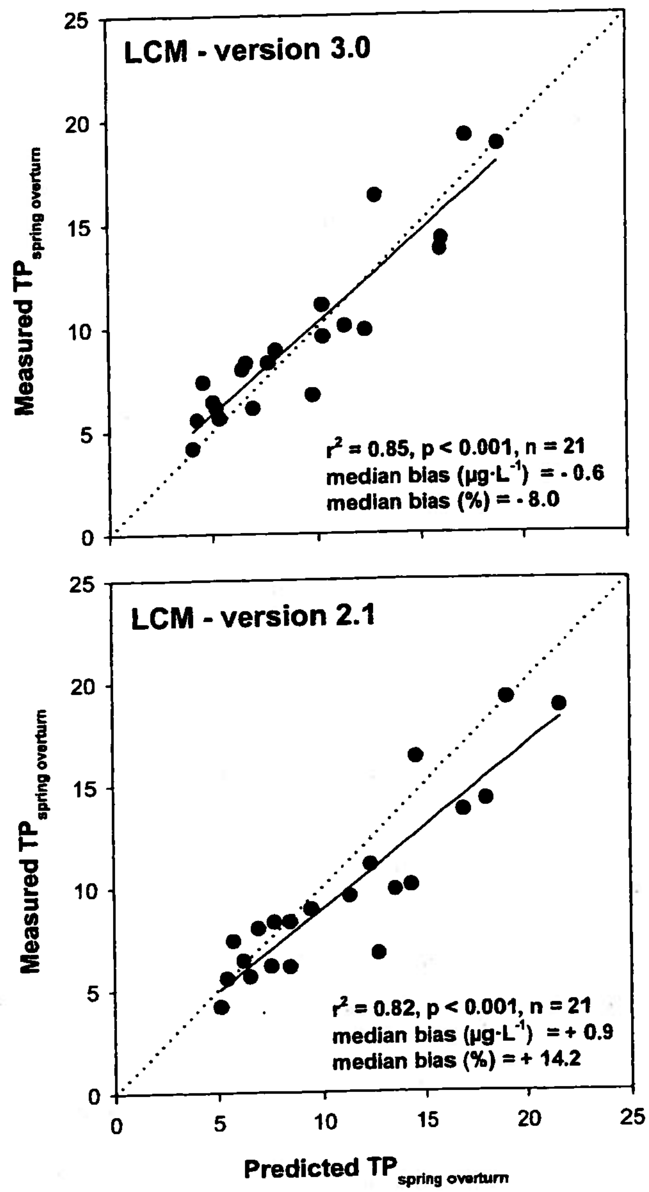


**Figure 5.**-Plots of predicted versus measured [TP] at spring overturn for 21 lakes in the Vernon Lake watershed (see Apppendix 1), calculated using LCM versions 3.0 (top figure), and 2.1 (bottom figure). The dotted lines represent 1:1.

## Acknowledgments

We thank J. Findeis, R. Girard, B. Hutchinson, and L. Scott for assistance with sample collections and hydrology measurements. We also thank Judi Brouse and colleagues at the District Municipality of Muskoka for providing the input data for model validation. This manuscript was greatly improved with suggestions from three anonymous reviewers.

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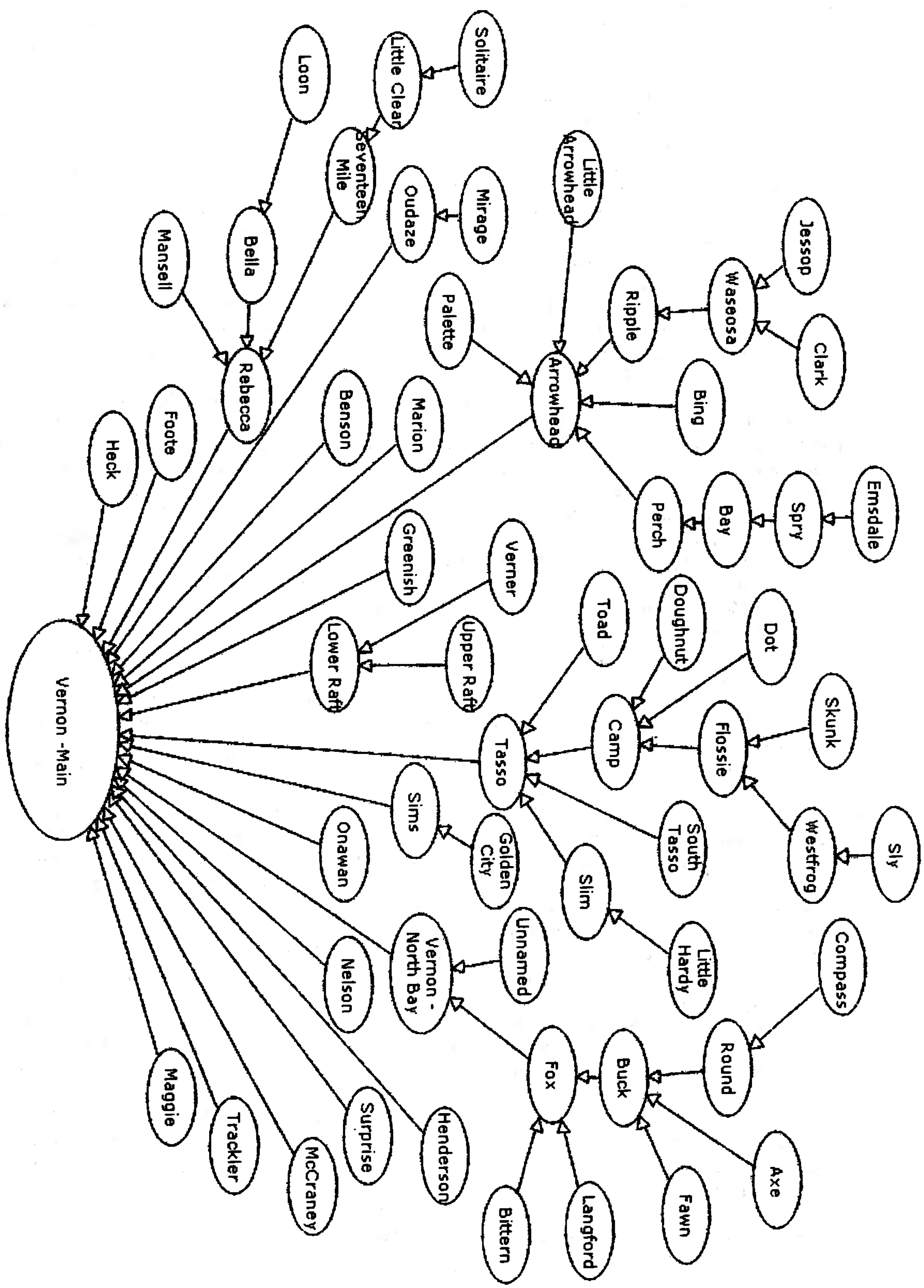
Appendix 1.-Schematic of the Vernon Lake watershed (modeled using LCM V. 3.0).



## ATTACHMENT G – EVALUATION OF THE ABSORPTION BED'S EFFICIENCY UNDER THE ECOFLOW BIOFILTER – ROGER LACASSE AND NAIDER FANFAN

# Evaluation of the absorption bed's efficiency under the Ecoflo® Biofilter

Roger Lacasse[1] and Naider Fanfan[2]

The Ecoflo® Biofilter is a wastewater treatment system designed for onsite and decentralized applications. Preceded by a septic tank, the chain of treatment comprises a biofiltration unit and a polishing field allowing the treated wastewater to be absorbed by the native soil. The Ecoflo® Biofilter performance has been evaluated by numerous organizations (Ministry of Environment of Quebec, BNQ, NSF, CSTB, etc.) for the last 20 years. All studies have demonstrated that the system produces an effluent with concentrations much lower than 10 mg/L in TSS and $BOD_5$. However, few data have been collected up to now to assess the efficiency of the absorption bed or of the polishing field receiving the treated effluent produced by the Ecoflo® Biofilter.

Within the process of the approval of the Ecoflo® technology in the State of Virginia, an independent study has been realized from 2003 to 2007 to determine the performance of the system comprising an Ecoflo® Biofilter and a 30 cm thick absorption bed in accordance with the State requirements. This study was directed by Dr. Robert A. Rubin, P.E., Ph.D., professor emeritus at North Carolina University.

## Material and method

The testing protocol includes the monitoring of six different residential sites during an 18-month period for the four types of soils present in Virginia and defined in Table 1.

**Tableau 1 Types of soils in Virginia**

| Type of soil | Permeability (min/inch) |
|---|---|
| I | ≤ 16 |
| II | 17 to 45 |
| III | 46 to 90 |
| IV | 91 to 120 |

Each site installation comprises a septic tank followed by an Ecoflo® Biofilter and an absorption bed located underneath the biofilter (two installations were installed using a closed bottom unit with separate dispersal field). The biofilter contains a non-mechanical tipping bucket and distribution plate on top of the peat/fiber media. Three suction lysimeters (High flow porous ceramic cup suction lysimeter model 1920F1-B01M3) have been installed on each site in order to measure the quality of the groundwater upstream of the absorption bed (lysimeter # 3 not illustrated in figure 1), at 12 inches under the absorption bed located just underneath the Ecoflo® Biofilter (lysimeter # 1), and at 10 feet downstream of the biofilter (lysimeter # 2). Figure 1 presents a typical installation of the biofilter and the monitoring equipment.

According to the protocol, samples were generally taken on a monthly basis at the septic tank effluent, at the Ecoflo® Biofilter's as well as at the three lysimeters. The following analysis was performed on samples: $CBOD_5$, TSS, fecal coliforms, nitrates and TKN. Additional monitoring was added for total phosphorus evaluation at 13 selected sites in soils type I to III. No sampling for $CBOD_5$ and TSS evaluations were performed in the lysimeters #1 installed underneath the biofilter, because the Ecoflo® Biofilter's effluent concentrations were already lower than Virginia's regulation requirement for these two parameters (30 mg/L).

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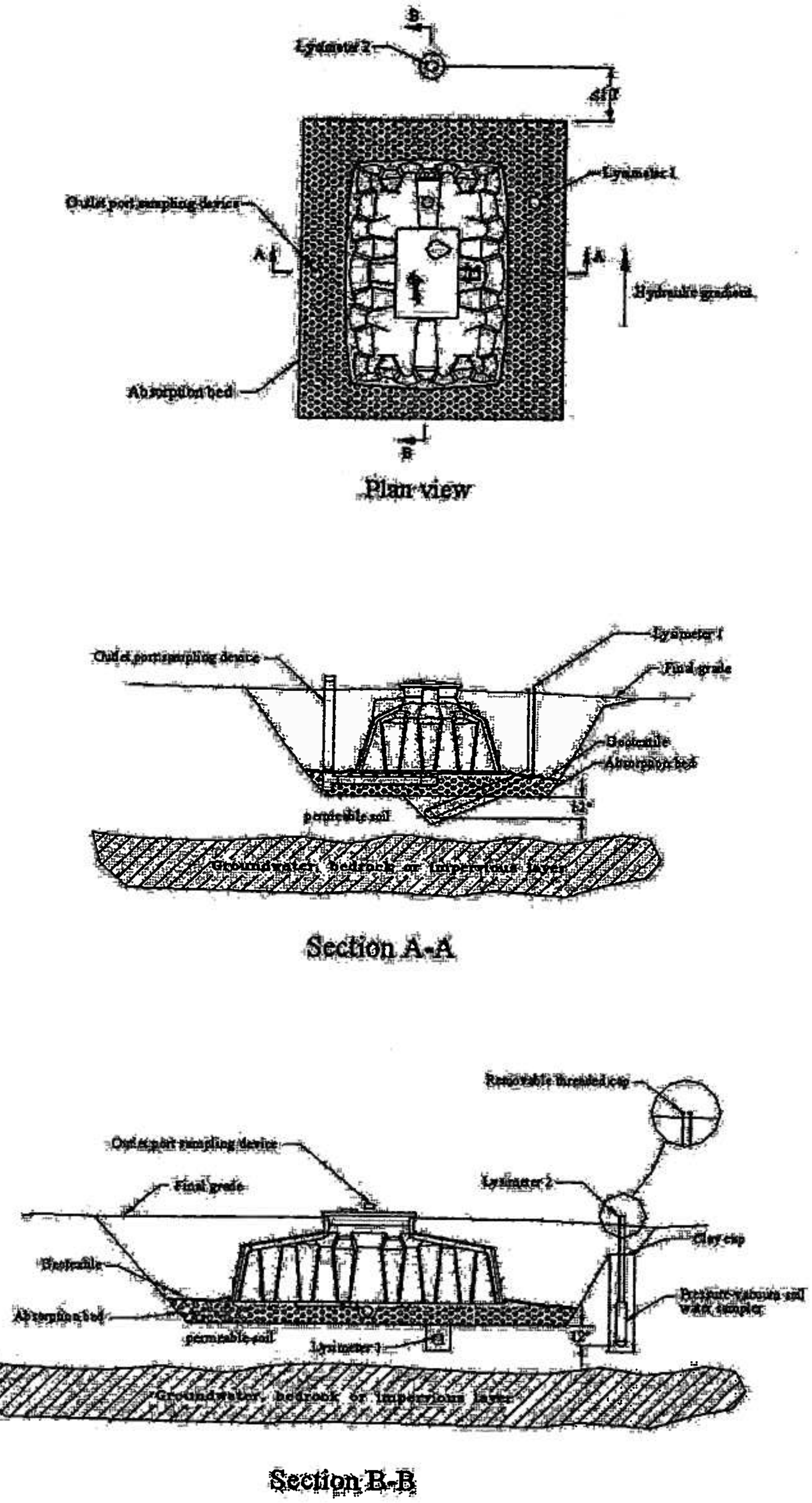


**Figure 1 Typical installation of the Ecoflo® Biofilter and the monitoring equipment**

Finally, all sites were equipped with an event counter that counted the number of tipping bucket events of the Ecoflo® distribution system. This will allow the evaluation of the total volume of wastewater that is treated by the Ecoflo® unit and discharged into the soil.

**Results**

At present, the monitoring has been completed for 18 sites which correspond to soil types I, II and III. Monitoring is ongoing for the two sites installed in type IV soils. The mean flow reached for the sites corresponds to a value of 156 gpd. Additionally, 90% of the measured values were lower or equal to 250 gpd. Lastly, it should be noted that, in some cases, the flow rates have exceeded the capacity of the system with values reaching up to 800 gpd. The $CBOD_5$ and TSS concentrations obtained at the Ecoflo® Biofilter's outlet are presented in Table 2. We observe that the biofilter's performance reached during the Virginia monitoring correspond to the values already measured in actual conditions by Premier Tech Environment (PTE) and during the various testing completed for the certification of the technology.

**Table 2 Performance of the Ecoflo® Biofilter in TSS and $CBOD_5$**

| Parameters | Virginia | | PTE's monitoring (11 years, n = 163) | | Certification (Ecoflo® effluent) | | |
|---|---|---|---|---|---|---|---|
| | STE | Ecoflo® Effluent | STE | Ecoflo® Effluent | CSTB (n = 30) | BNQ (n = 118) | NSF (n = 108) |
| TSS (mg/L) | 34 ± 23 (n = 141) | 6 ± 7 (n = 337) | 52 ± 48 | 4 ± 3 | 5 ± 4 | 2 ± 0.2 | 2 ± 0.7 |
| $CBOD_5$ (mg/L) | 186 ± 113 (n = 340) | 8 ± 8 (n = 337) | 176 ± 89 | 5 ± 5 | 3 ± 2 | 2 ± 0.4 | 2 ± 0.3 |

The evaluation sampling done in the absorption bed has demonstrated the potential of native soil to reduce nitrogen, phosphorus and fecal coliform still present in the effluent treated by the Ecoflo® Biofilter. The mean concentrations measured at the outlet of the different steps of the treatment are presented in Table 3. Table 4 shows the values corresponding to the 90% confidence level, which is an indication of the stability of the performance results percentile. The absorption bed composed of a 12-inch layer of soil fed by the Ecoflo® Biofilter's effluent is effective in polishing the treated effluent to a high quality level.

**Table 3 Mean efficiency (± standard deviation) of the Ecoflo® Biofilter & absorption bed**

| Parameters | STE | Ecoflo® Effluent | Effluent at 12 inches underneath Ecoflo® (L1) | Groundwater (L3) | Performance | | |
|---|---|---|---|---|---|---|---|
| | | | | | Ecoflo® | Absorption bed | Overall |
| Total nitrogen (mg/L) | 45 ± 24 (n = 72) | 32 ± 18 (n = 76) | 8 ± 9 (n = 77) | 4 ± 4 (n = 40) | 29% | 78% | 84% |
| Total Phosphorus (mg/L) | 5.9 ± 0.9 (n = 11) | 5.2 ± 0.9 (n = 11) | 0.12 ± 0.04 (n = 15) | - | 12% | 97% | 98% |
| Fecal coliform (CFU/100 mL) | 34,262 (n = 51) | 1,029 (n = 308) | 2 (n = 336) | - | 1.5 log | 2.7 log | 4.2 log |

**Table 4 90% Confidence level for total nitrogen, total phosphorus and fecal coliform**

| Parameters | STE | Ecoflo® Effluent | Effluent at 12 inches underneath the Ecoflo® (L1) | Groundwater (L3) | Performance | | |
|---|---|---|---|---|---|---|---|
| | | | | | Ecoflo® | Absorption bed | Overall |
| Total nitrogen (mg/L) | 77 | 56 | 22 | 10 | 27% | 61% | 71% |
| Total phosphorus (mg/L) | 7.1 | 6.5 | 0.2 | - | 8% | 97% | 97% |
| Fecal coliform (CFU/100 mL) | 240,000 | 34,300 | 2 | - | 0.8 log | 4.2 log | 5.1 log |

**Total nitrogen removal**

The result analysis presented in Tables 3 and 4 demonstrates that the system comprising an Ecoflo® Biofilter and a 12 inches thick absorption bed allows reduction in total nitrogen content by 84%. It is important to mention that this performance does not take into account the total nitrogen already present in the groundwater table upstream of the absorption bed (4 ± 4 mg/L). Considering this background, we realize that the total nitrogen concentration at a 12 inches soil depth underneath the biofilter is in average lower than 5 mg/L, which exceeds the mean requirement of 10 mg/L established by different regulations. The performance observed would be attributable to a good nitrification of the wastewater in the biofilter (more than 80 %) and to the presence of anoxic micro-zones in the soil matrix allowing denitrification of effluent with the presence of soluble carbon leached from the peat base filtering media.

*Fecal coliform*

The receiving environment composed of 12 inches of native soil allows fecal coliform reductions under the detection level of 2 CFU/100 mL and achieves this for 90% of the results. The 336 values measured are lower than the usual limit of 200 CFU/100 mL, with the maximum value corresponding to 170 CFU/100 mL. This attenuation of the fecal coliform bacteria is associated to the retention/fixation phenomenon at the soil particle surface and at the change in physico-chemical conditions of the soils. It is also important to mention that the geometric mean of the fecal coliform observed at the outlet of the Ecoflo® Biofilter during this study, which is 1,029 CFU/100 mL, corresponds to the results obtained in other monitoring and testing. Indeed, an average concentration of 1,000 CFU/100 mL has been reached within the context of voluntary PTE 's monitoring of around 80 sites in the field since 1995, with 1,250 CFU/100 mL during the BNQ certification testing and with 630 CFU/100 mL at the NSF site.

*Total phosphorus*

The Ecoflo® Biofilter reduces total phosphorus by 12% on average. However, the combination of the biofilter with a polishing field composed of a layer of at least 12 inches of native soil, allows an overall removal of 98% of the total phosphorus present in the septic tank effluent. The total phosphorus mean concentration at a 12 inch depth in the native soils equals 0.12 mg/L and 90% of the values are equal or lower than 0.2 mg/L. Remember that the usual disposal criteria correlates to 1.0 mg/L. These results have been obtained in the soil type I to III installations in operation for more than 40 months and no influence has been noticed with the permeability of the soils used. As per the existing literature, the phosphorus fixation in acid soils is mainly associated with its adsorption to the surface of the metallic elements (iron and aluminum) present in the soil (Pellerin and al. 2006). According to this reference, the acid to neutral soils can be classified in three groups as far as their ability to retain phosphorus (low capacity: 1.46, mean capacity: 3.04 and high capacity: 5.66 gP/kg of soil). The results have been obtained after analysis of more than 275 soil samplings covering 75 series of soils in a horizon varying between 0 and 30 inches of the surface. Analysis performed on the soils experimented in Virginia present results of the same order which is a retention capacity of the phosphorus of 3 gP/kg of soil for the three soil types used. The access to this phosphorus retention capacity of the soil and the stability of the phosphorus retained, depend on the following key factors: the quality of the effluent infiltrated to prevent clogging of the receptor soil; the withholding of a high redox potential (aerobic

The hydrodynamic and physico-chemical characteristics of the effluent produced by the Ecoflo® Biofilter facilitate phosphorus fixation in the soil. Indeed, the peat based filtering media releases humic and fulvic acids that cause soil particle alteration in the unsaturated soils, which increases the iron and aluminum availability in the soil to react with phosphorus. Also, the low pH conditions prevailing in the filtering media during its start-up phase allow release of iron and aluminum present in the peat, thus creating an additional doping of these metals in the receptor soil. Furthermore, the retention capacity associated with the peat-based filtering media ensures peak flow attenuation which is translated by a regulation of the flow infiltrated, thus facilitating the unsaturation of the soil. These unsaturated conditions are also maximized by the pulsed feeding to the biofilter, creating repeated wetting/draining cycles bringing air to the soil. Notice that the air present in the gravel area at the base of the biofilter is renewed by the aeration process integrated into the system. Lastly, to maintain this access to the soil capacity to fix phosphorus, it is essential that the treated effluent to be infiltrated presents an excellent quality in all conditions (variations in flow and in loads, start-up following a prolonged stop, etc.), in order to prevent clogging of the soil by the suspended matter and introduction of too large concentrations of organic matter. These two last factors diminish the redox potential in the soil and reduce soil capacity to fix phosphorus. As demonstrated during testing with particular stresses (NSF, 2005 and BNQ, 2005), the Ecoflo® Biofilter produces an effluent with low variation in TSS and $BOD_5$ concentrations (lower than 5 mg/L) even in peak conditions or at start-up after a many days period without system feeding zero flow. On the basis of the previous data, of the occupancy rate of the residences and of the quantity of phosphorus produced by the occupants, we estimate that the system « Ecoflo® Biofilter + 12 inches of native soils » allows retention of the phosphorus produced by a residence for a duration of at least 20 years in the majority of the cases, without taking into account the contribution in iron, aluminum, humic and fulvic acids associated with the peat-based filtering media.

**Conclusion**

The results obtained within the context of the independent study conducted in the State of Virginia demonstrate an interesting potential for the system comprising an Ecoflo® Biofilter followed by a 12 inches thick layer of soil for the removal of nitrogen, phosphorus and fecal coliform to levels of usual disposal requirements of less than 10 mg/L for total nitrogen, 1.0 mg/L for phosphorus and 200 CFU/100 mL for fecal coliforms. In accordance with the recommendations of experts in the decentralized sanitation field (Tchobanouglos, 2003), this study clearly demonstrates the importance of reserving natural soils for the polishing of an effluent having undergone a high level of treatment and presenting low quality variations. The use of soil for the treatment of primary or secondary effluent presenting variations would not allow exploitation of the full tertiary treatment potential of this natural matrix. On the basis of these promising results, experimentation goes on to optimize the approach in different conditions, in order to maximize the longevity of the system.

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# GLOSSARY OF TERMS

| Terms | Definitions |
|---|---|
| **Aerobic** | • Characterizes micro-organisms that need air or dissolved oxygen in water for their development. |
| **Anaerobic** | • Characterizes environments that lack dissolved oxygen and also organisms that can develop in the absence of air or oxygen. Oxygen is depleted from the deep waters of Gilmour Bay in late summer, resulting in an anaerobic environment. Fish are unable to survive in such conditions. |
| **Anthropogenic** | • Relating to or involving the impact of humans. |
| **Chlorophyll *a*** | • The green photosynthetic pigment in algae; its presence depends on physical, chemical and biological conditions that govern its production at any time. |
| **Clinograde** | • A zone or line of rapid or sudden decrease in concentrations of dissolved oxygen in lakes. |
| **Dissolved oxygen** | • The amount of oxygen dissolved in a given volume of water, typically expressed in milligrams per litre. |
| **DOC** | • See **Section 2.6**, page 14. |
| **Epilimnion** | • The uppermost layer of water in a lake characterized by essentially uniform temperature, generally higher than elsewhere in the lake, and by a relatively uniform mixing caused by wind and wave action. Also, the less dense oxygen-rich layer of water that overlies the metalimnion in a thermally stratified lake. |
| **Eutrophication** | • See text **Section 2.5**, page 12. |
| **Hydrogeology** | • The science of underground water. |
| **Hypolimnion** | • The lowermost layer of water in a lake with an essentially uniform temperature gradient (except during a turnover), generally colder than elsewhere in the lake, and often relatively stagnant or oxygen-poor. Also, the dense layer of water below the metalimnion in a thermally stratified lake. |
| **Lacustrine** | • Related to, pertaining to, produced by, or formed in a lake or lakes. |
| **Lake turnover** | • The complete, wind-induced, top-to-bottom circulation of water in deep lakes occurring when the density of surface water is the same or slightly greater than at the bottom. In temperate zones, lake turnover occurs in spring and fall. |

| Terms | Definitions |
| --- | --- |
| **Lake classification** | • One of the more commonly used lake classification systems recognizes two general categories of lakes; dystrophic lakes with brown coloured water, rich in humic materials derived from plants, and oligotrophic-eutrophic lakes with "unstained" water.<br><br>*Oligotrophic* lakes are poorly supplied with plant nutrients and support little plant growth. As a result, biological productivity is generally low, the waters are clear, and the deepest layers are well supplied with oxygen throughout the year. Oligotrophic lakes tend to be deep, with average depths greater than 15 metres (49 feet) and maximum depths greater than 25 metres (80 feet).<br><br>*Mesotrophic* lakes are intermediate in characteristics between oligotrophic and eutrophic lakes. They are moderately well supplied with plant nutrients and support moderate plant growth.<br><br>*Eutrophic* lakes are richly supplied with plant nutrients and support heavy plant growths. As a result, biological productivity is generally high, the waters are turbid because of dense growth of phytoplankton, or contain an abundance of rooted aquatic plants; deepest waters exhibit reduced concentrations of dissolved oxygen during periods of restricted circulation. Eutrophic lakes tend to be shallow, with average depths less than 10 metres (33 feet) and maximum depths less than 15 metres (50 feet). |
| **Lake trout spawning** | For inland Precambrian Shield lakes, lake trout spawning shoals are typically composed of rubble and boulders (4 centimetres to 40 centimetres in diameter) to a depth of greater than 0.3 metres, and are within about 10 metres from shore. Eggs are deposited in the fall, and small fish (fry) emerge in the winter. Young fish may remain within the boulders where they hatched for up to one month before seeking deeper water. A drawing showing locations of spawning shoals in Chandos Lake is attached. |
| **Landscape naturalization** | • The process of replanting shoreline property with indigenous trees and shrubs to: help prevent nutrients from entering a water body; provide nearshore fish habitat; protect habitat for small mammals, birds and in some cases deer; and maintain shoreline aesthetics. |
| **Limnology** | • The study of inland waters (running and standing waters, both fresh and saline), including their biological, physical, chemical, geological and hydrological aspects. |
| **Mechanistic reaction** | • Herein, refers to physical and chemical binding processes of phosphorus with iron and aluminum in "B" Horizon Precambrian Shield soils, the reactions of which are permanent. |

| Terms | Definitions |
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| **Metalimnion** | • The intermediate zone (mid-depth, metalimnetic layer) of lake water, where the temperature declines rapidly as the depth increases. The metalimnion is frequently referred to as the thermocline. |
| **Mean Volume Weighted Hypolimnetic Dissolved Oxygen** | • A method used to quantify habitat of coldwater fish species such as lake trout. Oxygen profiles are used in conjunction with individual lake strata volumes to derive a single volume-weighted oxygen value for a lake. The Ministry of Natural Resources policy of 7.0 mg/L is considered necessary to sustain healthy coldwater fish species. |
| **Organic phosphorus** | • Phosphorus derived from organic matter in the surface (humus) soil layer, which surrounds water bodies. Higher percentages are typical of uncultivated forest soils. |
| **Orthograde** | • Dissolved oxygen is not depleted with an increase in lake depth. |
| **Phosphorus** | • A naturally occurring, non-metallic element which promotes the growth of algae and weeds in a body of water. Septic systems can be a significant source of phosphorus to recreational lakes (see **Section2.5**, page 12). |
| **Phytoplankton** | • Collectively, all the microscopic plants living unattached, suspended in the water of aquatic habitats. Algae are phytoplankton. |
| **POC** | • Particulate organic carbon (see **Section 2.6**, page 14). |
| **Precambrian Shield lake** | • Lakes, which when on the Precambrian Shield (comprised of igneous or metamorphic rock), have similar water body characteristics. The lakes are often classified as nutrient poor and may support lake trout. Such lakes are usually subject to higher levels of protection and control. |
| **Precipitation with aluminum and iron** | • Refers to the process of phosphorus combining with aluminium and iron rich soils in septic fields. The process is permanent in acidic soils that typically characterize Ontario’s Precambrian Shield. |
| **PWQO** | • Provincial Water Quality Objectives, as set out in Water Management: Policies, Guidelines, Provincial Water Quality Objectives of the Ministry of Environment and Energy. July 1994. 32 pages. |
| **Secchi disc measurement** | • Measurement (depth in metres) of water clarity to which a black and white disc is visible when submerged on the shaded side of a boat. The measurement determines the depth to which light penetrates in water, determined as twice the Secchi disc depth. |
| **Spring turnover** | • Refer to lake turnover. |
| **Stratum** | • Layer of water in a lake as determined by morphometry and temperature. |
| **Total phosphorus** | • Includes dissolved and particulate forms of phosphorus. |

| Terms | Definitions |
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| **Trophic state** | • Characterization of a body of water in terms of position in a scale ranging from oligotrophy to eutrophy. |
| **Water clarity** | • Depth in metres to which one can see through the water, as determined through the use of a Secchi disc. |
| **Zooplankton** | • Microscopic animal life found unattached in a body of water. They include small crustaceans and single celled animals (i.e., protozoa). |