

Sources and Mechanisms of Delivery of *E.coli* (bacteria) Pollution to the Lake Huron Shoreline of Huron County

Interim Report:

Science Committee to Investigate Sources of Bacterial Pollution of the Lake Huron shoreline of Huron County

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The Science Committee to Investigate Sources of Bacterial Pollution of the Lake Huron Shoreline of Huron County¹

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¹ The committee was initiated by the Ontario Ministry of the Environment, in consultation with the Ontario Ministry of Agriculture and Food and Environment Canada, as a collaboration among technical specialists from provincial, federal and local agencies to work towards a science-based examination of the sources of *E. coli* and fecal pollution to the Lake Huron shoreline of Huron County impacting upon water quality at recreational beaches.

Executive Summary

Overview

In 2004, the Lake Huron Science Committee (LHSC) was initiated by Ministry of the Environment (MOE), in consultation with the Ontario Ministry of Agriculture and Food (OMAF) and Environment Canada (EC), in response to public concerns over bacterial (*E. coli*) pollution along the southeast shoreline of Lake Huron. The committee was created as an MOE-led, multi-agency technical committee to:

- i) identify sources of *E. coli* (as an indicator of fecal pollution) to the shoreline of Lake Huron,
- ii) investigate the extent of influence of fecal pollution sources on the shoreline, and,
- iii) make recommendations on possible actions to address the microbial pollution of fecal origin.

The approach was to develop a time-limited science work plan, and to identify partners to conduct the necessary monitoring and analysis for the identification of *E. coli* contamination and the associated fecal pollution sources. The LHSC adopted a phased strategy:

Phase 1: Review and Synthesis of Existing Information

Phase 2: Development of Additional Studies

Phase 3: Development of Recommendations

The committee is chaired by the MOE with memberships from other agencies and local organizations including OMAF, EC-National Water Research Institute (NWRI), the Maitland Valley Conservation Authority (MVCA), the Ausable-Bayfield Conservation Authority (ABCA) and the Huron Country Health Unit (HCHU).

The geographic extent of this study was limited to the Lake Huron shores of Huron County. Within this larger study area, a sub-section of the shoreline was selected for more detailed analysis of shoreline water quality and environmental conditions. The approximate 45 km of shoreline from Amberley Beach (Point Clark area) to Bayfield Beach includes a wide range of physical conditions and land-use features and is thought to be representative of much of the southeastern shores of Lake Huron. The Huron County shoreline also encompasses the five Lake Huron beaches permanently posted as unsafe for water recreation for the 2003 season by the HCHU. A primary objective was to undertake a detailed analysis of a limited geographic

area where efforts could be intensified and presumably a more robust synthesis of information could be achieved. A secondary objective was to develop a frame of reference for examining microbial pollution of fecal origin at beaches which would have a general applicability to the SE shores of Lake Huron as a whole.

This interim report presents the progress of the committee on Phase 1. The report presents a summary of existing information and attempts to provide a synopsis of the state of understanding of microbial pollution of fecal origin on the Huron County shores of Lake Huron, as inferred primarily from results for the fecal pollution indicator species *E. coli*. The approach taken was to examine information from three perspectives:

- i) the potential sources of microbial pollution of fecal origin to the shoreline were identified and described,
- ii) evidence for impacts on the shoreline by the various potential sources was reviewed, and
- iii) the potential mechanisms of delivery of microbial pollutants to the shoreline were explored.

Synopsis: Phase 1

A diverse range of historical and recent information was examined. These included bacterial pollution studies conducted by the MOE from the mid-to-late 1980s, bacterial pollution studies by Maitland Valley and Ausable-Bayfield Conservation Authorities, reviews and analysis of environmental conditions on the shores of Lake Huron by various groups, Huron County Health Unit beach monitoring data for 1993 to 2003, water quality data collected by local shoreline residents' associations and water quality monitoring by the MOE and other agencies in 2003. Selected background information is provided on the physical setting of the area of study and on limnological features of Lake Huron to assist in the interpretation of how microbial pollution is potentially expressed at the shoreline of the lake.

Microbial pollution of fecal origin along the southeast shoreline of Lake Huron, as inferred from levels of the fecal indicator *E. coli*, is an ongoing problem. Documented concerns related to microbial pollution of the shores of Lake Huron go back at least 40 years and have been the subject of work by many organizations and groups of people. In 1978, the International Reference Group on Great Lakes Pollution from Land Use Activities (PLUARG) identified

bacteria of public health concern as a Great Lakes water quality problem. At the time, the problem was described as a point source problem affecting nearshore or local areas of the Great Lakes at large and the final PLUARG report recommended that epidemiological evidence be examined to establish criteria for body contact recreational use of waters receiving runoff from urban and agricultural sources.

Historically, the MOE has been active in the environmental monitoring of the nearshore waters of Lake Huron, particularly in the early 1980s when beach postings in 1983 resulted in significant public concern. A series of extensive investigations in 1984 and 1985 attempted to quantify levels of fecal indicator bacteria at beaches, assess whether correlations with meteorological or physical variables could be made, and identify sources of fecal pollutants to beaches.

A number of observations, and some correlations were determined. It was noted that peak bacterial levels occurred during rainfall events when field soils were wet from previous rainfall, whereas bacterial levels did not increase significantly when rainfall events occurred on dry soils. Wind direction and increased wave action were correlated with increased fecal coliform (FC) counts and were likely related to FC counts through their role in the resuspension of lake sediment and the location of tributary plumes when present. Several Lake Huron tributaries and agricultural drains were examined more intensely to determine their role as potential sources of bacterial contamination to the beach areas. During this study, all drains and creeks sampled contained elevated levels of bacteria, particularly during rainfall events. Isolated situations of fecal contamination to Lake Huron tributaries were identified and remediated, including moving cattle away from creeks, and plugging a storm sewer that was connected to a sanitary line. These actions reduced bacterial levels. Several fecal input sources along small agricultural drains were also identified but no remedial action was taken. Additionally, little correlation between FC and number of swimmers, gulls, pets and temperature was noted but these observations were not discussed in any detail.

The use of *E. coli* as an indicator of fecal pollution in water is based on the assumption that *E. coli* does not survive long in aquatic environments. While this holds true in drinking water, where the levels of particles and nutrients are low and disinfection is common, there is increasing evidence that *E. coli* may survive longer than previously thought in surface waters and sediments of lakes and rivers. A component of the MOE studies was the examination of bacterial survival under environmental conditions. Diffusion chamber studies demonstrated that

E. coli survival in nutrient rich waters could exceed 3 weeks and that *E. coli* likely survive in excess of two months in sediments. Studies confirmed that bacteria could survive for extended periods of time adsorbed to suspended sediment particles and this process was likely an important transport vector for these organisms. In fact, prolonged survival and extensive transport of *E. coli* on suspended particles was demonstrated in an agricultural drain. The implications of these findings, which are now well recognized, is that elevated levels of fecal indicator bacteria in lake water may not necessarily reflect recent fecal pollution. The studies also demonstrated that bacteria of fecal origin could be transported for some distance via an agricultural creek or drain distal to the beach.

The early MOE studies also included work to elucidate the sources of fecal indicator bacteria in beach water using state of the art microbial source tracking techniques for the time (fecal coliform to fecal streptococci ratio, antibiotic resistance, serotyping). These efforts suggested that bacteria of agricultural origin were present, although, possible human origins could not be ruled out.

In the mid 1980s, the MOE developed the Provincial Rural Beaches Strategy Program (PRBSP) in which the Maitland Valley (MVCA) and Ausable Bayfield (ABCA) Conservation Authorities participated (the CAs with jurisdiction in Huron County). The objective of the program was to improve microbiological water quality at rural beaches in the Province with a focus on agricultural sources of fecal pollution. Over an approximately ten year period, the MVCA and ABCA conducted water quality studies and lead remediation programs in association with the PRBSP, the best known of which is the Clean Up Rural Beaches (CURB) program.

The initial component of CURB (late 1980s) was the development of the CURB plan which was intended to identify the relative impacts of fecal pollution sources and their modes of delivery to beaches, provide an analysis of remediation priorities and provide options culminating in a remedial strategy for identified pollution sources. During the development of the CURB plan, four potential agricultural sources of pollution were routinely identified through farm surveys: 1. runoff from barnyards and manure piles, 2. inadequate milkhouse waste disposal, 3. unrestricted cattle access to water courses, and, 4. winter manure spreading. Sources of fecal bacterial pollution (as inferred from the indicator fecal coliform) and their relative magnitude were identified in the MVCA and ABCA watersheds through a modeling exercise based on the application of the Pollution from Livestock Operations Predictor (PLOP) model developed by Ecologistics Ltd. The model applied by the conservation authorities predicted that the most

important sources of fecal pollution were: 1. septic system failure, 2. winter manure spreading, and 3. livestock access to water courses. The model predicted that continuous sources were more significant than pulse- or event-related sources and those sub-basins of the watershed closest to the lake contributed proportionately more to lake bacterial pollution. As such, remedial strategies targeting pollution sources based on lake-proximity would reduce loads most effectively. A number of the assumptions key to the predictions generated by the modeling exercise are uncertain and consequently the reliability of the analysis can be questioned. Little model verification or sensitivity analysis appeared to have been conducted. The predictions from the modeling exercise should be viewed with caution.

Despite support and funding under the CURB program, only a limited number of the estimated sources of bacterial contamination were addressed over the course of CURB implementation (1991-1996). CURB implementation also included a water quality monitoring component designed to assess changes at selected remediation sites. Although some remediation projects, including a tile drain diversion, septic system replacements, milkhouse waste and manure storage projects resulted in reduced *E. coli* levels, overall results of remediation strategies remained inconclusive. The ability to detect improvement may have been hampered by high system wide variability in water quality attributable to other sources.

The Huron County Health Unit monitors levels of fecal indicator bacteria at recreational beaches in Huron County for the purpose of providing advice to the public on beach water quality. Data on the occurrence of the fecal indicator bacteria *E. coli* on the shores of Lake Huron collected by the HCHU as part of its beach monitoring program was examined for this report to provide insight on: i) levels of occurrence of fecal indicator bacteria in shoreline waters, and, ii) patterns of occurrence in relation to environmental factors suggestive of possible sources or modes of delivery of fecal pollution to the shores of the lake. Data collected by the HCHU between 1993 and 2003 for ten beaches ranging from Point Clark (Amberley Beach) to Bayfield were examined in detail. While year to year variability was appreciable, no systematic trends among years could be established in levels of *E. coli*. It was observed, at all beaches, that *E. coli* levels varied widely over the recreational season and that while some days all or most beaches had relatively poor conditions indicating large scale effects, on many days only a subset of the beaches had poor conditions indicative of local sources. During times of large scale effects, no single environmental variable was consistently correlated to *E. coli* levels across the beaches. However, some significant variables were identified, including heavy precipitation, wind speed and to a lesser extent, wind direction and water turbidity. The best correlate of within-season

E. coli variability was wind speed. There may have been a weak negative correlation between *E. coli* and the frequency of upwelling events, measured as rapid changes in lake temperature of at least 4°C, and there was a weak positive relationship between *E. coli* and elevated turbidity. Indicators of land-based effects on the nearshore of the lake (major tributary discharge, precipitation, nearshore nitrate levels) were not consistently related to patterns in occurrence of *E. coli* at beaches.

Analysis of the beach monitoring data suggests that sources of fecal pollution differ between beaches. The physical settings and the anthropogenic features of beaches and the associated drainage areas vary substantially among locations. The analysis provided a sense that shoreline areas likely need to be examined individually with respect to dominating sources of fecal pollution since no environmental factor was found to overwhelmingly account for the wide variability in *E. coli* levels among beaches. At several beaches there have been years in which the frequency of data where *E. coli* levels have been above the recreational Provincial Water Quality Objective (100 cfu/100mL *E. coli*) has been high (>50% of geometric means of daily sample sets).

Numerous tributaries of widely varying drainage areas and hydrologic features discharge to the southeast shores of Lake Huron. There is strong evidence of frequent occurrence, and at times, appreciable levels of *E. coli* in tributaries, yet previous studies have rarely related the discharge of tributary water in the lake with water quality impacts at the shoreline. Overall, there is little understanding of the spatial extent or temporal frequency of the impacts of tributaries on the occurrence of *E. coli* in shoreline waters. The role of tributary discharge volume is ambiguous. For example, the years 2000 and 2001 were high and low tributary discharge years, respectively, yet both years showed elevated levels (relative to other years) of *E. coli* at recreational beaches. Similarly, responses to rainfall were very variable between beaches. However, on the limited number of occasions when *E. coli* was elevated at most beaches, rainfall and wind appeared to be the best correlates of such events.

Previous studies in the Lake Huron basin conducted by the MOE and Conservation Authorities provided evidence that a variety of agriculturally-based activities likely load fecal pollutants to surface waters draining to Lake Huron. The widespread occurrence of elevated levels of *E. coli* in tributaries over predominately rural lands provides circumstantial evidence that agriculture-based sources contribute to *E. coli* levels observed in surface waters draining to Lake Huron. Despite the undoubted potential of agriculturally-derived pollutants to reach the lake, there is

substantial uncertainty as to the extent, and under what conditions, loads of *E. coli* originating from agricultural activity and delivered by tributaries to the lake impact upon beach water quality. Furthermore, there is no suggestion of a progressive increase in *E. coli* levels as the water moves through tributaries, an observation which can be derived from monitoring data collected by several organizations for multiple years.

Analysis in the MVCA and ABCA CURB plans, predicted loads of fecal bacteria (fecal coliforms) from septic systems as the single largest source of fecal pollution to the shoreline and identified concern that failure of septic systems may be contributing to the fecal pollution of the tributaries and shoreline waters of Lake Huron. Failure was used in a broad sense as release of fecal bacteria to surface water due to shortcomings in design, operation and maintenance of septic systems. While there is reason to question the validity of the predictions of the CURB plan analysis concerning contributions from septic systems because of the assumptions used in the calculation, there is little doubt that homes that rely on private septic systems are abundant in the lands draining to SE Lake Huron and particularly on the lake fringe which is valued as recreational property. However, as noted in a 2004 report examining historical water quality information for SE Lake Huron by the Lake Huron Centre for Coastal Conservation, there is limited knowledge of the septic systems in the area and a comprehensive assessment of their potential impact on water quality is required. There have not been any studies undertaken to either track the migration of bacteria from septic systems adjacent to the beaches of Lake Huron to the shoreline, to determine how far bacteria migrate from these particular septic systems, or simply to determine if bacteria from these septic systems are reaching the lake. Thus very little is actually known. However, some factors have been identified that can enhance the potential for septic systems to contaminate shoreline water such as: 1) close proximity (15-20m) of septic systems to creeks and drains; 2) increased waste load to septic tanks due to year round, instead of seasonal, residences; 3) infiltration of shoreline wave runup on the water table; 4) residence clusters all requiring septic systems; 5) favorable (fine particles for attachment, nutrients and stable temperature) *E. coli* growing environment in groundwater.

Likely, the impact of failing septic systems will be mediated by groundwater impacts. While it is unknown whether significant inputs of *E. coli* to the shoreline are occurring through groundwater inputs, recent studies in Lake Huron suggest *E. coli* may be elevated in beach pore waters collected in shallow pits below the swash zone in comparison to open water adjacent to the beach and below the backshore area of the beach. These results suggest that *E. coli* delivered to the beach swash zone by groundwater, lake-water infiltration and/or other processes may be

stored in the beach sand adjacent to the lake. These contaminated pore waters and particle matrices may then be acting as vectors for contaminating lake water at the shoreline. Overall, the deposition and erosion rates of beaches will likely directly influence the occurrence of bacteria in both beach sand and lake water, however, more information is needed to understand how these processes affect occurrence of fecal indicator bacteria at the shores of the lake.

The overall level of urbanization over the lands draining to the Lake Huron shores of Huron County is low, however, there are a number of developed and urbanized areas where municipal or communal systems are used to collect and treat sewage and waste water. While the effluent of most of these sewage treatment plants provide primary treatment (settling of solids), secondary treatment (biological degradation of organic matter and nitrogen) and disinfection to comply with a 200 cfu/100 mL effluent guideline, discharge from these facilities can represent a significant source of microbial load to the environment. An additional STP-related load of microbes is found as sewage treatment bypasses and overflows which occur when the hydraulic capacity of the STP is exceeded during periods of heavy rain or snow melt when wet weather flows enter the sewerage system. The periodic release of raw sewage or partially disinfected sewage can contribute high concentrations and possibly large loads of *E. coli* to the environment at points of release. While the overall significance of unintentional releases of sewage by treatment facilities is unclear, there is evidence that such releases have occurred in the past at several facilities for which information was examined. The Town of Goderich sewage treatment facility is the only facility with a direct discharge to Lake Huron. The shore outfall is adjacent to a public beach but separated from the beach by a made-made barrier which serves to separate the beach from the outfall. The Maitland and Bayfield River serve as the conduits to the shores of Lake Huron for discharges from several STP plants within the drainage area of these rivers.

The lands draining to the southeastern shores of Lake Huron are, however, overwhelmingly agricultural lands. In fact, Huron County is one of Ontario's leading livestock producing regions. Research from other regions suggests that the quantity, rates and timing of shedding of microorganisms in manure can be expected to vary between livestock commodities and even between farms with the same animal type. Trends over recent decades include an increase in the fraction of larger, confined animal farming operations in the area including pigs, cattle and poultry livestock industries. There are numerous different manure handling and application processes in the area, and to date there has been limited research on manure microbial persistence and transport in the area of study for this report. It is known, however, that the field

application of manure is a significant pathway of fecal pollutants to the environment and the type of manure applied and the process by which it is applied will affect the transport of microbes to surface waters.

Fecal pollution from wildlife sources has been documented to contaminate drinking and recreational waters across Canada and on this basis there is a need to assess the potential for contributions of fecal pollutants from wildlife species to shoreline areas where beach water quality is problematic. The microbial loads generated by different animals are highly variable. Behavioral aspects are also significant – few mammals are likely to be found directly on beaches, except perhaps companion animals such as dogs. On the other hand, birds are found on Lake Huron beaches and there is evidence from other areas of the impact of birds on water quality. Wildlife living within the watershed may also contribute to tributary loads of fecal pollutants especially when animals use habitat near a watercourse such as riparian buffer strips. However, little is known of the actual contributions of *E. coli* to the shoreline of Lake Huron by wildlife populations over the area examined in this report.

Drains conducting surface runoff and terminating on or at the edge of beaches are commonplace along the Lake Huron shoreline over the study area examined in detail. The origin of the flows in these drains, which is likely runoff from residential properties adjacent to the shoreline, is confounded in some cases by the close proximity of the drains to agricultural lands. Because of the proximity of drain outlets to beaches, the periodic flows in the drains should be considered as potential sources of microbial input to beaches and adjacent lake water. Direct discharge of storm water to the lakeshore from urban or extended residential areas is limited to the town of Goderich and the community of Bayfield which in both cases there is a storm sewer outlet in proximity to recreational beaches. While little information could be found on the microbial quality of discharge from the storm sewers, they represent a potential source of fecal pollution to the shoreline.

Conclusions

The wealth of information acquired has provided a basis to identify the likely sources of bacterial pollutants to the shores of Lake Huron. However, there still remain significant gaps in the information precluding a clear causal link between potential sources and impaired beach water

quality. The key gaps may be summarized as follows:

1. Scale of effect of these sources (spatial and temporal).
2. Relative impact of local sources at the beach.
3. Relative impact of fecal loads coming from sources upstream from the watershed.
4. Role of bacterial adsorption to sediment in survival and transport to the beach.

Microbial water quality (as inferred from indicator bacteria) on the shores of Lake Huron is likely impacted by multiple sources of fecal loading. A difficulty which has been evident throughout the analysis is that the relative significance to shoreline water quality of local (beach-scale) versus regional (drainage basin-scale) sources is not understood. Confirming impacts of potential sources and determining the relative impacts of the sources of fecal pollutants on shoreline water quality remains a significant technical challenge. Areas of the shoreline may potentially need to be examined on a case by case basis.

Many of the potential sources of fecal contamination have been identified. However, little is understood regarding the mechanisms by which key potential sources impact the beach, particularly with respect to survival of the *E. coli* in the environment and transport to the beach.

It is through an improved understanding of the linkages between the loading of the pollutants to the lake and the effects on water quality at recreational beaches that the dominant, and/or the most environmentally manageable, of the drivers of adverse water quality may be appreciated. The relative extent of local, beach-scale sources compared to broader regional influences on microbial quality at the shoreline of Lake Huron remains an important question, the answer to which may potentially shape the geographic scale on which remedial measures are implemented. There are several facets of this issue which require further clarification and could potentially promote effective targeting and prioritization of remedial actions.

In the absence of a strong basis to interpret the causes of adverse water quality at recreational beaches on the shores of southeastern Lake Huron it is uncertain what the outcome of future remedial actions to reduce fecal pollution will be on beach water quality. The analysis conducted for this report suggests that in the future there will likely be periods of adverse water quality at the beaches examined and that without an investment in effort to better understand the sources of the impacts there will be little ability to manage water quality at the beaches or the economic repercussions resulting from the postings of these beaches.

Recommendations

It is recommended that an inter-related suite of four studies designed to: i) evaluate mechanisms by which key potential pollution sources impact upon beach water quality, ii) better resolve the scale of effect (spatial and temporal) of these potential sources, and, iii) evaluate suspected sources of bacterial pollutants on shoreline water quality be undertaken. The geographic extent of the proposed studies vary in scale from a strip of beach, to the immediate shoreline impacted by a small tributary, to an expanse of shoreline impacted by a large river system. The purpose of the nested geographic scales among studies is to provide insight on the landscape origin of the sources that impact beach water quality. Emphasis is placed on evaluating mechanisms of delivery of pollutants to shoreline waters and their subsequent delivery to beaches, because of the need to understand pollutant transport processes which determine their point of origin. The recommended studies are:

*Study 1. Investigation of beach-scale sources on the occurrence of *E. coli* at the shoreline*

Activities occurring at or adjacent to a beach may indeed be responsible for the adverse levels of microbial pollutants observed at a beach. Currently, little information is available to assess the relative impact of local (beach) sources of inputs compared to larger-scale inputs from the entire drainage area or inputs through transport from more distal shoreline locations. The potential water quality impacts of septic systems adjacent to beaches and the impact of drains and small watercourses delivering surface runoff from adjacent lands are additional information gaps that need to be filled. Wildlife, especially waterfowl, has been identified as an important source of *E. coli* at many beaches, but its impact along the shoreline of Lake Huron is unknown. Furthermore, the relative role of beach sand as a source of bacteria in the water has not been demonstrated yet many recent studies in Ontario, Michigan and California have found that populations of fecal bacteria, including *E. coli*, are present in the sands of beaches immediately adjacent to the lake (swash zone).

A study is recommended to fill these identified gaps, in particular, to examine the sources and loads of *E. coli* to the beach environment and the movement of these loads to the shoreline waters. The candidate study area should be a shoreline with residential developments, where beach monitoring indicates periodically elevated levels of *E. coli*, but, where there is a limited drainage area backing the shoreline. The over-arching objective of the proposed study should be to examine the significance of a variety of local small-scale sources of *E. coli* as a driver of

adverse beach water quality. The study should attempt to establish linkages between discrete local sources of *E. coli* and beach water quality impacts by verifying candidate sources and further evaluating the mechanisms by which the load from these sources reach the lakeshore. Candidate sources include: discharge from septic systems, surface drains, groundwater beach discharge, wildlife beach activity, beach recreational use and populations of *E. coli* in beach interstitial waters. Clearly delineating the overall significance of local sources relative to larger-scale loads on the incidence of adverse water quality will be challenging. However, the monitoring of physical processes in the nearshore of the lake and the application of microbial tracers and/or emerging microbial source tracking methodologies may provide insight on the degree to which local sources drive local water quality conditions.

Study 2. Investigation of the impact of tributary discharge from a small agriculturally-dominated watershed on E. coli occurrence at an adjacent beach

The beach water quality impact of the numerous small to moderate-size tributaries to the shores of Lake Huron is poorly understood. However, monitoring results indicate that the levels of *E. coli* in tributaries and drains are periodically elevated, suggesting their possible importance in influencing *E. coli*-based beach water quality. Given the abundance of watercourses reaching the coastline of southeastern Lake Huron, recreational beaches in Huron County are inevitably close to tributary mouths to varying degrees. Despite their potential for delivering significant loads of *E. coli* to the shoreline, much uncertainty remains with respect to the spatial or temporal extent of impact of small discharge volumes on shoreline water quality. The extent of impact on water quality will be highly variable over time and highly dependent on the volume and quality of the discharge, as well as depend on the mixing conditions at the tributary mouth at the time of discharge. The water quality impact will also depend on the watershed drained by these various small and moderate sized tributaries and any point sources of inputs to them. Land-use in these smaller tributary watersheds is dominated by agriculture with scattered communities of limited populations, and point sources of inputs, such as from sewage treatment plants, are essentially absent, found primarily within the drainage areas of the larger tributaries within Huron County.

A study is recommended in which the hydrological, land-use and water quality characteristics of a representative small tributary are monitored. This monitoring should be complemented by a concurrent assessment of the impact mechanism of the tributary discharge on an adjacent beach and an adjacent stretch of the shoreline. The primary objective of the study would be to

elucidate the relative role of smaller sized tributaries in driving adverse beach water quality at the shoreline of the lake. The study should examine how climatic and anthropogenic factors influence the extent and timing of any impact that the tributary discharge has on *E. coli* levels along the shoreline.

The study should document the spatial and temporal features of the impacts on water quality at the shoreline adjacent to the tributary mouth, using hydrodynamic modeling supported by in-lake water quality monitoring and instrument-based monitoring of physical conditions in the nearshore. By linking the extent of adverse water quality at the shoreline to tributary water quality features, discharge volume and lake mixing conditions, it should be possible to begin scaling the tributary impacts on the shoreline to the environmental factors that are driving water quality, discharge and mixing conditions over time. Distinguishing tributary discharge adverse water quality effects from those of sources along the immediate shoreline and/or from loads delivered to the shoreline by longshore transport will be necessary to conclusively isolate tributary discharge impacts. This will be a technically challenging task, however, chemical-based methods can be used to track tributary waters in the lake and this approach used in combination with microbial source tracking methodologies may provide a basis to distinguish tributary *E. coli* loads from other source loads.

Previous studies have shown a pronounced temporal variability in tributary water quality including levels of *E. coli*, suggesting a need to understand the basis of the fluctuations in water quality to ultimately recognize the potential effects of tributary discharge on the shoreline. The proposed study should attempt to assess how water quality in the lower tributary varies over time as a function of climatic and hydrologic factors known to affect water quality. In addition to the transport-related or dilution-related features correlated with weather and hydrology, it is expected that tributary levels of *E. coli* will vary as a function of the timing of the loads delivered to the landscape from various sources. To the extent possible, the study should therefore collect information on the temporal features of fecal pollutant loadings over the drainage area. The purpose of this aspect of the study should be to provide information on how tributary water quality may be expected to vary over time as a determinant of potential effects on the shoreline water quality.

Both the volume and the quality of the discharge from a tributary are important factors determining the potential impacts upon the shoreline. Volume of discharge will influence the

size of the mixing zone in the lake and will, along with pollutant concentrations (i.e water quality), drive the overall load of contaminants reaching the lakeshore. The proposed study will necessarily include both field and model-based examinations of the discharge characteristics of the tributary under study.

The findings of an intensive case study such as proposed here can provide a basis to frame predictions on the possible conditions at other areas of shoreline where watercourses discharge to the lake. Due to the wide variability in size of drainage areas and discharge volumes between watercourses, there is a need to understand how discharge characteristics vary among watercourses if the integrated impact of tributary inputs to the shoreline is to be assessed.

Study 3. Characterization of the Hydrology of Shoreline Tributaries of Huron County

There are a substantial number of watercourses which drain to the Lake Huron shores of Huron County, most of which are ungauged with the exception of the few larger tributaries (e.g. Maitland River, Bayfield River). However, the close proximity between the point of discharge of these smaller tributaries to the lake, and beach areas in general, suggests even minor watercourses may periodically affect beach water quality. The discharge patterns of the watercourses are likely highly variable over seasons and between years. Many of the smaller tributaries likely dry up or are reduced to minimal flows over the summer period except in response to wet weather events, while peak flows are expected during the spring melt and late in the spring and fall when precipitation events are more likely. In some cases, groundwater recharge may contribute to flows.

A study is recommended in which the hydrological features of a number of watercourses of representative sizes are evaluated to provide information on the discharge characteristics of watercourses to the shoreline of the lake. There is an uncertainty regarding the degree to which the delivery of pollutants by tributaries impacts upon water quality at recreational beaches over the bathing season when discharge rates may be low but possibly responsive to weather conditions. The volumes of water being delivered to the lake will greatly influence the size of the mixing zone. Volume is therefore a key factor in determining where and when the mixing zone extends to areas of the shoreline utilized for recreational activities, potentially having a direct impact on beach water quality.

A combination of hydrological modeling and field-based data collections should be used to provide information on expected patterns of discharge, and the variability in these patterns between watercourses across a gradient of sizes of drainage areas. The study should include an examination of the natural and anthropogenic factors contributing to short- and long-term temporal variability in discharge to the lake. The purpose of the study is to provide a basis to anticipate where, when and in what temporal sequence the watercourses to the lake may have an impact on water quality at the shores of the lake by virtue of the volume of water that they deliver.

Study 4. Examination of potential impacts of loading of particle-bound E. coli from a large river system to the lakeshore

Previous studies in Lake Huron have noted the role of particulate material in facilitating the transport and survival of *E. coli*. Bacteria may be found bound to the charged surfaces of particles composing bed sediments as well as to the suspended material in the water column. Within tributaries to the lake, *E. coli* associated with particulate material is transported downstream and is potentially discharged to the lake as particulate material. This association of *E. coli* with particulates is thought to prolong the survival of the bacteria in both the tributary and lake environments through protection from solar radiation, the supply of nutrients or possibly by providing a refuge from microbial grazers. Furthermore, the survival of particle-bound *E. coli*, once reaching the shores of the lake, is possibly enhanced as the particulate material settles out of the water column forming bed sediments. On the high energy shores of southeastern Lake Huron, deposits of surficial sediments are unstable and highly erosional. Consequently, bacteria deposited on the lake bottom at one time and in one place may subsequently be dispersed to other locations as the bed sediments continue to erode and deposit. This potentially expands the spatial and temporal scale of impact of the pollutant loads beyond that expected if the bacteria were suspended in the water column free of particulate material.

Despite the importance of bacteria-particle associations in *E. coli* dynamics around areas of the shoreline with appreciable tributary-based inputs of particulate material, there is little understanding of the spatial and temporal scales over which particle-bound loads of *E. coli* impact upon water quality on the shores of the lake. It is recommended that a study be conducted to investigate the transport and fate of loads of particle-associated *E. coli* delivered to the shoreline from a large tributary to determine the potential to impact upon water quality at the shoreline.

The potential for adverse effects on water quality during the recreational season from the loading of particulate-bound *E. coli* to the lakeshore will depend on the time of year of bacterial delivery, the bacterial survival time in surficial sediments, the resuspension of particulate material and its movement in the nearshore of the lake. A focus of this study should be on evaluating the potential for large tributary derived *E. coli* survival and transport along the shores of the lake. A key question to be addressed is whether adverse effects attributable to tributary loads of *E.coli* associated with particulate material are restricted to the mixing zone of the tributary (and area of greatest sediment deposition), or whether *E. coli* -laden particles can be dispersed to more distant areas of the shoreline with concentrations of *E. coli* elevated enough to impact water quality when surficial sediments are eroded.

The proposed study should also examine the time-frame over which loads of *E. coli* to the lakeshore may persist and contribute to *E. coli* measured at the shoreline during the summer period.

The proposed study will require detailed monitoring and physical modeling of the flux of particulate material from the mouth of a tributary over an extended area of shoreline in the lake. The spatial and temporal patterns of occurrence and persistence of *E. coli* in association with suspended particulates at the tributary mouth, in the nearshore of the lake and in the bed sediments of the lake will require evaluation. The difficulty level of the proposed study is high yet proportional to the potential value of information on role of particle-associated flux of *E.coli* in driving adverse levels of *E. coli* in shoreline waters. The present lack of understanding of particle-associated flux of *E. coli* is a significant impediment to understanding the linkages between pollution sources to adverse levels of *E.coli* at the shoreline of the lake.

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Appendix A : Selected Results of Water Quality Monitoring in the lower reaches of the Bayfield, Maitland and Suageen Rivers and adjacent nearshore areas of Lake Huron in 2003 by the Environmental Monitoring and Reporting Branch, Ontario Ministry of the Environment.264

1.0 Introduction

Microbial pollution of fecal origin of the shoreline waters of portions of southeastern Lake Huron has adversely affected their recreational use. Elevated levels of the bacteria *E.coli*, an indicator of fecal pollution, have resulted in the periodic posting of beaches as unsafe for swimming by local health units. In 2003, the permanent posting of five shoreline beaches of Huron County for the season as unsafe due to high levels of *E.coli*, as well as the efforts of a residents association in monitoring *E.coli* levels in tributaries draining to the shoreline along the Township of Ashfield-Colborne-Wawanosh, has focused renewed public attention on the existence of a water quality problem in southeastern Lake Huron, namely, fecal pollution of surface waters. Fecal pollution of recreational water can lead to health problems because of the presence of infectious microorganisms (World Health Organization 2003).

Evidence suggests that microbial pollution of fecal origin at the shoreline of southeastern Lake Huron is a long-standing problem with multiple sources of fecal pollutants from the lands draining to the impacted shoreline (Hockings and Dean 1989; Fuller and Foran 1989). The lands draining to southeastern Lake Huron support agricultural practices, including livestock operations and cropland to which nutrient biosolids (manure) are applied. Other domesticated and wild animals also inhabit the lands and waters adjacent to the shoreline. While urban development is limited, several small to moderate sized communities use communal sewage works that discharge to the lake shore or to rivers that drain to the lake. In addition, the shoreline is peppered with seasonal and permanent dwellings which rely on septic systems for sewage disposal.

The *Science Committee to Investigate Sources of Bacterial Pollution of the Lake Huron Shoreline of Huron County* was initiated by the Ontario Ministry of the Environment in the spring of 2004 to undertake a science-based examination of the sources of *E. coli* as an indicator of microbial pollution of fecal origin to the Lake Huron shoreline and investigate the extent of influence on the shoreline from various sources. The committee is working towards an improved understanding of the relative influence of the key sources of fecal pollution to the shoreline and of how these pollutants are delivered to the lakeshore through 1) a review of existing information; 2) the identification of knowledge gaps; and 3) potentially undertaking new work to address key unknowns. It is the view of the Committee that a better understanding of the relative influence of sources, combined with better definition of the mechanisms by which

these sources are reaching the lake shore will enhance the selection of actions to be taken to address microbial fecal pollution of the shoreline.

The Committee has proposed a three phased approach to meeting its two primary objectives of 1) contributing to the understanding of the mechanisms of microbial fecal pollution and relative sources of microbial fecal pollution to Lake Huron beaches, and, 2) based on the results of the first objective, providing scientifically sound advice for actions that might reduce the degree to which Lake Huron beaches are impacted by microbial fecal pollution. In the first phase, existing information and past studies were collected and reviewed to identify the current state of knowledge on the sources of fecal pollution to the shoreline, the ways in which these pollutants are delivered to the lake, and the degree to which the relative impacts on shoreline conditions from various sources of pollution is understood. As part of the second phase, a study plan is being developed, and may be undertaken, to work towards addressing some of the key information and knowledge gaps identified in phase 1. Finally at the conclusion of phase 2, the committee will update the state of understanding of the sources of fecal pollution to the shoreline and, to the extent possible, provide recommendations on actions which may contribute to a better management of fecal pollution of the Lake Huron shoreline.

The microbial fecal pollution of surface waters used for recreational purposes is a seemingly ubiquitous and complex problem in developed areas. Consequently, decades of research and monitoring have been dedicated to this problem. Many jurisdictions responsible for the Great Lakes coastline, and elsewhere, have been faced with identifying sources of fecal pollutants which impact shoreline waters. In response, these various governing bodies have undertaken, or are presently conducting, studies directed at assessing pollutant sources. It is well documented that many potential sources of fecal inputs to surface waters exist in a typical rural or urban landscape (e.g. World Health Organization 1999). The vectors by which pollution loads can be delivered to the shoreline are diverse and often difficult to assess. Furthermore, there is recognition that the die-off rates of microorganisms associated with fecal pollutants in the environment are highly variable and are affected by a range of physical, chemical and biological factors. The fluctuations of fecal indicator bacteria in the environment are the results of the interplay between loading and die-off rates. Linking potential sources of fecal pollution directly to beach impacts as evidenced by levels of fecal indicator bacteria and further assessing the relative significance of sources on outcomes at beaches remains a difficult and complex task.

The geographic extent of this study was limited to the Lake Huron shores of Huron County (see Figure 1). Within this larger study area, a sub-section of the shoreline was selected for a more detailed analysis of shoreline water quality and environmental conditions. The approximate 45 km of shoreline from Amberley Beach (Point Clark area) to Bayfield Beach includes a wide range of physical conditions and land-use features and is thought to be representative of much of the southeastern shores of Lake Huron. The shoreline also encompasses the five Lake Huron beaches permanently posted as unsafe for water recreation in 2003 by the Huron County Health Unit. A primary objective was to undertake a detailed analysis of a limited geographic area where efforts could be intensified and presumably a more robust synthesis of information could be achieved. A secondary objective was to develop a frame of reference for examining fecal pollution problems at beaches which would have a general applicability to the southeastern shores of Lake Huron as a whole. Information for other areas of Huron County and the southeastern shores of Lake Huron is used selectively in the report. Microbial pollution of the area selected for detailed study is not thought to be unique to or different from other areas of southeastern Lake Huron within and beyond Huron County. Rather, the findings for the study area are expected to be applicable to the broader area of southeastern Lake Huron.

This interim report presents the progress of the committee on Phase 1. The report presents a summary of existing information and attempts to provide a synopsis of the state of understanding of microbial pollution of fecal origin over the study area, as inferred primarily from results for the fecal indicator bacteria *E. coli*. The approach taken was to examine information from three perspectives: i) the identification and description of the potential sources of fecal pollution to the shoreline; ii) the review of the evidence for shoreline impacts by the various potential sources; and, iii) the investigation of the potential mechanisms of delivery of microbial fecal pollutants to the shoreline. In addition to publications, study reports and other documentation, selected primary data for monitoring and research conducted in recent years was acquired and examined. The composite information and analysis was then used to identify conceptual and informational barriers that are considered important to the assessment of the sources of fecal pollution to the shoreline and to the understanding of the extent of the various potential sources on shoreline water quality as inferred from *E.coli*. The final portion of the report outlines recommendations for proposed studies in Phase 2 that would contribute to the understanding of the shoreline pollution problem and possibly contribute to a reduction in the level of occurrence of microbial fecal pollutants at the recreational beaches of Huron County by providing a basis to more effectively target and prioritize remedial actions.

The majority of the information presented in this report infers microbial fecal pollution based on the occurrence of the indicator species *E. coli*. This bacteria species is widely used, and well recognized, as an indicator of fecal pollution (see National Research Council 2004 for discussion on use as an indicator). Pathogenic strains of *E. coli* do exist but represent a small proportion of the total *E. coli* population and are not distinguished during the enumeration of *E. coli* as part of routine water monitoring. The intent of routine microbiological analysis of water is to detect the presence of fecal contamination, and thus, the potential presence of pathogens. There is a Provincial Water Quality Objective (PWQO) (Recreational Objective) for *E. coli* and *E. coli* is the indicator species recommended by the Ontario Ministry of Health and Long Term Care in the monitoring of recreational beaches by local health units in the Province. The basis of the Recreational PWQO is the protection of recreational water uses (swimming, bathing and other recreational uses requiring immersion of the user) such that immersion should not cause disease (Ontario Ministry of the Environment and Energy 1994a). The PWQO for the indicator bacterium *E. coli*, which has been in place since 1993, in the Province of Ontario is 100 *E. coli*/100mL based on a geometric mean of at least five samples. The Canadian guideline for recreational waters is 200 *E. coli* /100 mL, also based on a geometric mean of five samples taken within a 30 day period (Health and Welfare Canada 1992).

Despite the wide-spread usage of *E. coli* as a microbial indicator of fecal pollution, there are concerns with short comings to its use as an indicator. A concern, particularly relevant to use as an indicator of fecal pollution is the potential that the indicator persists longer in the environment than previously thought, particularly in soil and sediment, and consequently detection of the indicator may not necessarily reflect recent fecal contamination. There are broader issues with the use of *E. coli* as an indicator of microbial water quality in recreational waters but these are beyond the scope of this report (see National Research Council 2004 for discussion). The decision to focus this study on *E. coli*, is based on the reasoning that this is the chosen indicator of microbial fecal pollution in recreational water quality guidelines recognized and widely applied in Ontario and Canada for water pollution monitoring and study and that much of the existing information addressing microbial fecal pollution for the shores of Lake Huron is for *E. coli*. An assumption made in this report is that the detection of *E. coli* in the environment indicates adverse microbial water quality due to the possible presence of pathogenic microorganisms derived from fecal material. This argument was put forward in the late 1960's based on epidemiological studies conducted by the US public health service in the late 1940's and 1950's showing increased incidence of gastroenteritis with elevated fecal coliform levels (Csuros and Csuros 1999).

2.0 Characteristics of Microbial Pollution of Fecal Origin (as Inferred From the Indicator *E. coli*) on the Lake Huron Shores of Huron County

A diverse range of information, both historical and current, exists concerning microbial pollution of the southeastern shoreline of Lake Huron. The information ranges from results of environmental monitoring and microbial research in the aquatic environment to studies which target the generation (and subsequent transport in the environment) of microbial pollutants from specific activities or sources. What are thought to be key sources of information, either specific to Lake Huron, or directly relevant to Lake Huron are summarized in this section. The findings of the original studies are presented, and for the more recent work where there was access to primary data, selected aspects of the original data are presented. In some cases the data has been combined with other sources of environmental information to further interpret the data. Documented concerns related to fecal pollution of the shores of Lake Huron go back at least 40 years and have been the subject of work by many organizations and groups of people. An attempt was made to be inclusive in the search for existing information, however, given the scope and diversity of the issue it is recognized that this review of information will be incomplete.

2.1 PLUARG Pollution Studies on the Great Lakes

In 1972, the Canada-US Agreement on Great Lakes Water Quality was signed. Through that agreement, the International Joint Commission (IJC) established the International Reference Group on Great Lakes Pollution from Land Use Activities (PLUARG) to conduct a study of the pollution of the boundary waters of the Great Lakes System from a multitude of land-use activities. Through their work, PLUARG confirmed and studied two major pollution problems in the Great Lakes basin: eutrophication and increasing contamination by toxic substances (PLUARG 1978). The focus of the PLUARG studies was to better understand non-point sources of pollution associated with specific land-use activities including agriculture, urbanization, forestry, transportation and waste disposal. A wide range of studies, including monitoring and detailed sub-watershed studies, were undertaken to define the relationships between land-use activities and water quality and further our understanding of the environmental features which defined the spatial-temporal characteristics of pollution impacts.

Bacteria of public health concern were identified as one of the Great Lakes water quality problems by the PLURAG study (PLUARG 1978). The problem was characterized as primarily a point source problem affecting the nearshore, or localized areas, of the Great Lakes. Land runoff was identified as a contributing factor to the problem, however, a more important problem identified was the input from combined sewer overflows (on the scale of the Great Lakes). The only recommendation concerning microbial pollution in the final PLUARG report was that epidemiological evidence be examined to establish criteria for body contact recreational use of waters receiving runoff from urban and agricultural sources.

A Lake Huron watershed was selected for analysis by the PLUARG program and studied intensively as one of several pilot investigations of watersheds draining to the Canadian shores of the Great Lakes (Onn 1980). The relationship between water quality and agricultural and urban land-uses were investigated in the watershed of the Saugeen River (O'Neil 1979; Neilsen et al. 1978), however, an evaluation of the associated microbial pollution was not included in this study.

2.2 Past Studies of Microbial Pollution of SE Shores of Lake Huron by the Ontario Ministry of the Environment

Historically, the Ontario Ministry of the Environment (MOE) has been active in the environmental monitoring and investigation of the nearshore waters of Lake Huron. Early microbial studies include investigations of recreational water quality concerns in Southampton in 1964 by the Ontario Water Resources Commission (OWRC) (Letman 1964), and an evaluation of nearshore bacteria levels along the Canadian side of Lake Huron and Georgian Bay between 1973-1976 (Young et al. 1977). Beach postings at Lake Huron beaches in 1983 and the resulting public concerns spurred significant ministerial efforts in 1984 and 1985, focusing on the factors affecting microbial water quality of Lake Huron beaches (Palmateer and Huber 1984; 1985). These and related studies contributed to the development of the province-wide Clean Up Rural Beaches (CURB) program between 1989-1996.

In 1984, Garry Palmateer of the MOE led a series of extensive investigations to quantify the levels of fecal indicator bacteria at beaches along the Lake Huron shoreline and assess whether correlations could be made with meteorological and physical information (Palmateer and Huber

1984). In 1985, a follow-up, intensive study was carried out to determine the precise location and significance of fecal pollution sources, particularly sources of agricultural origin (Palmateer and Huber 1985). One important finding was the significant role of bacterial survival in beach sediment. A more detailed analysis of fecal pollution sources, and survival of fecal indicator bacteria in sediments was therefore initiated and conducted in the Desjardine Drain in Grand Bend (Palmateer et al. 1993). The degree of colonization of suspended particulates by total viable bacteria and *Salmonella* was studied, in addition to the transport of sediment bound *E. coli* through the Desjardine agricultural drain (Palmateer et al. 1993, Hayman et al. 1994). Most studies were conducted along the southeast shore of Lake Huron, from Goderich, through Grand Bend and Ipperwash Provincial Park and more extensive studies and experiments were performed at and around Grand Bend. The following text evaluates the methodology and summarizes major findings of these studies.

2.2.1 Microbial Water Quality Evaluation

Characterization of microbiological water quality at Lake Huron was studied during 1984 and 1985 (Palmateer and Huber 1984; 1985). The initial objective of the 1984 study was to establish the level of microbial fecal contamination (as inferred from indicator bacteria) of beaches along the Lake Huron shoreline and major rivers potentially affecting these beaches. In 1985, the study area was narrowed to focus on the beach and region around Grand Bend. Many of the initial studies were repeated, with a greater focus on defining the locations and impacts of agricultural and urban sources. Daily monitoring for fecal coliforms (FC), fecal streptococci (FS) and the pathogen, *Pseudomonas aeruginosa* was conducted. Additional measurements and observations were recorded, including turbidity, lake roughness and number of swimmers at each beach in an attempt to correlate them with increased levels of fecal indicator bacteria. Experiments assessing the survival of *E. coli* were expanded based on results of the 1984 study. Finally, microbial source tracking methods were employed to determine the predominant sources of high beach bacterial numbers.

These initial studies measured FC and FS bacteria as indicators of fecal pollution. Since then, *E. coli* has been recognized as a better indicator of fecal contamination and it has been used for this purpose in more recent studies, as well as in current routine monitoring practices. In 1996, many of these studies were repeated as part of a rapid *E. coli* test evaluation. At that time, *E.*

coli was found to comprise 92 to 97% of FC concentration (Glaskin-Clay et al. 1996). The actual methods of bacterial recovery and enumeration are not discussed here, since the same methods were used within each individual study report, and many of the results were confirmed later using different methods.

Effect of Rainfall:

The 1984, 1985 and 1996 studies failed to establish a systematic relationship between rainfall events and levels of FC, FS and *E. coli*. At Grand Bend, FC counts in the Old Ausable River frequently correlated well with rainfall events. Some peak FC counts at the beach also correlated with rainfall. Correlations between major rainfall events and FCs were also noted at Ipperwash. Additionally the FC counts in Duffus Creek, the only waterway impacting Ipperwash beach, were extremely high following rainfall events. Goderich Beach had greater bacterial fluctuations than the other three beaches and it was difficult to identify a relationship between rainfall events and bacterial levels at this beach. Rivulets passing over Sauble Beach had elevated levels of FC and FS during and after rainfall events, however, bacterial counts were found to be elevated during dry periods in three agricultural drains discharging directly into the Sauble River (Palmateer and Huber 1984).

In 1985, there were fewer rainfall events significant enough to generate runoff or increased stream flow. However, it was noted that peak bacterial levels occurred during rainfall events when field soils were wet from previous rainfall, whereas bacterial levels did not increase significantly when rainfall events occurred on dry soils (Palmateer and Huber 1985). Finally, the 1996 study failed to find a correlation between levels of *E. coli* and rainfall events at a number of beaches along the southeast shore of Lake Huron (Glaskin-Clay et al. 1996).

Lake Roughness and Wave Height:

In 1984, indicator bacteria levels were found to be higher when the lake was rough and waters were turbid (Palmateer and Huber 1984). Similar results were observed in 1985 when increased bacterial levels at Grand Bend beach correlated well with increased wave height, particularly during periods when rainfall was minimal (Palmateer and Huber 1985). In the 1996 study, *E. coli* levels increased with both wave height and on-shore winds (Glaskin-Clay et al.

1996). In general, elevated southerly or westerly winds tended to be associated with increased bacterial counts along the southeastern shore of Lake Huron (Palmateer and Huber 1984; 1985). This relationship is likely a consequence of greater wave action and possibly sediment resuspension facilitated by these high lake-to-shore winds.

Tributaries (Rivers and Creeks):

On several dates in 1984, the Ausable River and 16 small creeks and agricultural drains were examined more intensely to track the potential sources of fecal contamination to various Lake Huron beaches. In 1984, over half the samples from the mouth of the Ausable River were below the PWQO of 100 FC/100 mL, while only 6% of samples taken upstream met those objectives. This suggested that sedimentation, die-off and dilution occurs downstream (Palmateer and Huber 1984). Similarly, in 1985, 68% of the samples from a site where many of the tributaries converge with the Ausable River exceeded the PWQO and 21% exceeded 1000 FC/100 mL (Palmateer and Huber 1985). Average FC counts were approximately half downstream compared to upstream, confirming previous results and supporting the hypothesis of downstream sedimentation, die-off and dilution. Low urbanization to agriculturalization in the watershed of southeastern Lake Huron suggests that bacterial contamination is likely from agricultural sources. For example, in 1985, cattle watering in four tributaries of the Ausable River was followed by extremely high bacterial levels (>10,000 FC/100 mL) in these creeks which subsequently decreased when the cattle were moved away (Palmateer and Huber 1985). It should be noted, however, that many peaks in bacterial levels were also correlated with rainfall events (see Effect of Rainfall section above).

All 16 agricultural drains and creeks sampled contained elevated levels of bacteria, particularly during rainfall events. In 1984, Walker Drain at Grand Bend and Duffus Creek at Ipperwash were badly contaminated and examined further. A septic tank was discovered to be connected to a farm tile drainage system which discharged directly into Walker Drain. At Duffus Creek, extremely high levels of bacteria were measured directly downstream from the storm water drainage from an army cadet camp. It was suspected that raw sewage from one of the buildings was contaminating the storm sewer which discharged into Duffus Creek. This was confirmed when the bacterial levels in Duffus Creek were significantly reduced once the storm sewer was capped and the water was pumped back into the sanitary line (Palmateer and Huber 1984).

In 1985, four of the 16 drains sampled in 1984 were chosen for a detailed study, using Duffus Creek at Upperwash as a reference (seasonal average of 60 FC/100 mL) (Palmateer and Huber 1984; 1985). Three drains at Grand Bend, drains 2, 14 and 15, had seasonal averages of 1,467, 2,044 and 2,480 FC/ 100 mL, respectively. Hourly sampling at different locations in drain 2 revealed an increase in bacterial levels between Highway 21 and the lake, suggesting localized inputs. Six drain pipes, of unknown origin, were observed entering the drain west of Highway 21, however, their combined effect resulted in a drainage entering the lake with very elevated bacterial counts (71,000 FC/100 mL) (Palmateer and Huber 1985).

At drain 14, FC counts increased significantly from one concession east of Highway 21 (1,179 FC/100 mL) to Highway 21 (30,058 FC/100 mL). The span of this drain passed exclusively through farmland and its drainage accumulated directly onto the beach rather than flowing into the lake. Cattle were routinely observed watering in drain 15 and bacterial levels always peaked during cattle access periods (Palmateer and Huber 1985).

Samples taken along a grid pattern where these drains entered the lake contained elevated bacterial levels. Grid samples in 30 cm of water where drain 2 entered the lake contained 2,000 to 23,000 FC/100 mL. Further out at 1.2 m depth, the counts were between 160 and 4,000 FC/100 mL. At drain 14, grid sampling at the lake showed counts between 2,800 and 10,200 FC/100 mL. These results clearly demonstrated the input of fecal indicator bacteria into the lake water from shoreline drains (Palmateer and Huber 1985).

Palmateer and Huber (1984) also examined the dispersion of the discharge plumes from Ausable River and the Maitland River as a mechanism for moving fecal pollutants in river water to the shoreline of the lake. The elevated conductivity of tributary water relative to lake water was used as a tracer of the dispersion of the river plumes. Spatial patterns in the Ausable River and the Maitland River plumes as a function of wind direction were noted, however, no information was presented by Palmateer and Huber (1984) on the potential significance of large river discharge on shoreline water quality. Palmateer and Huber (1985) noted that the Ausable River plume moved onto the Grand Bend beach when the wind direction was from the south or west and suggested that loading from the river contributed to levels of FC at the beach but the basis of this claim is unclear from the report.

Sunlight and Water Depth:

Intensive hourly sampling of solar insolation and corresponding indicator bacteria levels at Grand Bend showed substantial variation with no apparent trend in both 1984 and 1985 (Palmateer and Huber 1984; 1985). However, on average, bacterial levels decreased when daily hours of direct sunlight were greater.

Water column bacterial levels decreased greatly with water depth at Ipperwash, and to a lesser extent at Grand Bend (Palmateer and Huber 1984). Differences in the slope of the lake bed adjacent to the shore between different beaches and the resulting difference in proximity to shore at a given water depth was thought to influence the variable response with depth.

Swimmers, Seagulls and Pets:

There was little correlation between FC counts and the number of swimmers or number of seagulls present at Grand Bend beach, though, levels increased slightly in the afternoon when swimmer density was higher (Palmateer and Huber 1984). Furthermore, levels of *E. coli* did not correlate with water temperature, number of swimmers, pets or seagulls in the 1996 study at a variety of beaches along the southeast shore of Lake Huron (Glaskin-Clay et al. 1996).

Sanitary Surveys:

Storm sewers and drainage outlets were sampled at Grand Bend and Goderich in the 1984 study only. Additionally, discharge from the sewage treatment plant at Goderich was analyzed. Most storm drainage outlets at Grand Bend had moderate to low bacterial levels, with the highest being the discharge to Walker Drain. In Goderich, some storm sewers contained high bacterial levels following rainfall. The sewage treatment effluent was found to be adequately disinfected, though adequate disinfection was not defined in this document. On average, there were high fluctuations in bacterial counts in Goderich and it was very difficult to determine the likely factors impacting bacterial levels in this area.

Impact of Boats and Marinas:

Three boats out of a total of 160 inspected were out of compliance with waste holding regulations. Pleasure crafts were not thought to be contributing significantly to fecal pollution. Results from samples taken upstream and downstream from marinas were inconclusive (Palmateer and Huber 1984).

Bacteria in Sediment:

The use of *E. coli* as an indicator of fecal pollution in water is based on the assumption that *E. coli* does not survive long in aquatic environments. While this holds true in drinking water, where the levels of particles and nutrients are low and disinfection is common, there is increasing evidence that *E. coli* may survive longer than previously thought in surface waters such as lakes and rivers.

In the 1984 study, levels of fecal indicator bacteria were consistently higher in beach sediment than in the water above the sediment sampling sites, while intermediate bacterial levels were measured in the floc-like material which lies at the sediment-water interface (Palmateer and Huber 1984). Gradients of bacterial levels were not consistent across sediment and water samples. While bacterial levels were greatest at Goderich, both in the sediment and the overlying water, Ipperwash sediment samples had higher bacterial levels than at Grand Bend sediment, while Ipperwash water had lower levels of bacteria than Grand Bend water. It was hypothesized that the fine sand of Ipperwash beach supported higher concentrations of bacteria compared to the coarser sand of Grand Bend. Furthermore, lower turbidity values suggested that less sediment resuspension occurs at Ipperwash than at Grand Bend.

Finally, sediments in the drains and tributaries of the Ausable River were found to contain much higher levels of fecal indicator bacteria than in the water (Palmateer and Huber 1985). Particle analysis showed that 50 percent of the particles in the sediment of the Ausable River were less than 100 μm in diameter. This material is easily resuspended during high stream flow conditions, which may facilitate the downstream transport and ultimate input of microbial pollutants to the lake.

Survival of *E. coli* in Water and Sediment:

In an initial six day study, *E. coli* survival was measured in diffusion chambers submerged at various sites in Lake Huron and anchored to the bottom. During this investigation, five days were required to reduce initial *E. coli* levels of 10^9 cfu/100 mL to the PWQO of 100 cfu/100 mL. Furthermore, survival of *E. coli* depended on the initial concentration of bacteria added to the diffusion chambers (Palmateer and Huber 1984).

In 1985, this study was repeated and included diffusion chambers loaded with 10^7 cfu *E. coli*/mL in water and sediment from Lake Huron and water in the Ausable River. In the lake water, the half-life of *E. coli* was 36.5 hours and complete die-off occurred after 6 to 7 days. However, in the nutrient-rich Ausable River water, *E. coli* survived in excess of 3 weeks. After 1 month, *E. coli* was still present in the Lake Huron sediment in the range of 10^6 cfu/100 g. Following these experiments, Palmateer and Huber (1985) suggested that *E. coli* may survive at least 2 months in sediment.

2.2.2 Microbial Source Tracking (MST)

Microbial source tracking (MST) studies were rarely undertaken before the 1990s and reliable, time tested methods were not available. Nevertheless, attempts to differentiate between human and animal fecal inputs were made. Several methods were used. None of these methods are currently in use in the described format today to differentiate fecal contamination sources, however, it should be noted that at the time of this study, these would have been considered state of the art technologies. The methods, results and relative shortcoming of each technique are discussed below. Finally, these studies are burdened by an inherent bias, most of the MST studies reported here were performed in, and around, agricultural drains where the expected input from animal fecal sources was high. None of these studies took place in Goderich near the sewage treatment plant and the presence of cottages and septic tanks in the regions studied was not reported.

Fecal Coliforms to Fecal Streptococci Ratio:

Due to the differing relative abundances of fecal coliforms and fecal streptococci between human and animal sources of fecal material the FC:FS ratio has been used as a source indicator in past studies but is no longer considered a reliable approach. A FC:FS greater than 4 was considered indicative of human fecal contamination, whereas a ratio less than 0.7 suggested animal sources of fecal contamination. Problems with this method include differential survival rates in water among different fecal streptococcus species and the significant effect of wastewater disinfection on the ratio of these indicators. Furthermore, the ratio is also affected by different methods of recovery. These are summarized in "Standard Methods for the Examination of Water and Wastewater" (American Public Health Association 1998).

While the authors acknowledged the problems with this methodology, failing better methods, ratios were nevertheless calculated. In some agricultural drains, the FC:FS suggested contamination by animal wastes while the FC:FS of one drain surrounded by cottages and summer homes suggested contamination by human waste.

Antibiotic Resistance Analysis:

Resistance to 10 commonly used antibiotics was determined for both *E. coli* and *P. aeruginosa* isolated from many beach samples. Antibiotic resistance frequently develops in bacteria exposed to specific antibiotics over a period of time. Therefore, bacteria resistant to antibiotics commonly used by veterinarians and farmers were thought to originate from animal sources, whereas bacteria resistant to antibiotics used by physicians to treat human infections were considered to originate from human waste. Using these criteria, the majority of isolates recovered from the beach water in this study were of agricultural origin (Palmateer and Huber 1984). This analysis was repeated using some additional antibiotics in 1985 at Grand Bend beach and the Ausable River, tributaries and drains discharging at the beach. The frequency of resistance to antibiotics used in animal husbandry was much greater than for those used by physicians (Palmateer and Huber 1985).

While this method of MST was ingenious given the state of the science at the time, this type of antibiotic resistance analysis is quite different than antibiotic resistance profiling currently used in MST studies. Mechanisms of antibiotic resistance in bacteria are complex. Some

mechanisms confer multiple drug resistance. Some resistances are encoded by genes present on plasmids that can be transferred from one bacterium to another, a process that has been demonstrated in natural aquatic environments. Observed resistance to antibiotics may reflect growth conditions in the laboratory, rather than actual characteristics of the strain in nature. Finally, while the occurrence of resistance to veterinary associated antibiotics may suggest agricultural sources of fecal contamination, it does not eliminate the possibility of human sources of fecal contamination. As such, these results can be considered as evidence supporting, rather than proof of agricultural sources of fecal contamination in Lake Huron beach waters.

Current methods of antibiotic resistance analysis involve generating databases of antibiotic resistance profiles of various isolates. Rather than counting the number of isolates resistant to commonly used veterinary drugs, antibiotic resistant profiles are generated on an isolate by isolate basis. Isolates obtained from human and different animal sources (e.g. pig, cow, chicken) are tested against a wide variety of antibiotics. Profiles from each strain are generated and recorded in a database or library. Isolates obtained from water samples are also profiled and compared to the library of source isolates. This allows the same treatment of source and beach isolates, rather than assuming the characteristics of environmental isolates will reflect conditions present within an animal or human host. Additionally, patterns of multiple drug resistance are identified and characterized as part of the profile. These results are also semi-quantitative in the sense that the proportion of isolates matching different source libraries can be calculated.

Serotyping:

Serotyping refers to an identification based on the use of antibodies to specific antigens on the bacterial surfaces. For example, *E. coli* is serologically identified by the O and H antigens. Hence, *E. coli* O157:H7 is a serotype of *E. coli*. Serotyping is a broad mechanism for strain identification and characterization. Among enteric bacteria such as *E. coli* and *Salmonella*, many serotypes can be isolated from a variety of sources. For example, *E. coli* O157:H7, while predominately found in cattle, has also been isolated from sheep and pigs (Chapman et al. 1997).

In the MOE studies, isolates of *P. aeruginosa* were characterized by serotype. *P. aeruginosa* is not considered an indicator of fecal pollution and is found in a variety of other sources. As such, no attempt was made to relate serotypes found at the beach to sources of fecal contamination in this study. Instead, this data was used to find similarities between isolates found in various creeks and the beach to which they drain. In some instances at and immediately north of Grand Bend, the same serotype was recovered from an agricultural drain or small tributary as well as at the beach (Palmateer and Huber 1984). Similar results were observed in the 1985 studies at Grand Bend. While serological identification of isolates at Ipperwash and Goderich were less informative (Palmateer and Huber 1984), on average, these results support the hypothesis that bacteria from agricultural creeks and drains can be transported to the beach.

2.2.3 Impact of Sediment on Survival and Transport of *E.coli*

Survival of fecal indicator bacteria in sediments, and the subsequent resuspension of these sediments appeared to play a significant role in beach bacterial contamination in 1984 and 1985. Consequently, additional studies were done to evaluate the survival of bacteria in sediments, as well as the potential for bacteria adsorbed to sediments to be transported through agricultural drains to the beach. These studies were performed in the late 1980s and early 1990s (Palmateer et al. 1989; Palmateer et al. 1994; Hayman et al. 1994). One of these studies was published in a Ministry of the Environment report (Hayman et al. 1994) as well as a chapter in a text book (Palmateer et al. 1994).

The populations of *E. coli* and FS in sediment within the Desjardine Drain were examined in a joint study done in 1987 by the MOE and the Ausable Bayfield Conservation Authority (Palmateer et al. 1989). Over a 1 year period, concentrations of both *E. coli* and FC ranged between 10^5 and 10^7 bacteria/100 g sediment. Bacterial levels declined between February and May and then increased throughout the summer and early fall. It was hypothesized that bottom scouring during spring peaks in stream flow may have resulted in reduced bacterial sediment levels, while summer and fall reduced stream flow favored the settlement and accumulation of bacteria laden particles on the bottom. Caution should be exercised when drawing conclusions based on sediment bacteria levels since observed levels likely reflect variability in loading of bacteria to the sediment, their persistence in the bed sediment, as well as the flux of the bed sediment.

Beginning in 1991, a 2-year follow-up study examining bacterial survival in sediments at three agricultural drains was performed by the MOE in conjunction with the Upper Thames River Conservation Authority (Palmateer et al. 1993; Hayman et al. 1994). In addition to the Desjardine Drain at Grand Bend, the Arthur Vanatter Drain near Kintore in Oxford County and the Central School Drain near Shakespeare in Perth County were also added to the study. The water quality in all locations exceeded the PWQO of 100 *E. coli* / 100 mL. Water from all three drains had consistently lower levels of *E. coli* and FS in the spring and higher levels during the low flow period of the summer and fall.

Adsorption of Viable Bacteria to Sediment Particles:

The effect of sediment size on the adsorption of bacteria was investigated. Particles from two locations in the Desjardine Drain, the Ausable River and the beach water at Lake Huron were analyzed using particle counting and sizing instrumentation, and separated by size-selective filtration. Particle concentration decreased with increasing particle size. Total viable bacteria adsorbed to particles and in the interstitial waters were determined using acridine orange staining and epi-fluorescence microscopy. Particles ranging from 11 to 84 μm diameter were highly colonized with viable bacteria. On average, 50.5% of the attached bacteria at all sites were viable compared to 24.3% viability among unabsorbed bacteria. At the beach, over 70% of attached bacteria were viable. Acridine orange stains all viable bacteria, and does not differentiate between different species. The proportion of viable *E. coli* and FS could not therefore be determined (Palmateer et al. 1989).

The degree of colonization on sediment particles by total viable bacteria and *Salmonella* were examined in subsequent studies (Palmateer et al. 1993; Hayman et al. 1994). This time, viable bacteria were stained with 4',6'-diamidino-2-phenylindole (DAPI). *Salmonella* was detected by immunofluorescence microscopy after staining with a fluorescein isothiocyanate labeled antibody (FITC-FA). The same slide preparation can be stained with DAPI and FITC-FA simultaneously and be visualized microscopically using different light filters. Similar methods are currently in use for the detection of a variety of microorganisms including bacteria and parasites. The number of viable, adsorbed bacteria was considerably higher than free floating viable bacteria in three agricultural drains studied. In most cases, the quantity of adsorbed, viable bacteria increased or remained unchanged throughout the spring, while free floating

bacterial levels increased in the fall. *Salmonella* was observed most frequently adsorbed to particles between 30 and 70 μm in diameter at the Arthur Vanatter and the Desjardine Drains. Free floating *Salmonella* were observed in all three drains, although less frequently in the Arthur Vanatter Drain, indicating that pathogenic bacteria were present and viable in these agricultural drains.

These studies comparing the three agricultural drains took into account sediment characteristics in each drain. The Desjardine Drain had less sand and more silt than the other two drains. Additionally, it had a significantly higher cation exchange capacity, which may allow greater bacterial adsorption than the other two drains. This drain had the highest level of total viable bacteria and *Salmonella* adsorbed to suspended particles (Palmateer et al. 1993; Hayman et al. 1994). The Desjardine Drain is the only one of the three that discharges to Lake Huron.

Potential sources of these bacteria were discussed but not scientifically examined. Buffer strips and cattle fencing had been installed at the Arthur Vanatter Drain as remedial measures, but a significant wild deer population may have contributed some fecal pollution there. The Central School Drain, which had the poorest water quality of the three drains, had a tile drain discharge known to contain milkhouse waste water in addition to subsurface manure runoff near the main sampling site. Additionally, cattle regularly watered in the drain upstream from the main sampling site. Sources of fecal bacteria in the Desjardine Drain may have been poorly performing septic tanks and manure from cattle watering in the stream (Palmateer et al. 1993; Hayman et al. 1994).

Survival in Sediment – Effects of Sunlight:

The relative survival rates of *E. coli* in sediments and overlying water column within the Desjardine drain were examined using diffusion chambers spiked with a marked strain of *E. coli*. *E. coli* NAR, a nalidixic acid resistant strain, was used in these experiments so that it could be differentiated from native populations. Diffusion chambers containing water were submerged in water in direct sunlight and under a bridge away from direct sunlight. Diffusion chambers containing sediment were placed at the bottom of the Desjardine Drain. *E. coli* NAR survived 11 days in chambers (water only) exposed to direct sunlight and 12 days in chambers sheltered from sunlight. This difference did not appear to be significant. In the chambers with sediment, the *E. coli* NAR population appeared to stabilize at approximately 1500 bacteria/100 mL after 10

days and was still present at this level after 32 days when the experiment was terminated. Nutrient availability on sediments may be responsible for greater survival rates (Palmateer et al. 1989).

Transport of *E.coli* Through Agricultural Drains:

Given the observed survival rates of fecal indicator bacteria in both water and sediments, appreciable transport of fecal bacteria within drains seems possible. Experiments were conducted to specifically evaluate the transport of *E. coli* within the Desjardine Drain. *E. coli* NAR was added to the top of the drain at a concentration of 1.1×10^6 cells/100 mL. A fluorescein tracer dye was also added to help determine sampling frequency and sites. Quantitative and qualitative sampling was done at 16 sites, roughly 1 km apart along the drain. This experiment was conducted three times: once in November during a period of high stream flow and twice in April when the stream flow was lower. In November, *E. coli* NAR was detected at all 16 locations, though the numbers decreased over the course of 4 days. In April, *E. coli* could be enumerated in the near stations (2 and 3 km from the point of addition) and then sporadically by qualitative (presence-absence) methods up to 9 km away 22 hours after addition of bacteria.

In addition to reduced stream flow, the hours of bright sunlight during April were 3.5 and 2.3 times greater than in November. Upon further examination of the effects of bright sunlight, four hours of sunlight was experimentally shown to result in a nine logarithms decrease of *E. coli* NAR in Desjardine Drain water dispensed into open trays. This experiment was repeated with an isolate of *E. coli* from manure to verify that *E. coli* NAR was not unusually sensitive to sunlight. These experiments demonstrated that sunlight may have bactericidal effects (Palmateer et al. 1989). However, observations from the *in situ* studies show a lesser effect due to sunlight. Other parameters which affect bacterial survival, such as turbidity, may vary between the creek and artificial trays leading to discrepancies between the *in situ* and experimental results.

Bacterial Transport in Suspended Sediment

In a subsequent study, sediment spiked with *E. coli* NAR at 3.3×10^8 bacteria per 100 g was discharged into the Desjardine Drain, along with a fluorescent dye tracer. Within 24 hours, *E. coli* NAR was detected up to 5 km away at all sampling stations with exception of the south beach at Grand Bend. The *E. coli* NAR concentration remained at approximately 1000 bacteria per 100 mL for 2 days, and began to decrease at the point of insertion on day 3. Levels of tracer bacteria were even higher at the south beach station by this time. Tracer bacteria levels declined after a period of 8 days, but were still detected qualitatively up to 85 days following the addition to the drain, after which the bacteria had either died off or passed through the drain (Palmateer et al. 1993; Hayman et al. 1994).

These studies confirm the presence of appreciable levels of fecal indicator bacteria in sediments in an agricultural drain and the increased survival of *E. coli* in sediment compared to water. More importantly, the ability for *E. coli* to survive transport for the entire length of a drain was demonstrated. While some uncertainty surrounds these experiments, some conclusions can be drawn: 1. viable bacteria are able to colonize easily transportable sediment particles; and 2. once adsorbed, transport and extended survival of *E. coli* can occur which may be an important mechanism by which fecal bacteria are carried to the lakeshore.

2.2.4 Major Findings

Summary of 1984 and 1985 Studies:

Several key associations between a variety of factors and fecal indicator bacteria levels were identified in the intensive studies done in 1984 and 1985. These are listed in Table 1 below.

Table 1: A summary of the findings from the 1984 and 1985 studies of bacterial pollution in SE Lake Huron by the Ministry of the Environment.

Factor	Effect on Bacteria	Remarks
Rainfall Events	Increased levels in some instances but not all	Some locations may have too many confounding inputs to respond to rainfall alone (e.g. Goderich)
Lake Roughness / Wave height	Increased due to sediment resuspension	Measurements somewhat empirical and not quantitative
Hour of Sunlight	Decreased levels	ND
Swimmer Density	Some increase	ND
Number of Seagulls	No correlation	ND
River Plumes from Major Rivers	Increased when wind directed plume towards the beach	Related to wind direction
Wind Direction	South-westerly winds increased levels	Influenced lake roughness, wave height and river plume direction
Storm Sewers	Little effect except in unique circumstances	Sewage inputs discovered and remedial action taken to reduce bacterial counts
Sewage Treatment Plant (Goderich)	Difficult to assess	Beaches at Goderich had potential inputs from too many sources
Sewage Bypass Events	Only one event recorded – not significant to data collected during study period	
Boats and Marinas	No significant contribution	
Agricultural Creeks and Drains	Highest Bacterial levels within creeks, high levels at drainage points,	Some evidence that contamination at beaches coming from these drains
Sediment Resuspension	Higher levels with greater turbidity	Somewhat anecdotal as data isn't presented in this manner. Factors and events that cause sediment resuspension also result in increased bacterial levels.
Human Fecal Input	Localized situations (Duffus Creek and Walkers Drain)	Evidence came from investigation of a variety of factors
Agricultural Input	Localized situations and some microbial source tracking evidence and investigative observation	Human fecal input not excluded Some bias due to locations chosen for source tracking studies

ND: Not discussed in detail in these reports

Subsequent studies examining survival in sediment and bacterial transport:

1. Numbers of fecal bacteria detected in sediment from an agricultural drain were higher than in the water column.
2. Survival of *E. coli* in water was long enough to allow for appreciable transport within tributaries and drains.
3. *E. coli* spiked to sediment survives at least one month in sediment.
4. Viable bacteria colonize particles of sizes that are easily transported in streams
5. *E. coli* suspended in water survived long enough to be transported 16 km through an agricultural drain to Lake Huron
6. *E. coli* adsorbed to sediment was transported 5 km through an agricultural drain

7. *E. coli* adsorbed to sediment was detected 85 days following the addition to an agricultural drain.

Conclusions:

The results of these extensive studies demonstrate a number of environmental factors correlated with levels of fecal indicator bacteria at beaches. Lake roughness, wave height and wind direction had the strongest relationships with levels of fecal indicator bacteria. These variables are associated with increased sediment suspension in water. Follow-up studies demonstrated increased survival of fecal indicator bacteria in sediments. Resuspension of lake sediment is one mechanism by which bacterial levels are likely increased at the beach. These bacteria may not reflect recent fecal contamination. However, viability of *Salmonella* adsorbed to sediment was also demonstrated, indicating the ability of fecal-associated pathogens to also survive in sediment. Other factors, such as rainfall, distance from shore and number of swimmers were less correlated with bacteria levels and with correspondence detected only at some beaches.

Sources of high fecal indicator bacteria levels were determined in some localized situations such as Duffus Creek and Walkers Drain. The evidence for these came from investigations of a variety of factors, including comparison of bacterial levels upstream and downstream from potential sources. Detailed studies were performed on an agricultural drain that primarily serviced farmland. Experiments performed using an antibiotic-resistant strain of *E. coli* demonstrated that fecal bacteria delivered to the drain could be transported rapidly to the shoreline of Lake Huron on suspended sediment particles. The drain studies provide strong support for the idea that fecal pollutants washed from rural lands can reach the shores of Lake Huron.

Potential human sources of fecal pollution included the sewage treatment plants and numerous septic tanks scattered throughout the region. With the exception of localized situations, the potential input from septic tanks, and release of fecal indicator bacteria into the environment was not examined in any of these studies. Additionally, data reflecting the presence of seagulls and pets was mentioned, but not examined in detail.

2.3 CURB Program and Selected Past Studies of Microbial Pollution of Fecal Origin by Maitland Valley CA and Ausable-Bayfield CA

Two conservation authorities are active over the watersheds draining to the Lake Huron shores of Huron County. The jurisdiction of the Maitland Valley Conservation Authority (MVCA) extends over the northern section of shoreline to south of Goderich and the jurisdiction of the Ausable-Bayfield Conservation Authority (ABCA) extend over the southern shoreline from of Bayfield to the Ausable River. The conservation authorities conduct programs to promote sound management and conservation of the water and land resources within their areas of responsibility. The MVCA and ABCA have been active for many years in the monitoring, study and management of water quality. Since the 1980s, the CA's have conducted, or participated in numerous activities designed to understand, to monitor and to promote management of microbial water quality pollution in tributaries to the shores of Lake Huron. From the mid-1980s to 1996 the CAs collaborated with the Ministry of the Environment in programs under the Provincial Rural Beaches Strategy. This next section provides a summary of selected elements of this work.

2.3.1 1986 and 1987 – Maitland Valley Conservation Authority Manure Management Studies

In 1985, the MOE developed the Provincial Rural Beaches Management Strategy Program which identified watersheds having beaches that were potentially impacted by agricultural practices. Seven of these watersheds were classified as high priority and targeted for special studies, with the Maitland Valley watershed being among them. The MVCA initiated the Manure Management Study (MM) in 1986, which was supported with funding, scientific and technical assistance by the MOE (Evans and Fuller 1987; Foran and Fuller 1988) with the objective of improving water quality in the Maitland watershed and the adjacent onshore waters of Lake Huron. The long-term goal of the program was to prevent closures of beaches on Lake Huron due to bacterial pollution originating from agricultural sources within the Maitland Valley watershed. The studies aimed to achieve this goal by working to: 1) determine the extent of bacterial pollution occurring from livestock sources in the Maitland Valley watershed and its resultant effect on Lake Huron beaches; 2) identify livestock sources of pollution on a sub-watershed level; 3) promote greater awareness of farms sources of water pollution; and 4)

promote better manure management practices through the adoption of remedial measures to reduce water pollution from agricultural sources (Evans and Fuller 1987). A two pronged approach was taken to reach these objectives. Within the sub-watersheds under study, information on agricultural practices from, high water pollution risk farms was gathered through site visits and questionnaire surveys. As a first approach, these visits allowed the promotion of remedial measures and of the financial assistance available through programs sponsored by the Ontario Ministry of Agriculture and Food (OMAF). A second approach was a water quality study that sampled for indicator bacteria (fecal coliforms, fecal streptococci, *P. aeruginosa*, and *E. coli*) and chemical parameters at selected sites throughout the study area. Sites were selected downstream from micro-basins of polluting farms in order to describe the cumulative effects. In addition, when permission was given from the farm owner, sites immediately upstream and downstream from the individual farms were sampled.

In the 1986 MM study, weekly samples were taken between June and September at 17 sites throughout the southeast corner of the Maitland River watershed. Results showed that the 17 stations sampled routinely exceeded MOE surface water quality guidelines for both chemical and bacterial parameters. Where upstream and downstream from farm samples were taken, the geometric means of fecal coliform counts increased downstream of farms. This was thought to be directly related to cattle access to the immediate water course. Of note are the results for one farm where samples were collected over June to August at sites above and below a pasture where cows had access to a watercourse. The geometric mean of fecal coliform for the sample period was 677 /100ml at the upstream sites compared with 5896 /100mL at the downstream site.

A follow up study was conducted in 1987, and this investigation was extended to include the middle Maitland River upstream of Brussels and the Little Maitland River upstream of Mayne. During this study, weekly water quality samples were collected between March and December at 24 sites. These samples were analyzed for bacterial indicators, including *E.coli*, and a number of water chemistry variables. As in 1986, a large proportion of samples (>97%) had fecal coliform levels in excess of the PWQO of 100 FC/100mL at the time. Geometric means of FC for the study period ranged from 264 to 6730 /100mL among stations with one exception where counts were low (85 /100mL). Wet weather geometric means of FC were typically higher at all but one station and indicated a strong negative relationship between rainfall events and water quality over the study area. Levels of *E.coli* were similar to FC in most samples and comprised 71 to 94 % of the measured FC.

The short-term effect of remediation efforts on water quality was captured by sampling at farm demonstration sites where remedial work was undertaken. Sampling at sites above and below a pasture for periods with and without cattle in the fields, and periods before and after restricted cattle access to the tributary, indicated that cattle access to the watercourse had an adverse effect on levels of FC. However, there was little improvement in water quality downstream during dry weather. Results from another farm site, where samples above and below an area of cattle access and runoff from a manure storage pile indicated substantially higher FC levels downstream. Fecal coliforms counts declined precipitously when the cattle were removed from the pasture and the manure pile was removed. The conclusion reached by the authors was that the common livestock management practices employed on the farm had detrimental impacts on water quality (Foran and Fuller 1988). The potential effect of cumulative impacts of multiple farms, however, was demonstrated at a third upstream and downstream pair of sites where the elimination of runoff from a manure pile failed to show an improvement on water quality. The poor background (upstream) water quality was thought to have obscured any measurable improvements of the remediation work. The study acknowledged, however, that, in these study areas, the impacts of livestock sources on bacterial levels of Lake Huron beaches remained to be determined (Evans and Fuller 1987).

The survey results identified three potential factors common to all the farms that participated: 1) runoff from solid manure storages; 2) improper disposal of milkhouse wastes; and 3) livestock access to streams. The majority of farmers surveyed also felt that financial assistance for manure storage systems improvements should be provided by the provincial government, as the capital spent on such projects would benefit society at large. Recommendations for increased funding to farmers for improving their manure storage facilities were made by the study, as well as development of planning guidelines for use by municipalities to implement manure management regulations or by-laws.

2.3.2 1982-1987 – Ausable Bayfield Conservation Authority Manure Management Awareness Program

In 1982, the Ausable Bayfield Conservation Authority (ABCA) initiated the Manure Management Awareness Program (MMAP). The goals of this program were to facilitate the understanding of the sources and causes of manure pollution from farms and to promote remedial action by

farmers to improve water quality (Ryan 1987). There were three program elements: 1) farm site visits to better understand the causes of manure pollution; 2) an education component to promote remedial measures in the farm community at large, and 3) a water quality monitoring program at selected agricultural drains.

The water quality component of the study focused on the identification of problem areas. Surveys of bacterial indicators and water chemistry were conducted at 10 agricultural drains in 1985 and 1986 (Ryan 1987). These were drains previously surveyed in 1984 by MOE as part of the Lake Huron Beaches study (Palmateer and Huber 1984). From March to December 1986 the levels of FC were <100 cfu/100 mL in fewer than 24% of 347 samples collected at the drain sites. Geometric means over the period ranged from 162 to 1,332 cfu/100 mL among the sites. Maximum levels exceeded 10,000 cfu/100 mL at all but two sites. There was a systematic seasonal variability in FC geometric means, with significantly lower FC counts in the spring (March to May) than the summer (June to August). Levels of FC were positively related to turbidity at some sites. Levels of *E.coli* were monitored for a limited time over which *E. coli* levels were noted as varying little from those of FC. Water quality varied among drains. Initial surveys of land use suggested a relationship with livestock operations and it was advocated that an inventory of land use impacting water quality be undertaken. The levels of FC observed at drains in the 1986 surveys were similar to the levels observed in 1985.

The farm contact component of the program relied on the interpretation of air photos, supported by their ground truthing, to select candidate farms to participate in the surveys. In 1985 and 1986, 83 and 84 farms were contacted and requested to participate as consultants and to complete a questionnaire designed to evaluate the aspects of manure management and attitudes towards manure-related environmental concerns. Solid manure storage was the prevalent mode of storage of manure and judged to be a common source of manure contaminated runoff (Ryan 1987).

2.3.3 1987-1991 – Ausable Bayfield Conservation Authority Target Sub-Basin Studies

The ABCA initiated a target sub-basin study in 1986 in cooperation with the Provincial Rural Beaches Strategy Program (Hocking 1988). The first objective of the study was to investigate and address agriculturally related pollution sources in a tributary sub-basin selected for detailed

study. The intent was to improve water quality in the target sub-basin through remediation projects and evaluate the effectiveness of these corrective measures through a water quality sampling program. A second objective was to develop and promote an abatement strategy for bacterial contamination (Hocking 1987). Further, the study was meant to assist with MOE sponsored abatement activities and rural water quality research sponsored by ABCA and MOE. The findings from this 1986-1991 study are summarized by Hocking (1992).

The Desjardine Drain in the Parkhill Creek watershed, which discharges into the Ausable River near Grand Bend, was selected as the study (target) sub-basin. A nearby sub-basin just south of the Desjardine Drain, the Turner Drain, was chosen as a reference sub-basin. Several farms in the target sub-basin participated in remediation projects designed to reduce the bacterial loading into the Desjardine drain. Funding for projects, such as manure storage improvements, run-off tank construction and septic systems was provided under the program.

The water quality component of the study was intended to measure improvements, if any, in water quality downstream from the remediation projects. Measures of water quality included fecal indicator bacteria and nutrients. Samples were collected at sites along the Desjardine Drain selected to bracket remediation sites and landscape features. The control sub-basin was sampled at two locations in the headwaters of the Turner Drain. Other sampling locations in Parkhill Creek were sampled including a site downstream from the point where both the target and control sub-basins discharged into Parkhill Creek and a site downstream from the point where only the control sub-basin discharged into the creek.

Water sampling during the pre-remediation periods in 1986 indicated appreciable levels of *E.coli* and FC at points over the study area with the highest geometric means over the sampling period observed at headwater sites. Water quality in the Parkhill Creek watershed was judged to be poor. Of the 17 sites in the Desjardine Drain, geometric means of *E.coli* exceeded 1,000 cfu/100 mL at eight sites with maximum levels exceeding 10,000 cfu/100 mL at most sites. In the Turner Drain, geometric means for the survey period exceeded 1000 cfu/100 mL at two of five sites and maximum values exceeded 10,000 cfu/100 mL at all but one site. An evaluation of the lands within the watersheds of the study basins indicated that in some cases, sources of the contamination could be traced directly to surface drains. Contamination of drains by domestic septic systems was observed and noted as being common (Hocking 1987).

There was an appreciable variability in the levels of *E.coli* at sites observed among the six years of monitoring (1986 to 1991). Years with higher flow in Parkhill Creek (1986 and 1991) tended to have higher geometric means of *E.coli* but not all sites. The observation that some sites exhibited a lower response to high flow in 1991 was taken as evidence of the positive effects of remedial actions taken since 1986 (Hocking 1992). Mean annual levels of *E.coli* were highest in 1986 at all but one site and were lowest in either 1989 or 1991, the years with the lowest discharge in Parkhill Creek (as inferred from discharge to Parkhill Creek reservoir).

Over the six years of study annual geometric mean levels of *E.coli* at sites in the Turner Drain, the reference watershed, ranged from ca. 600 to <4000 cfu/100mL at the north branch site and from ca. 150 to <600 cfu/100mL at the south branch site. Interestingly, the sites with the least variable and lowest annual geometric means were the sites nearest to the lake on the Ausable River. Annual geometric means ranged from ca. 60 to 250 and ca. 40 to 160 cfu/100 mL at these sites, respectively.

The levels of *E.coli* in bed sediments were also monitored at a subset of sites in the Desjardine Drain. In 1986, the levels of occurrence ranged from 10^3 to 10^7 cfu/100g, with many of the observations in the range of 10^4 to 10^6 cfu/100g. Three sites were further sampled between 1986 and 1989. The sites included a location near a farm where remedial work was undertaken, a location near a farm with no remedial work and a reference location. Fecal bacteria levels in sediments were appreciable at all sites and evidence for improvements at the remediation site was inconclusive. In 1986, the year with highest upper range in *E.coli* samples among sites, levels ranged from just under 10^5 to 10^7 cfu /100 g of sediment.

2.3.4 1989-1996 CURB Program (MVCA and ABCA)

The CURB program was a component of the Provincial Rural Beaches Strategy Program (PRBSP) which was developed in 1985 by the Ministry of Environment. The PRBSP functioned with the participation of other Provincial Agencies and working at the local level in cooperation with Conservation Authorities. The objective of the program was to identify the relative impact of pollution sources, and develop a course of action leading to restoration and long term maintenance of acceptable water quality at provincial rural beaches. The premise behind the program was that there was a significant enrichment of bacterial contamination in southern

Ontario rivers and lakes originating from rural sources. Known sources highlighted at the time included urban sanitary and stormwater runoff, direct livestock manure access to watercourses, inadequate manure management practices, direct discharge of milkhouse wastes, contaminated field tile system and faulty septic systems. Further, there was recognition that these sources, singly, or in concert, could adversely impact beaches' recreational uses.

The PRBSP provided funding and technical assistance to Conservation Authorities to conduct studies and programs directed at advancing the objective of the PRBSP. A key activity was the development of CURB plan. The CURB plan was intended to identify the relative contribution of different pollution sources and the modes of delivery to the beaches. Further, the plan was to provide an analysis of remediation priorities and options culminating in a remedial strategy for identified pollution sources. Development of the MVCA and ABCA CURB plans occurred in the late 1980s. The PRBSP advanced to an implementation phase of CURB in 1991. During the implementation of CURB, support for remediation projects was provided by the Province and the CAs lead the solicitation, evaluation, selection and assessment of remediation work designed to mitigate the delivery of bacterial pollutants to watercourses from rural lands. The implementation phase was terminated in 1996 as presumably was the PRBSP.

CURB Plans:

The information amassed from the previous work mentioned as part of the Provincial Rural Beaches Strategy Program contributed to the design and implementation of the Provincial Clean Up Rural Beaches Program (CURB). In the late 1980s CURB activities focused on identifying and prioritizing sources of bacterial pollution within watersheds on a provincial scale, with the MVCA and ABCA participating in the program (Fuller and Foran 1989; Hocking and Dean 1989). The MVCA and ABCA completed CURB plans in 1989.

The relative magnitude of key sources of bacterial pollution (as represented by levels or loads of FC) identified in the watersheds under the jurisdictions of the MVCA and ABCA were evaluated and used to evaluate priorities for remediation in the CURB plans. A modeling exercise based on the application of the Pollution from Livestock Operations Predictor (PLOP) model developed by Ecologistics Ltd. was used in the source evaluation work along with various types of demographic and land use information and information from water quality monitoring and research. While the PLOP model was designed for general application in the CURB program,

each conservation authority had the ability to change specific parameters within the model as per the conditions in their watershed. Data that was used as input for the model came from voluntary surveys distributed to participating farmers within the watersheds.

A further objective of the CURB plan report was: 1) to outline strategies to reduce fecal pollution loads from specific sources, and, 2) work towards preventing postings of public beaches (Fuller and Foran 1989) or ensure good water quality at beaches (Hocking and Dean 1989).

A substantial base of information was collected by the MVCA from 1986 to 1988 on farm features in its jurisdiction through 431 (out of 3,100) farms visits and questionnaires in the MVCA jurisdiction. This information was used in the development of the MVCA CURB plan. The site visits were supplemented by a watershed inventory based on visual observations of livestock operations. Based on the collection of information on farming practices, four activities with a potential to pollute were routinely identified: 1) runoff from barnyards and manure stacks, 2) inadequate milkhouse waste disposal, 3) unrestricted cattle access, and 4) winter spreading of manure.

The information collected by ABCA through the Manure Management Awareness Program which operated from 1982 to 1988 was used to provide input into the modeling conducted as part of the development of the ABCA CURB plan (Hocking and Dean 1989). Surveys were conducted at farms with information collected on size of livestock operations, manure storage facilities, milkhouse waste treatment, and access to water courses. A total of 171 out of 826 farms identified as having pollution potential in the ABCA's jurisdiction were surveyed in detail. The program also included water quality surveys in rural drains and a provision of technical assistance to livestock farmers to reduce negative impacts on water quality. Information collected in the ABCA's Target Sub-basin study was also used in the development of the CURB plan.

Many of the potential bacterial inputs considered in the analysis of sources were based on features of livestock operations. Urban non-point sources of bacterial contamination were also identified and estimated by the MVCA as a flat rate of 3.1×10^{10} FC/ha. Loading from sewage treatment plants were evaluated on the basis of monitoring and operational data collected from the plants. In the MVCA work, loading contributions from septic systems were based on estimated rural population size and a series of assumptions including: concentration of fecal bacteria in drainage tile effluent (10^6 fecal bacteria/L), per capita effluent volume (275 L/day),

septic system failure rate (30%) and the associated delivery rate (50%). The ABCA based predictions on loads from septic systems on similar, but modified assumptions. Results of a voluntary survey which provided participants grants to upgrade their septic systems was used to guide the estimate of failure of septic systems which was set at a rate of failure of 60%. The volume of effluent per day was set at 340 L, however, lakeshore septic systems were treated separately and the contribution reduced by 50% to reflect seasonal usage. A 100% delivery of effluent to a watercourse was used.

Wildlife population at beaches or within the watershed was not considered as a significant contributor of fecal pollution and was not included in the analysis by the MVCA or ABCA.

The PLOP model was designed to address the potential loading of bacteria by livestock operations to an immediate stream or watercourse. Considerations of transport or delivery of the bacteria from the stream to other downstream areas (i.e. beaches) had to be built into the original model. A general delivery model was used by the MVCA and ABCA conservation authorities, a part of which described hydrologic information in the form of contaminant travel times within and between drainage areas. These travel times were determined independently from local hydrologic information. For example, the MVCA divided loadings as being influenced either by baseflow conditions (“continuous sources”) or event flow conditions (“pulse sources”). “Continuous sources” included milkhouse wastewater, livestock access, septic systems, and sewage effluent. “Pulse sources” included manure stacks, feedlot/barnyard runoff, manure spreads, manure spills, and urban runoff. This design did not incorporate the responses from “continuous sources” under event flow conditions (i.e. septic system following a rain event). Flow rate data from hydrographic gauging stations which represented increases in stream discharges, were assumed to be due to rain events, and were used as “event” flows in the model. The lack of temporal information with regard to the rain events did not allow for any assessment of how the different “pulse” and “continuous” sources behaved in relation to each other under the influence of variable levels of precipitation.

A bacterial decay rate was required in order for the delivery model to account for daily bacterial die-off as the load traveled to the Lake Huron shoreline, and seasonal decay rates were assumed. The MVCA used 0.2 log/day for winter, 0.26 log/day for spring and fall, and 0.35 log/day for summer. The ABCA used a flat decay rate of 0.38 log/day. How these rates specifically reflect bacterial population dynamics and other fluctuating physicochemical parameters (e.g. UV radiation, temperature) is not explained. The decay rate is also insensitive

to “event” conditions such that, for example, effects of a storm on sediment transport could not be described. This would potentially diminish the estimated contributions of erosion-sensitive sources (e.g. fields with a higher grade which contribute to higher amounts of runoff) on a temporal scale.

The MVCA expressed the model predictions on the basis of FC load to watercourses and to Lake Huron and further discriminated between the loads to the Maitland River and those to the sub-basins north and south of Goderich draining directly to the lake (drainage area south of Nine Mile basin and north of Bayfield basin). The breakdown of loadings to the lake from the Maitland River and those from the sub-basins are given in Table 2. The predominant sources of bacteria to the lake predicted from the PLOP model were septic system failure, winter spreading of manure and livestock access to water courses. These sources were defined as the priority sources in the plan. Other sources were predicted as relatively minor in terms of global loading. The magnitude of the loadings of FC from the watershed reaching the lake was predicted to be appreciably less than total loading delivered to tributaries, however, the dominant sources of the load remained similar. An interesting feature of the analysis was the relatively high importance of the shoreline watershed component of the load in terms of the load reaching Lake Huron, despite its overall smaller magnitude than that of the Maitland River. This was particularly true for the septic system component of the predicted load. Urban non-points sources from the town of Goderich were also singled out as a significant source of loading reaching the lake from the shoreline watersheds (6%), however, the basis for the claim is unclear. The majority of the load to the lake (75%) and the watercourses (88%) was predicted to originate from “continuous” sources suggesting that “pulse” sources might be less important than intuitively predicted. Analysis of load forecasts from sub-basins lead to the prediction that sub-basins closest to the lake contributed a disproportionately higher load than the respective land areas would suggest. The main Maitland River basin and the shoreline basins accounted for 70% of the predicted load to the lake yet composed only 29% of the land area. The conclusion was that targeting the remediation of the sub-basins closest to the lake would most effectively reduce loads to the lake (Fuller and Foran 1989). The fecal bacteria loads predicted by the modeling was validated against measured loads at three sites on the Maitland River using data from seasonal water quality sampling in combination with flow measurements. This was completed for five site & year combinations with the range in the predicted to observed loads from 0.34 to 0.74.

The ABCA expressed load results much the same as the MVCA with the exception that the contribution from shoreline areas (lakeshore cottages and Lake Huron Gullies) was treated

separately. Similar to the MVCA load predictions, the dominating role of septic systems was evident accounting for the majority of the load reaching the lake, followed by winter spreading as the primary predicted sources (Table 2). Proximity to the lake was again a factor in determining the proportion of the load to the environment that potentially would reach the shores of the lake. The Ausable River was predicted to contribute only 1.4 % of the load of fecal bacteria to the lake despite accounting for 48% of the land area in the ABCA jurisdiction and being the dominant source of loading of bacteria to the environment. Surprisingly, the lakeshore cottages and Lake Huron gullies were predicted to account for 69% of the total load to the lake while they made up 12% of the combined drainage area examined. The more intriguing observation was that this area had only 9.1% of the livestock farms that were judged to be potentially contaminating and 24.5% septic systems in the study areas. Again, “continuous” sources were predicted to load more bacteria to the lake than “pulse” sources. The ABCA report acknowledges that the role of sediments in facilitating the deliver of bacteria to the lake may be under represented in the analysis and cautions that the load estimates are first approximations that should be used with prudence beyond the purpose of the CURB plan.

Table 2: Percent fecal coliform load delivered to Lake Huron by various sources as determined by the PLOP models used by the Maitland Valley and Ausable Bayfield Conservation Authorities in the development of CURB plans.

Source type	MVCA (%)	ABCA (%)
septic system failure	65	77.5
winter manure spreading	21	16.1
livestock access	8	2.7
spring to fall manure spreading	0.4	2.5
feedlot runoff	0.2	1.06
milkhouse waste	1	0.10
manure stack runoff	0.3	0.02
urban STP	1	0.13
manure spills	0.4	n/a
urban non-point sources	3	n/a

The source-specific model predictions in the CURB plan to do not appear to have been tested, nor does it appear that were some of the key assumptions in the model were evaluated or verified. Given the complexity of environmental system the model was attempting to describe

combined with the uncertainty in the reliability of the modeling approach, the predictions derived from the model must be viewed with caution. The information provided on the results of the modeling exercise serve as record of the directions identified by the CURB plans and to illustrate the approach that was taken to manage fecal pollution at the time.

Implementation of CURB Program:

The implementation of the CURB program was announced in 1991 and ran until the termination of the program in 1996. The MVCA and ABCA actively participated in the CURB implementation with final CURB reports released in 1996 (Hocking 1996; Loeffler 1996).

A focus of the implementation phase was the solicitation, selection and implementation of proposals for local-scale remediation projects designed to reduce the load of bacterial pollutants and nutrients from various rural sources expected to impact surface and beach water quality. Evaluation of proposals for remediation projects and allocation of grants were managed and administered by each CA. Conservation authorities were responsible for identifying and prioritizing “high risk” or “high potential” polluting farms. Many remediation projects were completed with direction and financial support under the CURB program by the MVCA and ABCA. The MVCA and ABCA final reports cite 752 and 670 water quality improvement projects completed, respectively. Remediation activities supported included, projects to restrict cattle access to water courses, replacement of septic systems, improvements of manure storage facilities and improvements to milkhouse waste handling. It is of note that the final MVCA CURB report indicated that of the ca. 5,000 potential sources of bacterial contamination in its watershed, only 15 to 17% of these had been addressed over the CURB implementation years.

As part of the CURB implementation the MVCA also conducted a water quality monitoring program at selected tributary and drain sites beginning in 1992. The surveys were in part designed to evaluate the impact of remedial actions at selected sites. The 1992 annual report (Loeffler and Robinson 1993) provides the initial collections of data for 16 sites, however, analysis of the remediation effects were restricted due to limited sampling periods or the lack of follow-up surveys. The 1993 (Loeffler and Robinson 1994) annual report provides results for 17 sites, 11 of which were selected to evaluate remediation projects and the others to monitor trends in water quality. With the exception of a site near a tile drain discharging diverted

milkhouse waste and manure pile runoff, results did not conclusively demonstrate a reduction in *E.coli* after the implementation of remedial actions. The levels of *E.coli* were appreciably elevated at most of the sites which may have obscured any changes due to remediations. In 1994, additional surveys were conducted (Loeffler and Strome 1995). Upstream and downstream collections at a beef farm where cattle access to the water course was eliminated demonstrated a reduction in levels of *E.coli* at the downstream site from the previous years, yet downstream levels were still elevated and higher than at a site further upstream. Nevertheless, appreciable reductions in levels of *E.coli* were observed from 1992 to 1994 at sites downstream of several remediation projects including septic system replacements, milkhouse waste and manure storage projects. The geometric mean of *E.coli* in 1992 was >1200 cfu/100mL compared with < 200 cfu/100 mL in 1994. Furthermore, water quality improvements (*E. coli* geometric means) for six sites monitored between 1992 and 1995 were noted in the MVCA CURB program final report (Loeffler 1996). In 1992 geometric means were >300 cfu/100mL at all but two sites while in 1995 medians were <200 cfu/100mL at all but two sites. In the absence of any additional information it is difficult to interpret if the trend represents long-term improvements or inter-annual variability.

The MVCA CURB 1991 progress report (Loeffler 1992) notes an under estimation of the significance of the role of tile drains in transporting manure applied to field in the CURB plan (based on information from new studies by the ABCA). Furthermore, it states that the number of farms with high potential to pollute was likely underestimated in the CURB planning phase, however, no revisions of the modeled loads were provided.

During the implementation phase of CURB, the ABCA continued with water quality surveys in the Desjardine Drain and added surveys to evaluate selected remediation projects (Hocking 1993). In the 1992-1993 CURB progress report (Hocking 1993), information is presented for 11 site surveys over the Desjardine Drain, Turner Drain, Bayfield River, Parkhill Creek and Ausable River near Fort Franks. The median values of *E.coli* over surveys in 1992 were >100 CFU /100mL at all but two sites and said to be statistically similar among all sites. Improvements in water quality were observed at three upstream and downstream pairs of sites evaluated after remediation work (manure storage and septic system repairs). In the 1993-1994 progress report (Hocking 1994), water quality results from additional remediation projects (septic system repair and milkhouse waste storage) were reported and suggested variable success at improving downstream water quality. A summary of data collected in 1993 at the 11 sites of ongoing water quality sampling were also presented; there was little obvious changes in levels

E. coli from the previous year. The report also provided a listing of the number of days of beach postings at selected beaches in the area of ABCA responsibility from 1986 to 1993. Notable is that a number of beaches experienced postings in 1992 which was attributed to higher discharge from surface drains that year, however, no information is provided to substantiate this claim.

The final CURB report of the ABCA (Hocking 1996) provides a summary of water quality results for 10 sites surveyed in the monitoring component of program and discussed in previous CURB reports. In some cases, sites had been sampled annually from 1986 to 1995. The time plots of annual geometric means and maximum values of *E. coli* provided little clear evidence of progressive changes in levels over the years with the exception of sites which experienced maximum levels exceeding 10^6 *E. coli* cfu/100 mL in 1986 and where appreciable remedial work had been completed. At a site near the mouth of Parkhill Creek, annual means were near 100 cfu/10mL over several years including 1987, 1988, 1994 and 1995 and suggested little change over the study period.

The final report noted that while the improvements were not as great as hoped, the research and innovative projects under the ABCA CURB contributed to the understanding of the ABCA watershed (Hocking 1996).

While not directly part of CURB implementation the need to understand movement of fecal pollutants to surface waters associated with common agricultural practices prompted several process-investigative type studies by the ABCA in early 1990s. For example, studies were conducted to investigate operational factors affecting movement of bacteria after manure applications to tiled lands (Foran 1992; Foran and Taylor 1993).

2.3.5 1995-1986 Rapid *E. coli* Test Evaluation

During the summers of 1995 and 1996 MOE, ABCA, and GAP EnviroMicrobial Services Inc collaborated in a field evaluation of a rapid *E. coli* detection method for Health Canada (Glaskin-Clay et al. 1996). The test that was examined was based on the measurement of the activity of β -D-glucuronidase, an enzyme present in most strains of *E. coli*, as a quantitative measure of *E.*

E. coli abundance. Frequent sampling of *E. coli* at selected Lake Huron beaches was conducted during the summers of 1995 and 1996 in support of the study. Multiple sample sets per week (up to five per week), and in some cases multiple sample sets per day were collected. While the focus of the study was an evaluation of the performance of the *E. coli* assay under variable environmental conditions, the environmental analysis of the dataset by Glaskin-Clay et al. (1996) provided insight on the variability in occurrence of *E. coli* among beaches and over short time intervals.

An evaluation of incubation times found that six hour incubations produced acceptable correlation coefficients between the results of the assay method and those of the standard membrane filtration method. The study promoted the idea that beaches could be sampled at 6 a.m. so that results could be ready for beach managers by 12 noon, just before peak beach user time (2-4 p.m.). The most extensive sampling was completed at Grand Bend beach where, in 1996, samples were taken at 6:15 a.m. and analyzed at 12:00 p.m. for *E. coli* levels by membrane filtration and the assay methods. This study found that the correlation between the results of the membrane filtration method and the rapid *E. coli* test was weaker when *E. coli* levels were above 100 cfu/100 mL. This was attributed to the relation between turbidity and high *E. coli* levels and the presumption that turbidity interferes with the β -D-glucuronidase assay.

Levels of *E. coli* were correlated ($r > 0.54$ to 0.77) with wave height at six of eight lake sites and it was suggested that this correlation was related to the resuspension of bed sediments contaminated with *E. coli* into the water column. However, there was no direct information on levels of suspended solids or turbidity at the time of sampling.

Rainfall and water temperature were not found to be related to increased *E. coli* levels (Glaskin-Clay et al. 1996). Observations on the presence of seagulls suggested that seagulls were not related to elevated levels of *E. coli*, however, estimation of seagull abundance was approximate (beaches with five gulls or more were considered as having the potential for gull fecal input irrespective of the numbers of individuals present).

No significant differences were found in *E. coli* levels between early morning and mid-day samples based on fifty-four paired comparisons from the Grand Bend beach site.

In 1996, geometric means for the study period ranged from 57 to 234 cfu/100 mL *E. coli* among the eight beaches. The two beaches where the mean exceeded 100 cfu/100 mL were the two

shallowest beaches with the lowest slope. Beach quality was improved in 1996 over 1995 when geometric means for five of the same sites ranged from 109 to 259 cfu/100 mL.

The raw *E. coli* data for the 1995 study are provided in the Hocking (1996) report along with a summary of beach postings for the year at the study beaches.

An unspecified amount of sampling for *E. coli* was also conducted at nine non-beach sites including creeks, drains and a harbour as part of a methods comparison investigation. Levels of *E. coli* were periodically elevated at these sites with maximum levels ranging from 1,000 to 10,000 cfu/100 mL among sites (geometric means ranged from 70 to 381 cfu/100 mL).

2.4 Monitoring of *E.coli* at Recreational Beaches by the Huron County Health Unit

The Huron County Health Unit (HCHU) monitors public beaches in Huron County in accordance with guidelines published by the Ontario Ministry of Health and Long-term Care (1998) and provides advice to the public on beach water quality and suitability for bathing. Beaches on the shoreline of Lake Huron ranging from Amberley Beach, south of Point Clark to Port Blake Park, south of Grand Bend are regularly sampled for *E. coli* over the swimming season (late May to end of August) (Figure 1), although the frequency of sampling and the initiation of sampling in early summer have varied among years. Water samples are usually collected at a depth of approximately 1-1.5 m. Multiple samples, usually sets of five replicates in recent years, are collected per sampling event and used to calculate geometric mean abundance estimates which are used by the Health Unit to report on short-term (e.g. website postings) and long-term (e.g. Brodsky 2003) water quality, which is then used to develop advice for the public on suitability for water recreation.

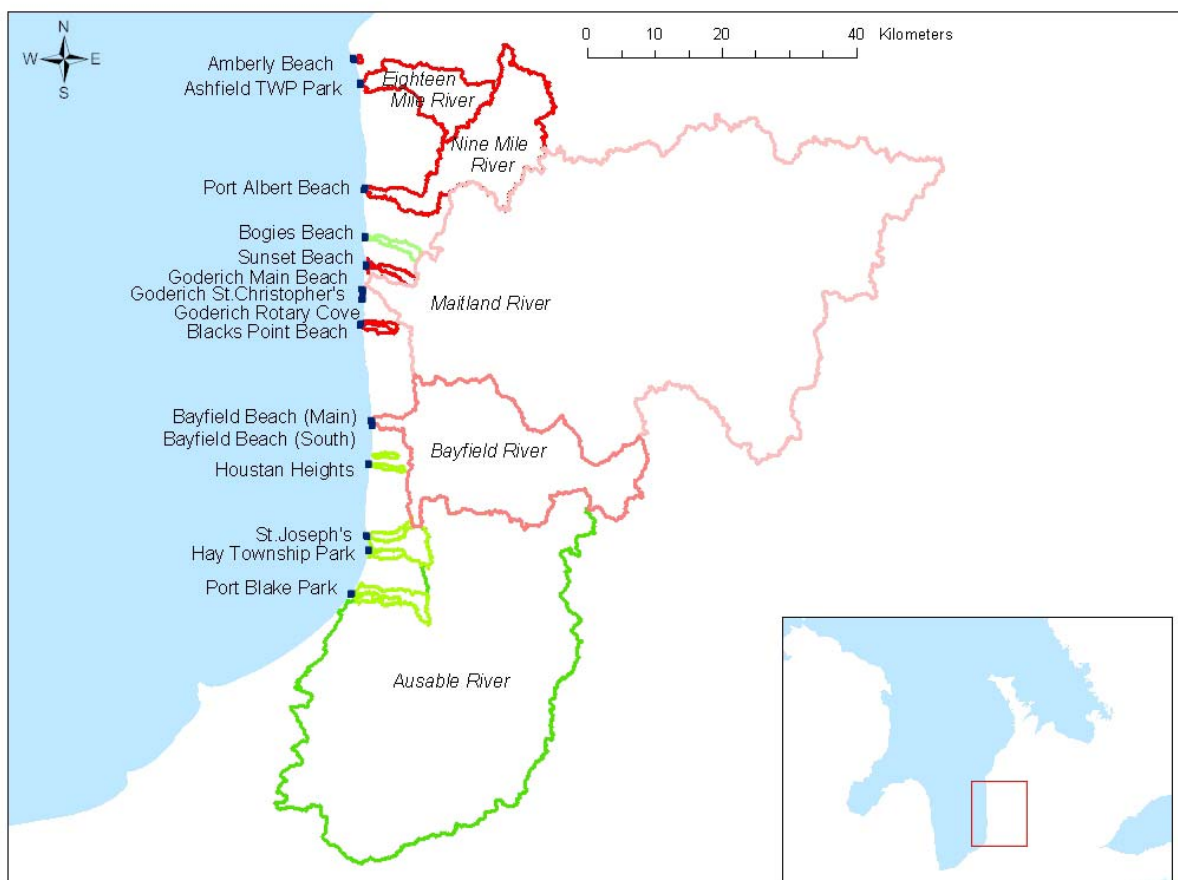


Figure 1. Recreational beaches on the shores of Lake Huron monitored >5 times in 2003 by Huron County Health Unit. The coloured areas define drainage areas immediately adjacent to the beaches. Data for the 10 beaches where drainage area is outlined in red/pink were examined in detail in this report.

The beach monitoring data collected by HCHU over the period 1993 to 2003 was examined in detail for ten beaches on the shoreline of Lake Huron. The beaches were selected to cover a range of physical and environmental conditions encountered over the Great Lakes shoreline of Huron County and include the five beaches posted for the 2003 season as unsafe for swimming. A criterion in the selection of beaches was that the beach had been monitored on a regular basis (>5 visits/season) over the last 11 years. Patterns of occurrence in *E.coli* over the years and among beaches were examined. Correspondence between patterns of occurrence in *E.coli* and environmental factors thought linked to loading, transport or survival of *E.coli* were also examined. The analysis empirically explores associations between the occurrence of *E.coli* and the potential sources and modes of delivery of *E.coli* to beaches. It further serves to confirm findings from earlier studies and contribute to the understanding of microbial fecal pollution at these beaches.

A potential limitation of the analysis is that the design of the beach monitoring programs targeted the collection of specific data for a defined purpose. As such, adequate information to effectively identify environmental correlates may be lacking, and the absence of a correlation between *E. coli* data from the beach monitoring data and an environmental feature may simply be a consequence of inadequate or inappropriate data. This analysis assumes that beach monitoring methods have been consistent through time and recognizes that variations in sampling protocols among years (e.g. sample depth or time of day) may introduce additional variability into the analysis.

Table 3: Beaches monitored by the Huron County Health Unit selected for analysis of *E.coli* monitoring data from 1993-2003

Beach	Substrate (beach)	¹ Approximate Distance from 1 to 3 m depth contours (m)
Amberley	sand (depositional)	700
Ashfield Township Park	sand (depositional)	980
Port Albert	mixed beach (70% sand/30% pebbles)	140
Sunset Beach	mixed beach (50-60% sand/40-50% pebbles)	200
Goderich Main Beach	mixed beach (80% sand/20% pebbles)	200 ⁴
Goderich St.Christophers	cobble and bordered by boulder groins ² ?	200 ⁴
Goderich The Cove	mixed beach (80% sand/20% pebbles)	320
Blacks Point	exposed sediment bluff & mixed beach(70% sand and 30%pebbles)	300-400 ⁴
Bayfield Main	mixed beach (80% sand/20% pebbles)	140
Bayfield South	sand ³	250

Note: beach substrate information is from the Great Lakes Environmental Sensitivity Atlas (Environment Canada 1994); bathymetry data based on a draft version of the NOAA compilation of 1m depth contours for Lake Huron except Goderich Main and St.Christophers which is from CHS chart 2261. 1- The maximum distance (over the beach areas monitored by HCHU) between 1 to 3 m depth contours for lines perpendicular to the shoreline. 2- classified as retaining wall and harbour structure in Great Lakes Environmental Sensitivity Atlas; 3- classified as rip-rap in Great Lakes Environmental Sensitivity Atlas; 4- complex depth contours make determination very sensitive to the shoreline angles used to estimate inter-contour distance. The surficial sediments of the nearshore of Lake Huron along the shoreline of Huron County are identified as consisting of sand and undifferentiated till or bedrock by Thomas et al. (1973).

2.4.1 Patterns of Occurrence

The PWQO for *E. coli* of 100 cfu/100 mL, has, over the years, been exceeded at the ten beaches selected for analysis (referred to subsequently as study beaches). At times the degree of exceedance has been appreciable. Among the years 1993 to 2003, 20% to 45% of the composite total number of samples sets (i.e. geometric means) collected at these study beaches has exceeded the PWQO (Figure 2). The data suggest that there are good and bad

years with respect to levels of *E.coli* detected in the beach monitoring, and median levels of *E.coli* and the numbers of sample sets with geometric means above the PWQO have shown large interannual variability (Figure 2). The years 2000 and 2001 stand out as relatively poor years and 1995, 1999 and 2003 as relatively good years (Figure 2). The composite of the data failed to indicate a consistent and systematic temporal trend in the levels of *E. coli* at these beaches over the period under study (1993 to 2003).

The range in median values across samples sets for individual beaches can be highly variable within one year. In 2002 and 2003 the range was relatively small among sites (all <50 cfu/100mL) in contrast to 2000 when the range was substantial (under 50 to over 300 cfu/100mL) (Figure 2). At face value this suggests that in the poor years, some beaches are more susceptible to the factors that contribute to elevated *E. coli* levels than other beaches.

Median values of geometric means across sample sets collected in a year are used to represent the annual levels of *E.coli*. At times, the beach monitoring programs reported an upper limit of *E. coli*. It is thought that by using median values to assess trends and relationships, the patterns in the data would be less susceptible to these fixed upper limit values and thereby would facilitate interannual comparisons.

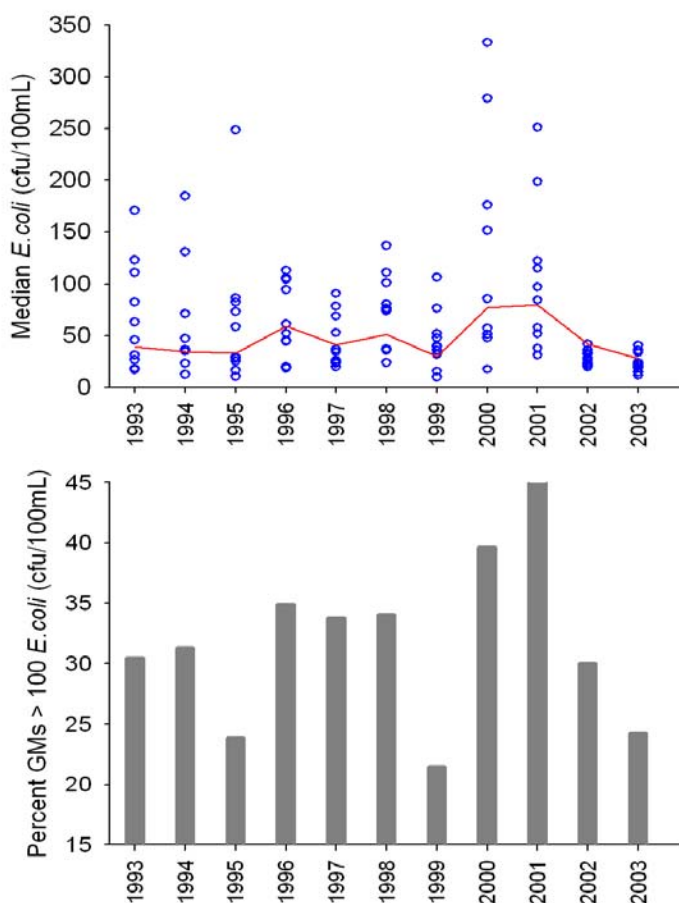


Figure 2: The annual median values of geometric means of *E. coli* (cfu/100mL) for samples sets for the 10 study beaches (blue circles) along with the median values across sample sets at all beaches (red line) is given in the top panel. The percentage of all sample sets where the geometric mean exceeded 100 *E. coli* (100/mL) is given in the lower panel.

The temporal correlations in the timing of elevated levels of *E. coli* across the ten beaches over the period 1993 to 2003 is examined in Figures 3, 4, and 5. The number of samples sets where the geometric mean exceeds the PWQO across the ten beaches on a given day of sampling are shown relative to the number of samples for the day along with median value across the samples sets.

Collectively, the plots demonstrate several previously recognized features in the fluctuations in the levels of *E. coli* detected in beach sampling data. Firstly, levels vary widely over the sampling season with variable numbers of good days and bad days in any one year. Secondly, there have been a limited number of days when all, or most beaches, appear to have experienced poor conditions (defined as a sample set above 100 cfu/100mL). Finally, there are

many days when a small subset of the beaches has *E.coli* levels exceeding 100 cfu/100mL suggesting that factors affecting beach quality vary among locations.

Over the years 1993 to 2003, there have been 17 sampling dates identified as high effect days (Table 4). The criteria used to select these high effect dates was that at least seven sample sets were collected on a day and that 80% or more of the geometric means for a sample set exceeded 100 cfu/100mL. The number of sample sets (days) meeting these criteria in a sampling season was low ranging from none in 2003 and four in 2001.

In efforts to understand the processes leading to large scale *E. coli* problems, climatic and environmental variables on these 17 dates were examined. However, the data failed to establish any overarching correlations between climatic or environmental variables and high *E.coli* levels. However, some broad relationships could be identified. Recent heavy precipitation events coincided with seven of the 17 days as suggested by cumulative (two days prior plus day of sampling) precipitation (averaged across gauges at Lucknow, Benmiller and Varna), or a marked change in the rate of flow of the Maitland River at Benmiller (Table 4). Average wind speeds in excess of 10km/h at the Goderich Airport were measured on 13 of the 17 days, eight of which had no or light precipitation. Increased turbidity of the raw water at the Goderich Water Treatment Plant (WTP) on eight of the 13 windy days suggest sediment resuspension from the lake bottom. Wind direction on the 13 windy days was predominantly from the S and SW, although NW and W winds were also observed. There were four days which were neither wet nor obviously windy when geometric means in most of the sample sets were elevated, however, for three of these days the levels and fluctuations in turbidity were suggestive of disturbances in the nearshore. The only feature notable of the remaining day is that it had the highest water temperature (24°C) of any of these 17 high effect days (Table 4) as inferred from daily temperature measurements of raw water at the Goderich WTP. The correspondence in fluctuations in *E.coli* levels with environmental variables for the full dataset is explored in the following sections of this report.

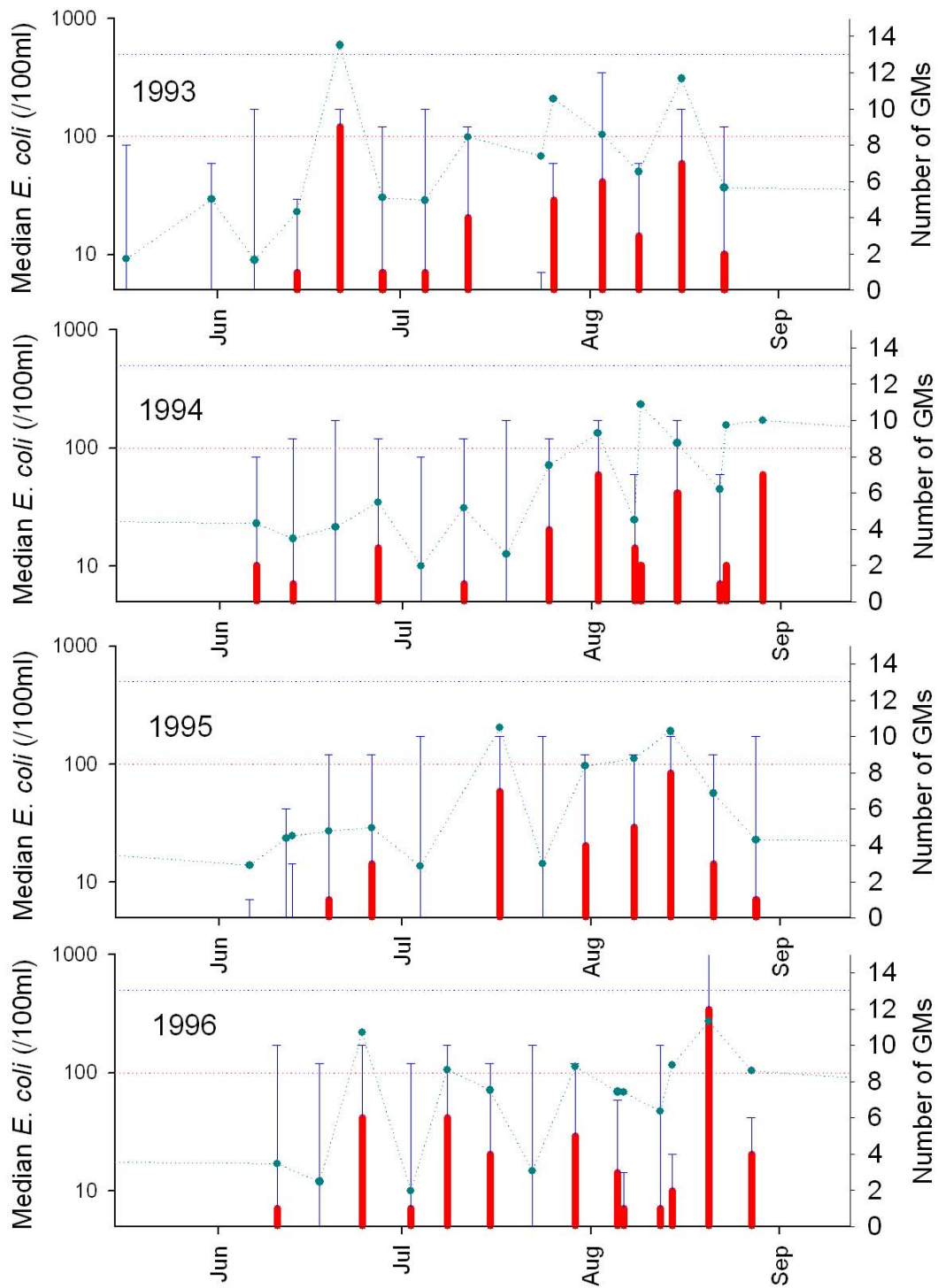


Figure 3: For the years 1993 to 1996, the median values of geometric means for sample sets collected on a day (blue circle) are shown with the number of sample sets where the geometric mean exceeded 100 cfu/100mL (red bar) and the number of sample sets for the day (blue bar).

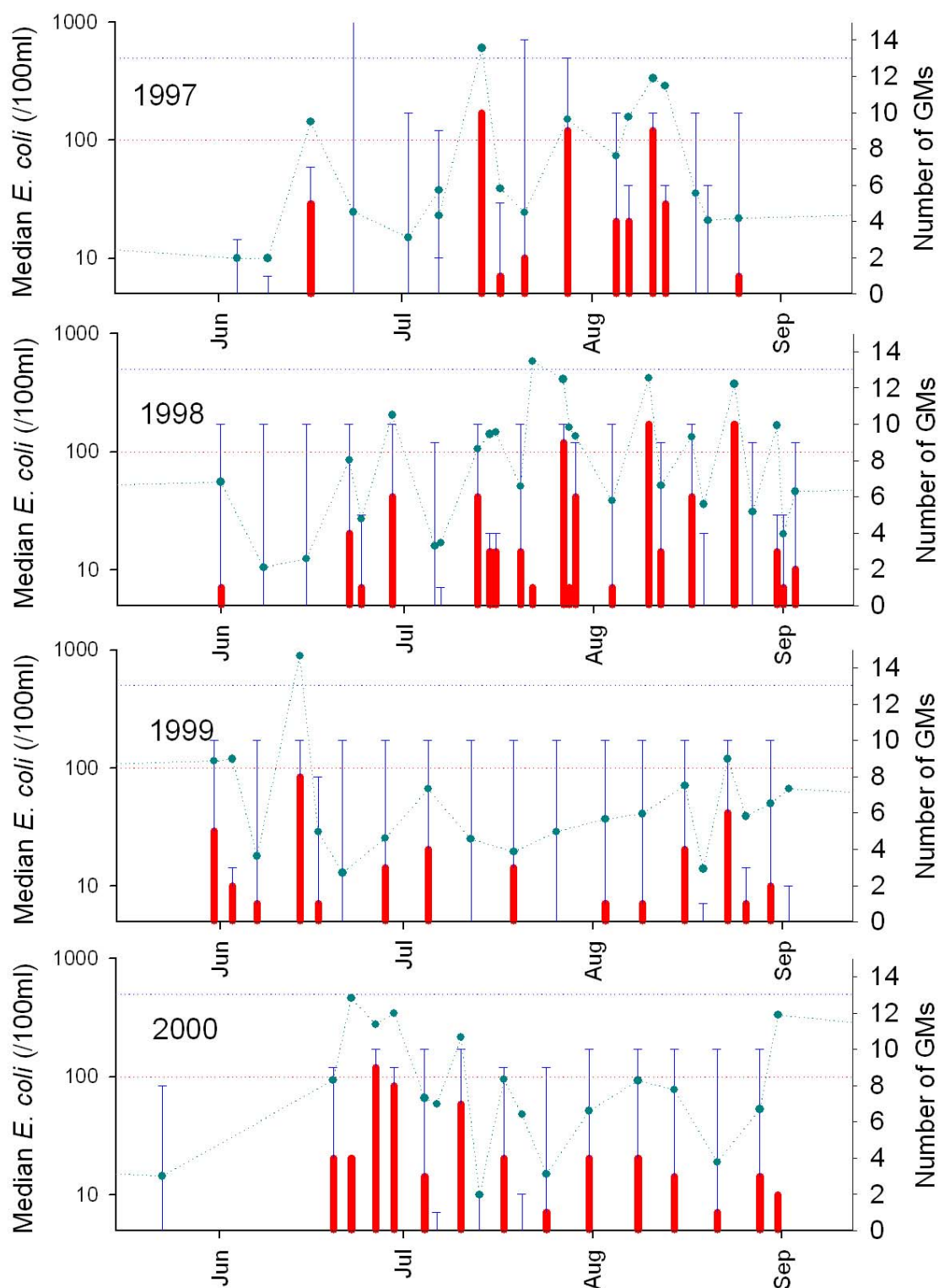


Figure 4: For the years 1997 to 2000, the median values of geometric means for sample sets collected on a day (blue circle) are shown with the number of sample sets where the geometric mean exceeded 100 cfu/100mL (red bar) and the number of sample sets for the day (blue bar).

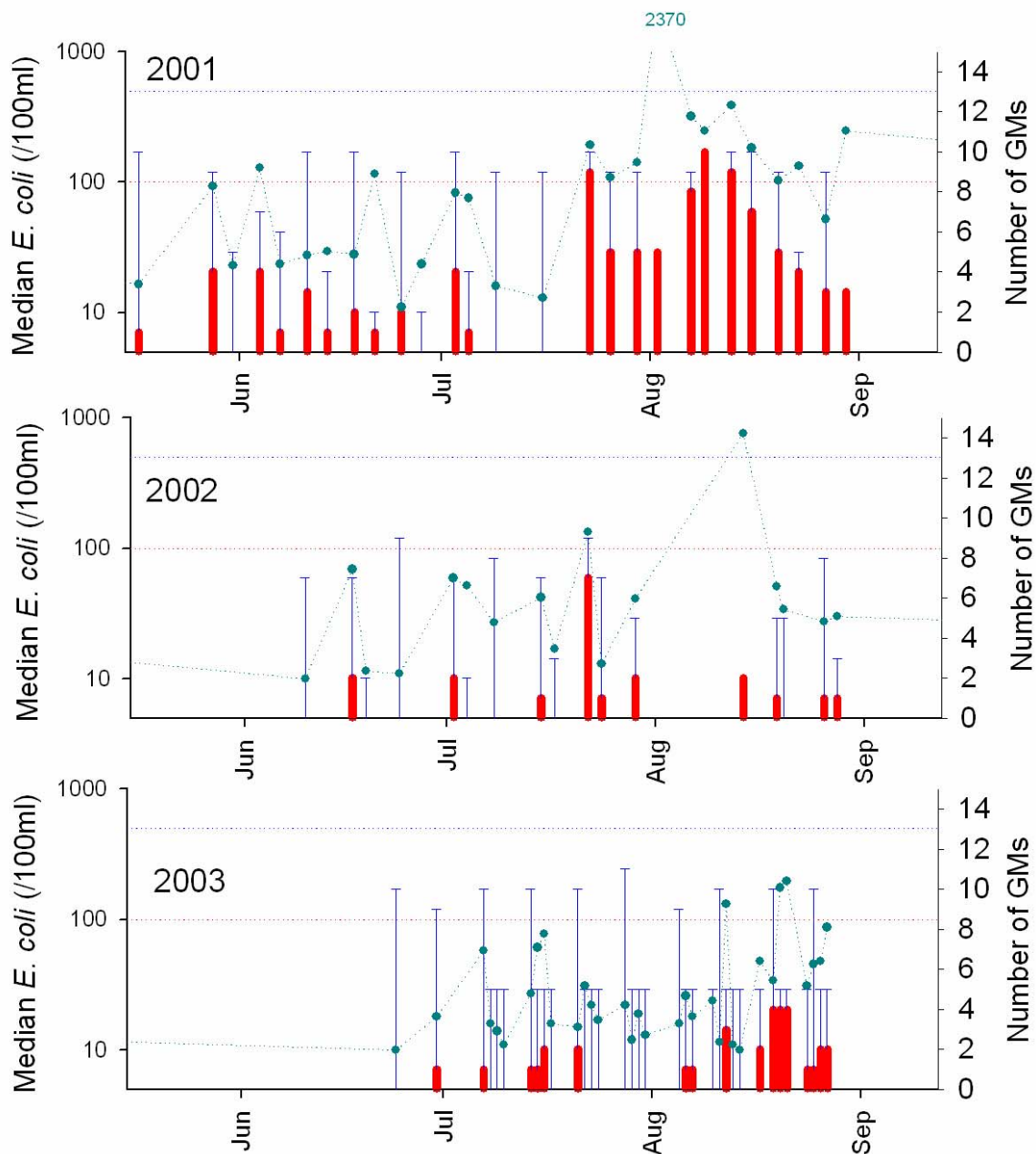


Figure 5: For the years 2001 to 2003, the median values of geometric means for sample sets collected on a day (blue circle) are shown with the number of sample sets where the geometric mean exceeded 100 cfu/100mL (red bar) and the number of sample sets for the day (blue bar).

Table 4: Listing of dates when geometric means exceeded 100 cfu/100mL in 80 % or more of the samples sets for the day. Only the days where seven or more samples sets were collected on a day were considered in the analysis.

Date Tested (#gms)	%GMs Above 100 cfu/100ml	Rain (mm)	Rain 3 Day (mm)	Flow Maitland River At Benmiller (m³/s)	Change in flow previous day (m³/s)	Wind Speed (km/h)	East Wind Vector	North Wind Vector	Turbidity (FTU)	Turbidity Change from previous day	Water Temperature (°C)	Water Temperature Difference from past day	Air Temperature (°C)	Water Level (cm)	Change Water Level from previous day (cm)
21-Jun-93 (10)	90	15	50	80.5	54.7	nd	nd	nd	2.8	0.3	15.5	2	nd	88	7
29-Aug-94(7)	100	0	3	4.6	-0.2	14.8	-7.5	9.1	14	-11	20.5	-0.5	15.5	87	1
14-Aug-95 (10)	80	9	36	93.3	-34.7	14.6	0	-13.4	10	4.3	20	0	24.3	62	0
20-Aug-96 (15)	73*	5	5	2.6	0	15.1	-3.0	-12.2	2	0.9	20.5	2.5	22.6	82	2
14-Jul-97 (10)	100	33	33	5.7	0.6	17.4	-4.0	-14.3	6	5.3	20	7	25.0	117	-1
11-Aug-97 (10)	90	2	2	2.6	0	9.1	2.4	8.2	6.8	2.8	20.5	-0.5	15.1	117	0
27-Jul-98 (10)	90	1	1	1.5	-0.1	20.1	-11.4	-14.7	15.3	13.9	22	-1	21.2	89	4
10-Aug-98 (10)	100	0	0	2.4	-0.1	8.7	-4.6	-1.5	1.3	-0.7	24	1	22.5	83	0
24-Aug-98 (10)	100	0	2	1	0	14.7	-8.9	-10.8	1	0.5	20	3	24.6	81	8
14-Jun-99 (10)	80	13	13	4.1	0.1	22.9	-6.4	10.7	0.6	-0.5	20	1	16.1	39	4
26-Jun-00 (10)	90	10	53	308	174	16.3	-0.6	-6.8	3.4	0.4	19	-1	19.5	12	-1
29-Jun-00 (9)	88.9	7	8	130	-95	12.2	-5.6	2.3	7.1	2.4	19	.	15.5	15	2
23-Jul-01 (10)	90	0	4	4.6	0	15.3	-4.9	-13.3	4.4	-3.1	21	1	25.9	5	1
07-Aug-01 (9)	88.9	0	0	2.0	0	9.4	-6.2	-0.5	5	-20.6	22	1.5	25.4	5	2
09-Aug-01 (10)	100	4	4	1.9	0	19.2	-7.7	-16.1	6	2.5	23	1	26.6	4	0
13-Aug-01 (10)	90	0	3	1.8	0	17.2	-3.5	12.2	5.7	-36.3	21	2	19.7	5	2
22-Jul-02 (9)	80 ⁺	11	13	5.2	0.1	19.4	-5.9	-17.8	2	-0.5	19	3	25.7	33	5

* 11/15 samples (73%) had greater than 100 E. coli /100ml;+ 7/9 samples (77.8%) had greater than 100 E. coli /100ml;rainfall data is average of measurements at Benmiller, Varna and Lucknow; Wind and air temperature data is for Goderich Airport; Water temperature and turbidity are for raw water at the Goderich Water Treatment Plant; Water level is for the Goderich Gauge.

2.4.2 Trends in Occurrence of *E.coli* at Individual Beaches

The annual medians of the geometric means for the 10 study beaches are presented in Figure 7 to 15, along with the annual percentages of the sample sets exceeding 100 cfu/100mL. The years in which >50% were and <30% of the sample sets were above the PWQO are tabulated in Table 5 for each beach.

The frequency with which the geometric means exceed the PWQO varies appreciably between beaches. Four beaches were observed with four to five years with >50% of the samples sets above the PWQO and, conversely, only three beaches with geometric means below the PWQO in >70% of samples in all consecutive years (with the exception of one year) were identified. Amberley Beach, Ashfield Township Park Beach, Port Albert Beach and Goderich Main Beach are at the poor end of spectrum and Sunset Beach, Goderich Cove Beach, Bayfield Main and Bayfield South Beach at the good end of the spectrum.

The years 2000 and 2001 stand out as poor years, followed by 1998 and 1993 based on the number of beaches with >50% of samples exceeding the PWQO during the year. The years 2002 and 2003 were relatively good years at all beaches with samples sets exceeding the PWQO in less than 30% of the sample sets at all beaches.

Table 5: Years in which geometric means exceed the PWQO in >50% and <30% of sample sets at selected beaches between 1993 and 2003.

<i>Beach</i>	<i>Years => 50%</i>	<i>Years<=30%</i>
Amberley	94,98,00,01	02,03
Ashfield Township Park	94,98,00,01	02,03
Port Albert	96,97,98,00,01	93,02,03
Sunset Beach	no year	all years
Goderich Main Beach	93,95,96,99,01	02,03
Goderich St.Christophers	93,96	95,97,99,00,02
Goderich The Cove	no year	all but 94,98
Blacks Point	93,00,01	94,02,03
Bayfield Main	no year	all years
Bayfield South	no year	all except 97

At Amberley Beach the years 1994 and 2000 stand out as years with a high frequency of geometric means exceeding the PWQO, however, the low numbers of sample sets in 1994 (n=5 samples sets) limits the robustness of the observation (Figure 6). In general the geometric means for many of the samples sets exceeded the PWQO, with the possible exception of 2002 and 2003 years, suggesting chronic sources of loading fecal pollutants to this particular beach.

There has been no obvious trend in *E. coli* levels over the years, however, there have been variable levels of sampling efforts over the years and it is possible that the annual results are dependent on sampling effort.

At Amberley Beach, there does not appear to be any clear intra-annual pattern in variability in levels of *E. coli* over the limited sampling period. The only observation that can be made is that elevated levels of *E. coli* may be observed at any time over the sampling season, but they appear to occur most frequently in August followed by June and least frequently in early July (Figure 6).

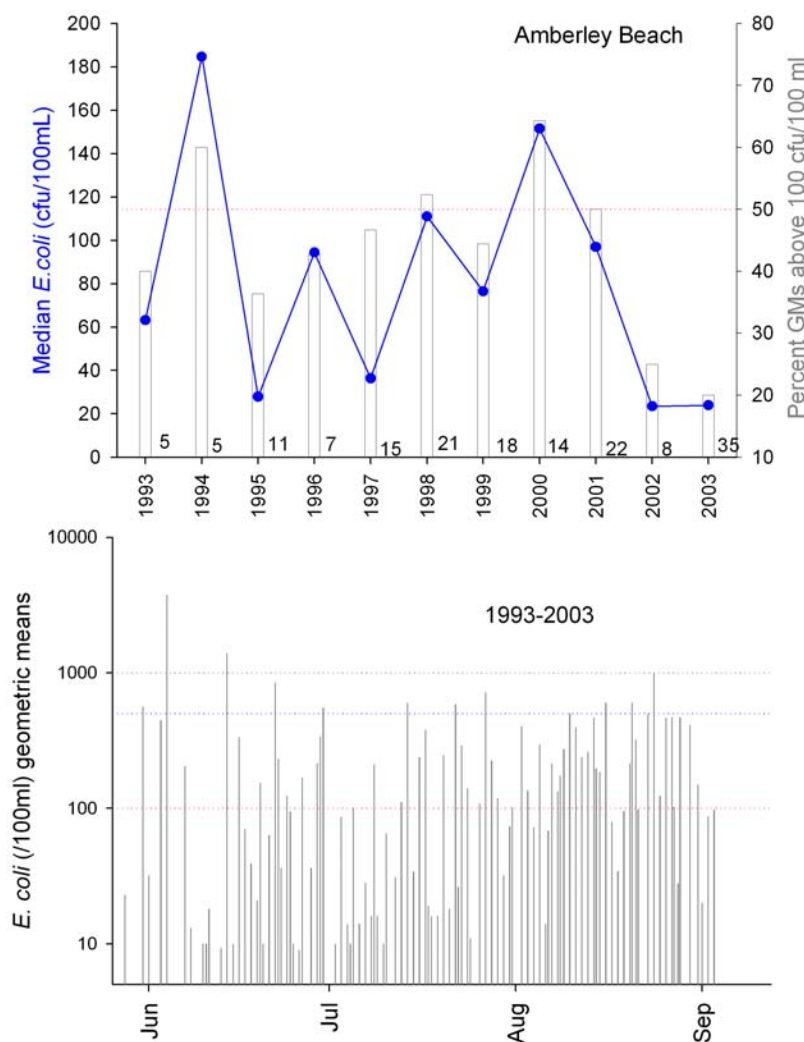


Figure 6: The median value of geometric means for sample sets collected at Amberley beach for the years 1993 to 2003 (blue circle) and percentage of sample sets where the geometric mean was above 100 cfu/100ml (bars) are shown in the upper panel. The number of sample sets in a year is shown by the numbers at the bottom of the bars. The lower panel shows the distribution of geometric means over the sampling season as a composite of all years.

The frequency of sample sets exceeding the PWQO at Ashfield Township Park was appreciable in most years, with the exception again of 2002 and 2003 (Figure 7). There were similarities in annual variability in levels of *E.coli* at Ashfield Township Park with Amberley Beach only a short distance to the north, however, the range in geometric means among sample sets appears to be higher at Ashfield Township Park. Comparable to Amberley Beach, >50% of geometric means at Ashfield Township Park Beach exceeded the PWQO in the years 2000 and 1994 and, in addition, 2001 was notable with a high frequency of sample sets exceeding the PWQO.

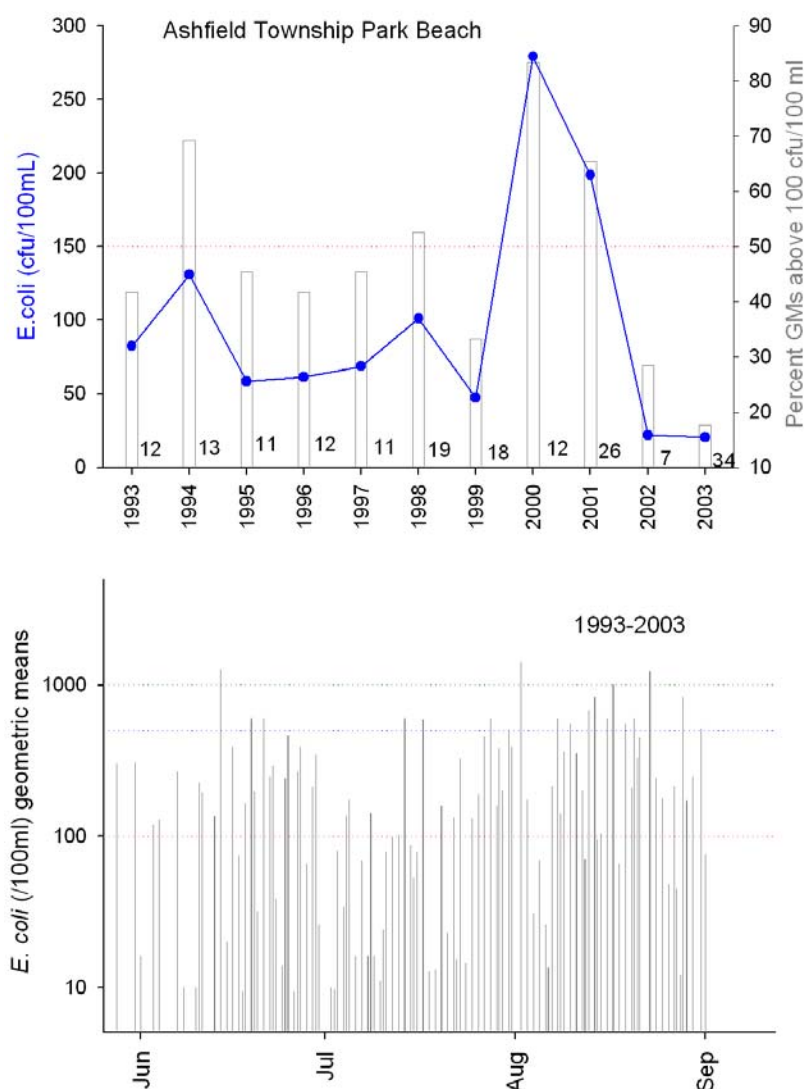


Figure 7: The median value of geometric means for sample sets collected at Ashfield Township Park beach for the years 1993 to 2003 (blue circle) and percentage of sample sets where the geometric mean was above 100 cfu/100ml (bars) are shown in the upper panel. The number of sample sets in a year is shown by the numbers at the bottom of the bars. The lower panel shows the distribution of geometric means over the sampling season as a composite of all years.

Port Albert beach is another example of a beach experiencing chronic problems in levels of *E. coli*. As evident in Figure 8, in a number of years the majority of geometric means for sample sets exceeded the PWQO. The year 1998 stands out more as a poor year at this beach and 1994 less so than at Amberley and Ashfield Township Park Beaches. The flip-flop in conditions between 1998 and 1999 is perplexing. The results for the years 1993 to 1998 are suggestive of an increasing trend but this pattern disappeared in the subsequent years (Figure 8). However, as observed in Amberley and Ashfield Township Park Beaches, the 2002 and 2003 years were relatively good years with <30% of the sample sets' geometric means exceeding the PWQO. No seasonal trends could be identified at Port Albert Beach.

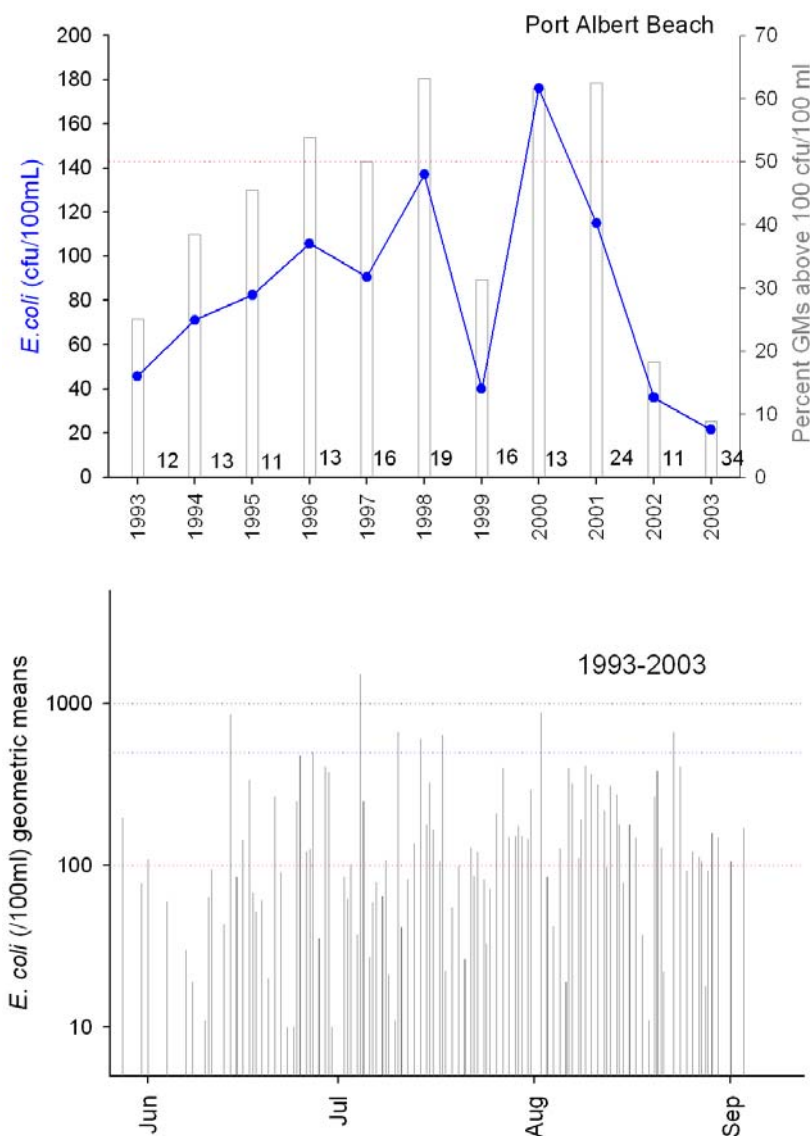


Figure 8: The median value of geometric means for sample sets collected at Port Albert beach for the years 1993 to 2003 (blue circle) and percentage of sample sets where the geometric mean was above 100 cfu/100ml (bars) are shown in the upper panel. The number of sample sets in a year is shown by the numbers at the bottom of the bars. The lower panel shows the distribution of geometric means over the sampling season as a composite of all years.

Sunset Beach is representative of a number of recreational beaches in Huron County where beach conditions appear to be less affected by fecal pollution (i.e. levels of *E. coli* determined in the beach monitoring program). Only in 2001 did the geometric means of approximately 50% of sample sets exceeded the PWQO at Sunset Beach. In other years the frequency of sample sets exceeding the PWQO was < 30% and in the years 1995, 2002 and 2003, no samples sets exceeded the PWQO (Figure 9). Once again, the years 2002 and 2003 were relatively good years even for this low impact beach site.

On a seasonal basis, elevated levels of *E.coli* could be encountered at any time throughout the monitoring season at Sunset Beach, however, more frequent incidents of elevated levels tended to occur later in the summer (Figure 9).

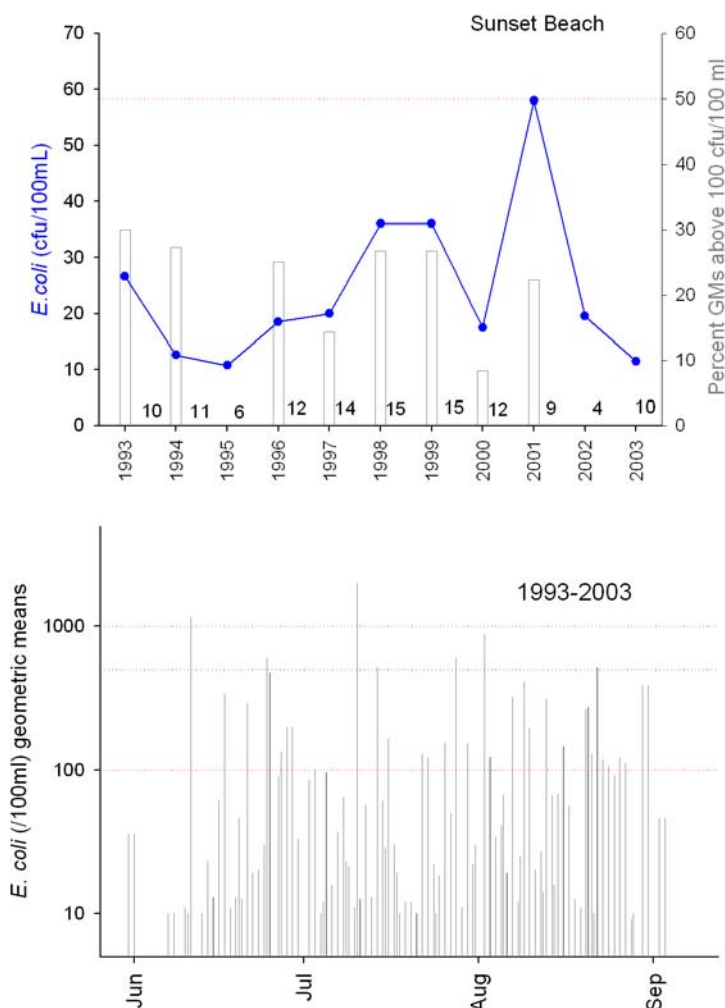


Figure 9: The median value of geometric means for sample sets collected at Sunset beach for the years 1993 to 2003 (blue circle) and percentage of sample sets where the geometric mean was above 100 cfu/100ml (bars) are shown in the upper panel. The number of sample sets in a year is shown by the numbers at the bottom of the bars. The lower panel shows the distribution of geometric means over the sampling season as a composite of all years.

Goderich Main Beach, immediately south of Goderich Harbour, has the most frequent incidence of elevated levels of *E.coli* of all three beaches on the Goderich waterfront. In five of the 11 study years more than 50% of sample sets from this beach have exceeded the PWQO (Figure 10). Unlike other beaches, the trend in levels of *E.coli*, over years is suggestive of a decline in levels over the years, however, this observation is speculative given the considerable

interannual variability. The worst years appear to have been 1993, 1995 and 1996, followed by 1999 and 2001 while the best years were again 2002 and 2003.

On a seasonal basis, there appears to be a higher frequency of PWQO exceedances in August than earlier in the summer, however, elevated levels of *E. coli* can be detected throughout the monitoring season (Figure 10).

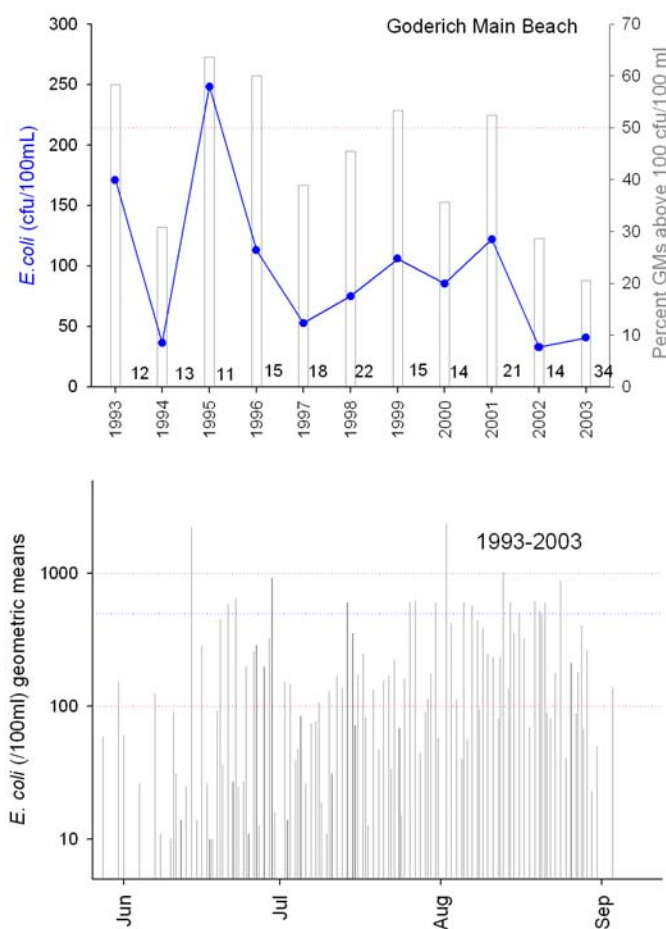


Figure 10: The median value of geometric means for sample sets collected at Goderich Main beach for the years 1993 to 2003 (blue circle) and percentage of sample sets where the geometric mean was above 100 cfu/100ml (bars) are shown in the upper panel. The number of sample sets in a year is shown by the numbers at the bottom of the bars. The lower panel shows the distribution of geometric means over the sampling season as a composite of all years.

The levels of *E. coli* at St. Christophers Beach, a short distance south of the Main beach in Goderich, are noticeably lower than at the Main Beach but not without some problematic years (Figure 11). The years 1993 and 1996 were the poorest years. As with the Main Beach data, the variability over the years hints at a decline in levels over years with the years 2002 and 2003

among the lowest. Over the monitoring seasons, the frequency of elevated results appears to be highest in late summer.

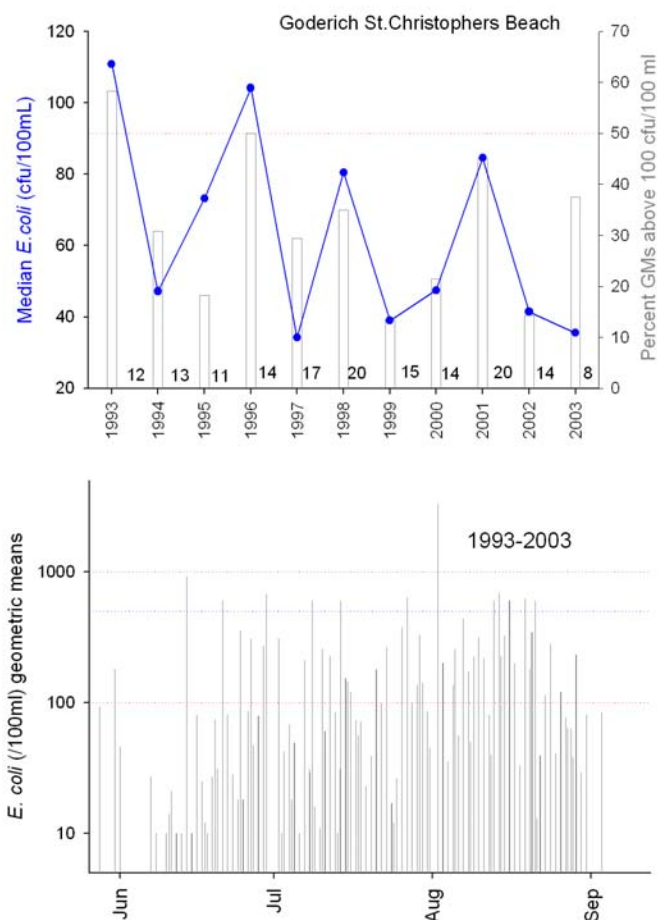


Figure 11: The median value of geometric means for sample sets collected at Goderich St. Christophers Beach for the years 1993 to 2003 (blue circle) and percentage of sample sets where the geometric mean was above 100 cfu/100ml (bars) are shown in the upper panel. The number of sample sets in a year is shown by the numbers at the bottom of the bars. The lower panel shows the distribution of geometric means over the sampling season as a composite of all years.

Goderich Cove Beach is the most southerly of the Goderich beaches and appears to have the lowest levels of *E. coli* (Figure 12). The pattern in variability among years in levels of *E. coli* is distinct from the other Goderich beaches. The years 1993 and 1996 do not stand out as poor years as in the other Goderich Beaches. However, the years 2002 and 2003 are again among the lowest at this site. The number of sample sets exceeding 100 cfu/100mL was less than or equal to 30% in all years except 1994 and 1998.

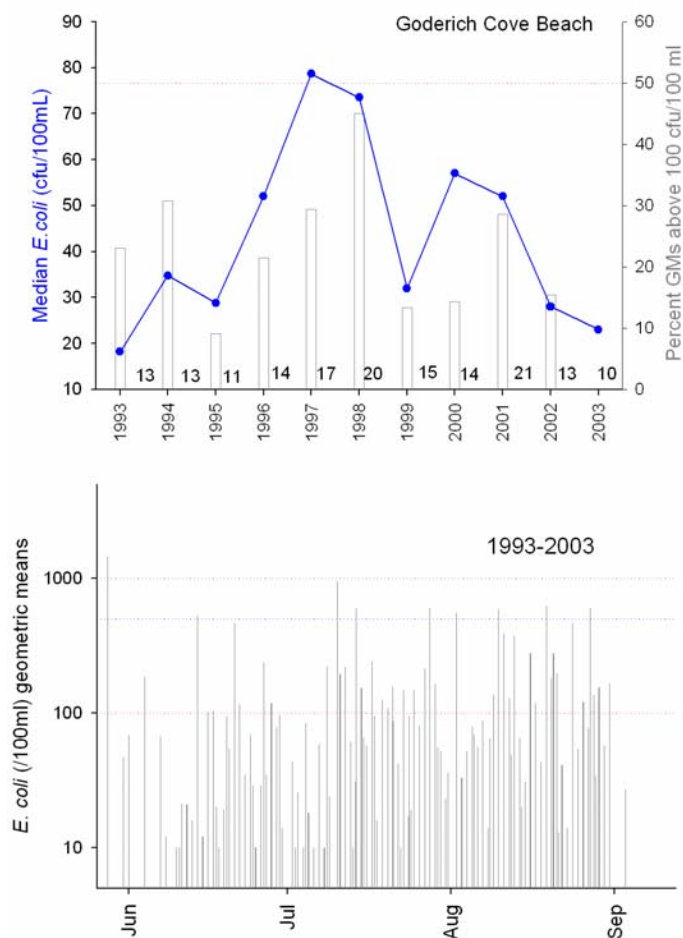


Figure 12: The median value of geometric means for sample sets collected at Goderich Cove beach for the years 1993 to 2003 (blue circle) and percentage of sample sets where the geometric mean was above 100 cfu/100ml (bars) are shown in the upper panel. The number of sample sets in a year is shown by the numbers at the bottom of the bars. The lower panel shows the distribution of geometric means over the sampling season as a composite of all years.

The trends in *E. coli* levels over the years at Blacks Point are intriguing with widely varying numbers of sample sets exceeding the guideline among years (Figure 13). The years 2000 to 2003 highlight this contrast. Greater than 70% of samples sets exceeded the guideline in 2000 and 2001 compared to the years 2002 and 2003 where approximately 20% of sample sets exceeded 100 cfu/100mL. From 1994 to 1999, 27 to 41% of annual samples sets were above the PWQO.

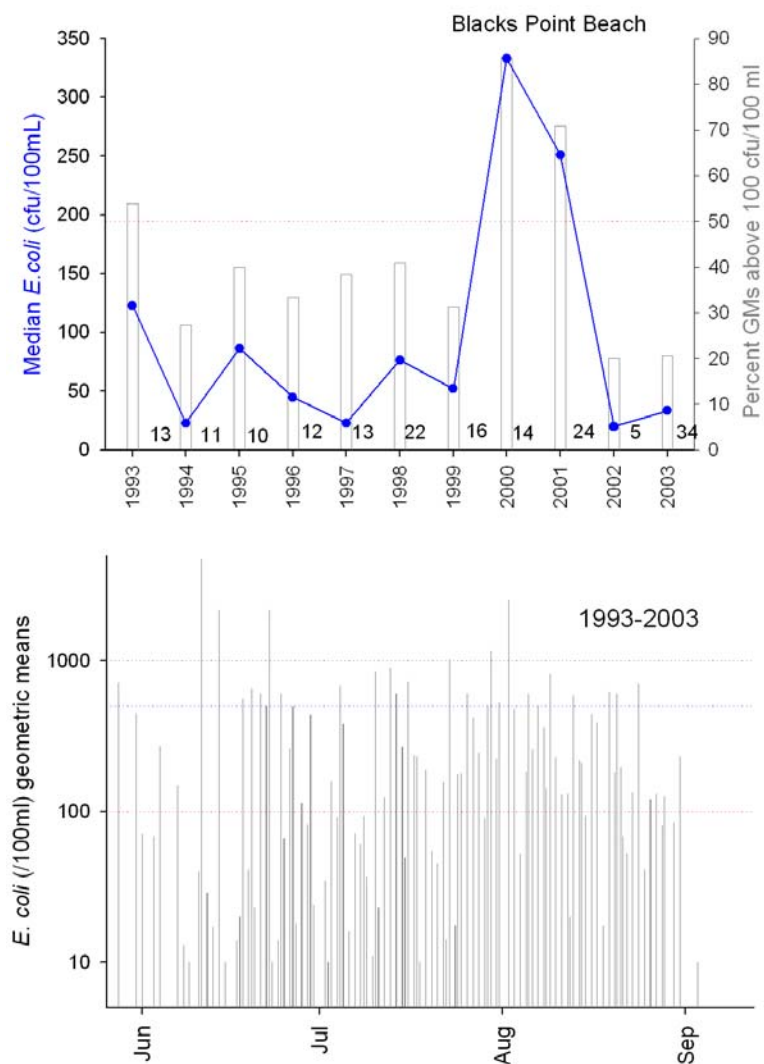


Figure 13: The median value of geometric means for sample sets collected at Blacks Point beach for the years 1993 to 2003 (blue circle) and percentage of sample sets where the geometric mean was above 100 cfu/100ml (bars) are shown in the upper panel. The number of sample sets in a year is shown by the numbers at the bottom of the bars. The lower panel shows the distribution of geometric means over the sampling season as a composite of all years.

The southern most beaches examined in detail here are located along the shores of the community of Bayfield, adjacent and to the south of the mouth of the Bayfield River. Relative to some of the other beaches examined, *E. coli* levels are comparatively low (Figure 14 and 15). While in many years, levels for individual sample sets have exceeded 100 cfu/100mL, the percentage of samples sets exceeding this level was less than or equal to 30% for all years with the exception of 1997 at Bayfield South Beach (Figure 15). In 2003, geometric means did not exceed 100 cfu/100mL in any samples sets. In contrast to the other beaches, the years 2002 and 2003 are not easily classified as relatively good years for these two Bayfield beaches. This is perhaps a result of the relatively low background levels of *E. coli* throughout the years.

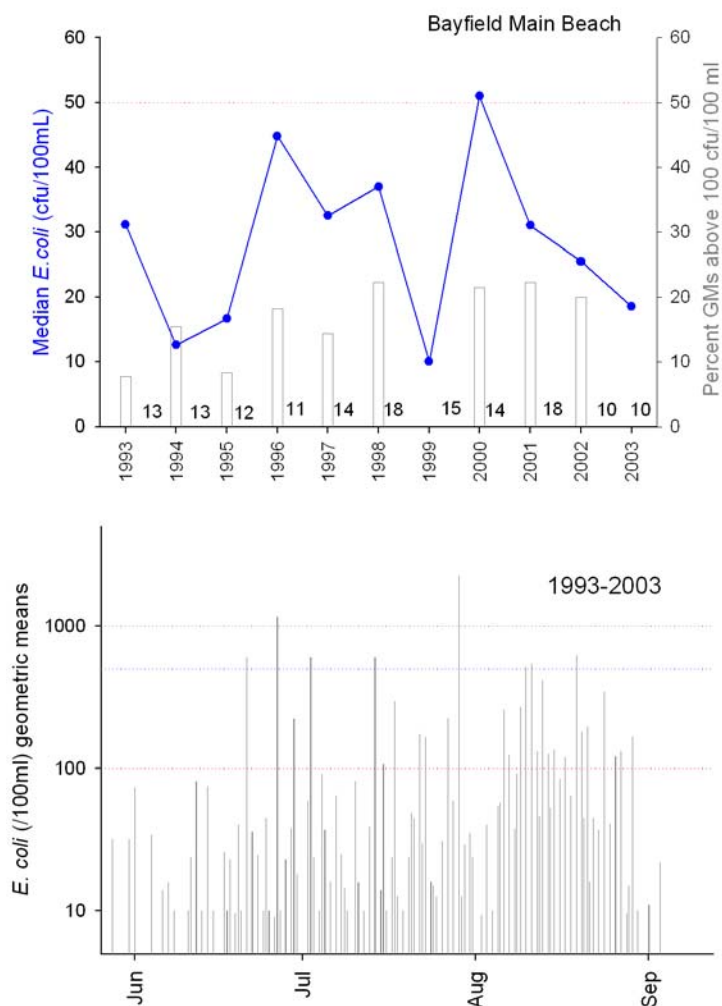


Figure 14: The median value of geometric means for sample sets collected at Bayfield Main Beach for the years 1993 to 2003 (blue circle) and percentage of sample sets where the geometric mean was above 100 cfu/100ml (bars) are shown in the upper panel. The number of sample sets in a year is shown by the numbers at the bottom of the bars. The lower panel shows the distribution of geometric means over the sampling season as a composite of all years.

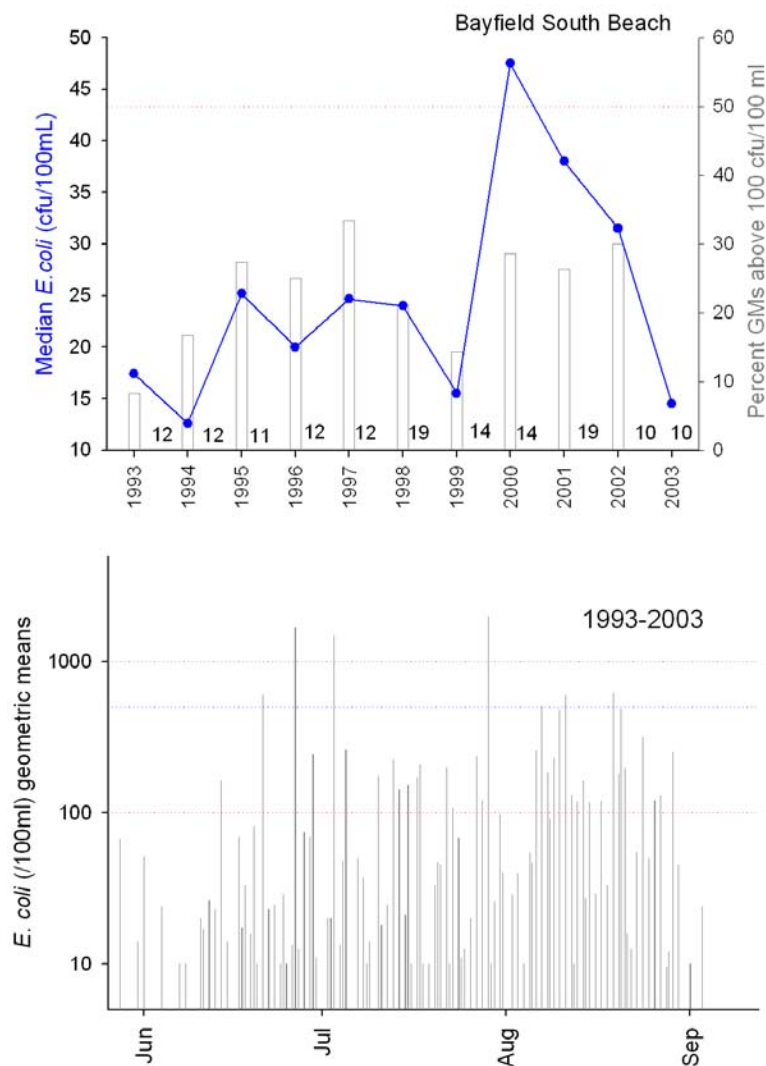


Figure 15: The median value of geometric means for sample sets collected at Bayfield South Beach for the years 1993 to 2003 (blue circle) and percentage of sample sets where the geometric mean was above 100 cfu/100ml (bars) are shown in the upper panel. The number of sample sets in a year is shown by the numbers at the bottom of the bars. The lower panel shows the distribution of geometric means over the sampling season as a composite of all years.

The range and the fluctuation in levels of *E. coli*, at the 10 beaches examined strongly suggest that the factors contributing to the observed *E. coli* levels vary between beaches. Whether the differences are due to variations in the sources or magnitude of the *E. coli* loading, due to the beach features that contribute to *E. coli* persistence or a combination of factors remains unclear.

2.4.3 Patterns of Occurrence in *E. coli* in Relation to Shoreline Tributary Discharge

A route by which microbial pollutants, which have been released into the environment, are delivered to shoreline beaches is through transport by tributaries which discharge to the shores of the lake. The loading of *E. coli* from the land areas draining to the shoreline of the lake likely increases under wet weather conditions and in wet years when tributary discharge rates are higher and larger volumes of water are being delivered to the lake.

Information collected at a flow gauging station on the lower Maitland River (Benmiller Gauge 02FE015 – Environment Canada) was used to explore the possible relationship between the volume of water being loaded to the lake and the observed variability in levels of *E. coli* at the recreational beaches. Over the study area, downstream gauging stations in relative proximity to the lake are found on both the Maitland and the Bayfield Rivers. Downstream flow information for the Maitland River, the larger of the two, was selected to represent interannual trends in relative delivery of water to the lake. The mean spring and summer daily flows and 90th percentile for the years 1993 to 2003 are presented in Figure 16. The spring and summer flows are presented separately because of the possibility that beach conditions during the bathing season may be responsive to either immediate conditions (summer flows) or conditions prior to the bathing season (spring flows). Indicators of high flow events (Figure 16) are also presented to explore the possibility that the magnitude or frequency of high flow events may be related to interannual fluctuations in *E. coli* levels shoreline beaches.

There was appreciable variability in the flow rates of the Maitland River, and presumably other nearby tributaries in general, over the years 1993 to 2003 (Figure 16). The relatively low flows during the summers of 1998 and 1999 and high flow of 2000 indicate a contrast between dry and wet years, respectively. The year 1999 further stands out as year with particularly low spring flows.

The years of high or low flows in the lower Maitland River show little correspondence with the annual fluctuations in levels of *E. coli* at the ten recreational beaches. The relatively wet summer of 2000 was a relatively poor year at several beaches in terms of *E. coli* levels, however, low summer flows were also observed in another poor *E. coli* year (2001). In contrast, the spring flows for 2000 and 2001 were similar in magnitude and comparable to those measured in most other years. Annual median concentrations of *E. coli* for individual beaches

are plotted against flow features of the lower Maitland River for the summer period in Figure 17. There appears to be little relationship between average flow, the 90th percentile or the number of days of flow $>40 \text{ m}^3/\text{s}$ with annual median levels of *E. coli*. However, the ranges of median values among beaches for the year of highest flow (2000) is wider than other years. Similar analysis using the spring data (not shown) failed to identify relationships between features of flow volume and annual median levels of *E. coli*.

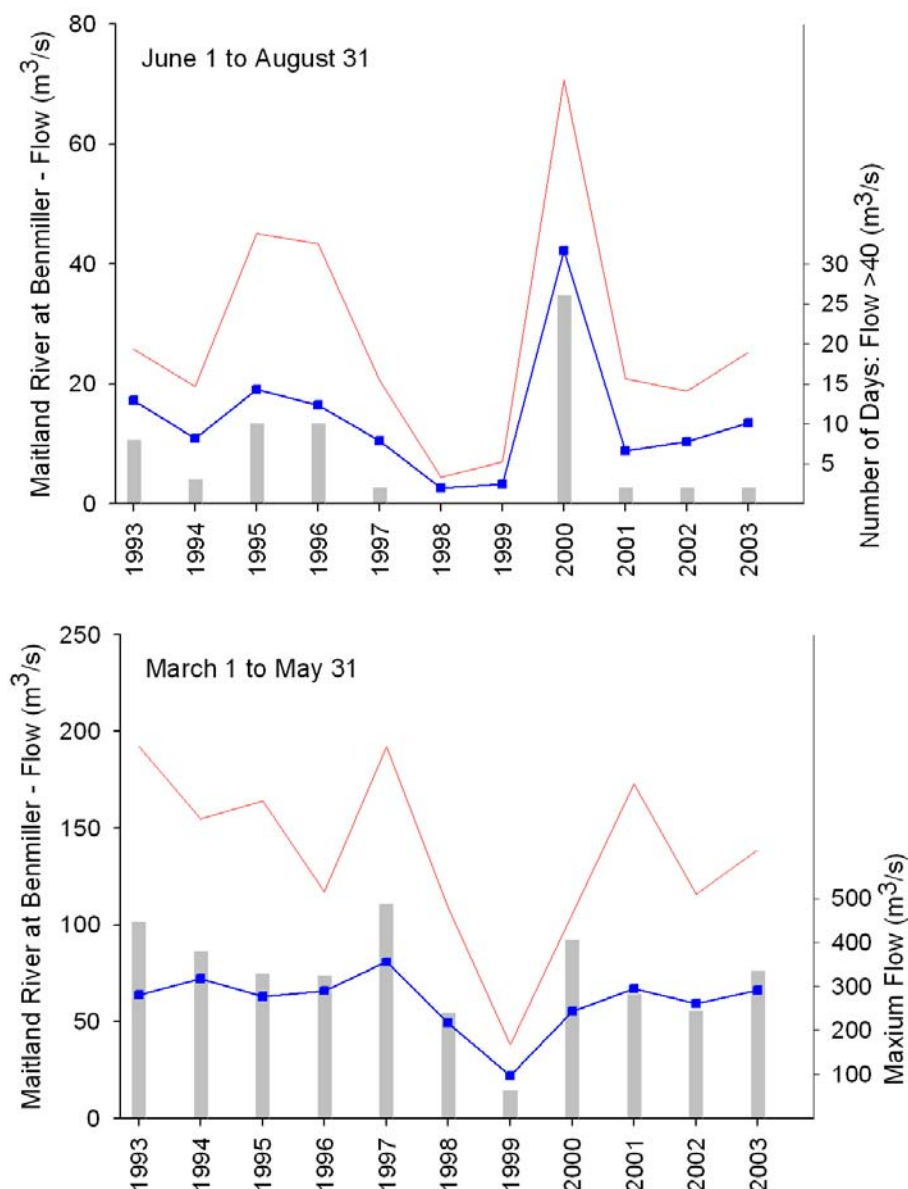


Figure 16: Mean of daily average flow in the Maitland River at the Benmiller Gauge over the summer and late spring periods. The red lines present the 90th percentile and the blue line are the mean flows. The gray bars present the number of days where flow exceeded $40 \text{ m}^3/\text{s}$ in the top panel and maximum daily flow in the lower panel.

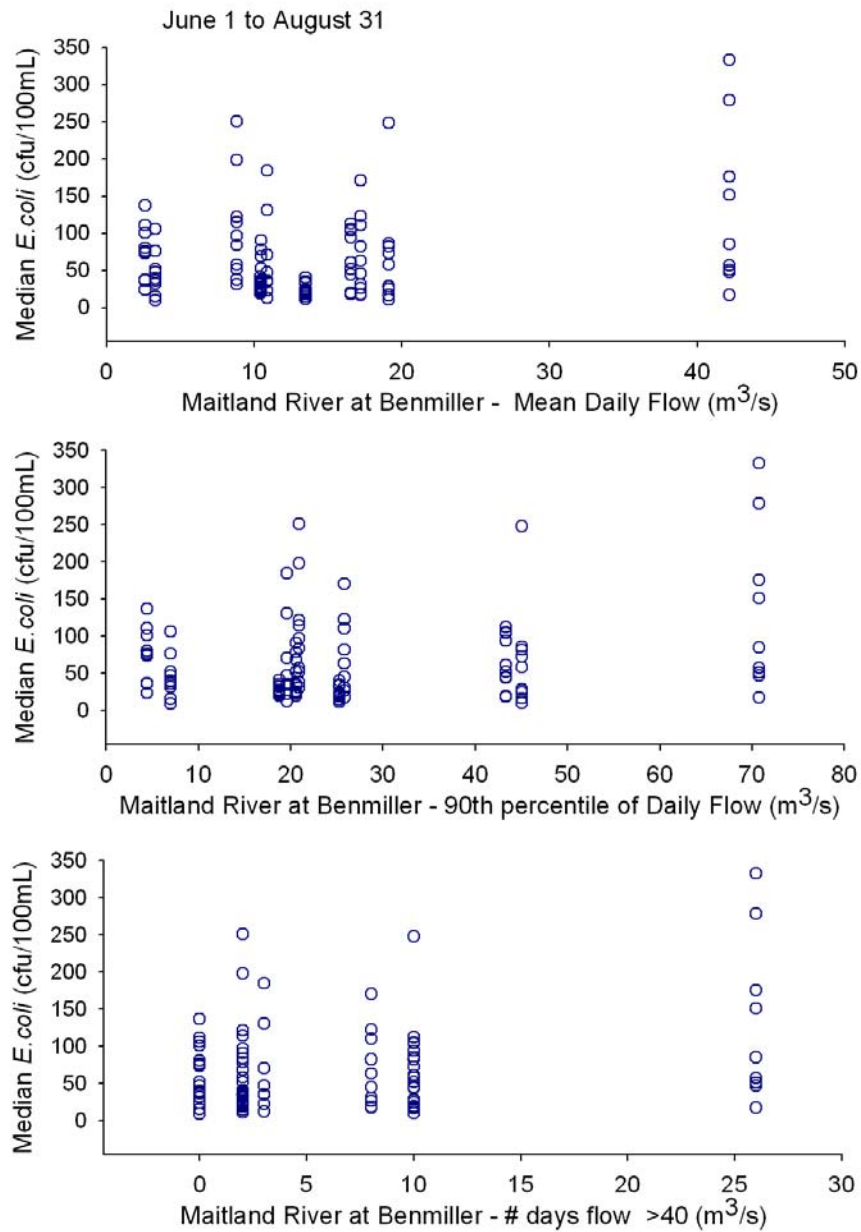


Figure 17: Annual median *E. coli* (cfu/100mL) over the geometric means at individual beaches in relation to mean flow, 90th percentile and number of days with flows >40 m^3/s over the June 1 to August 31 period for the Maitland River at the Benmiller gauging station.

The quality of the water in a tributary is a key factor in determining the impact of its discharge to the lake on environmental quality at the shoreline. Typically, the quality of water in tributaries is variable, responding to long term (seasonal) and short term (wet weather events) alterations in flow. Information on tributary flow alone is unlikely to capture the temporal variability in the effects that a tributary may be having on an area of shoreline to which it discharges.

Regrettably, there is limited *E. coli* monitoring data for the downstream reaches of water courses to Lake Huron over the study shoreline. While there are several sources of tributary data for the period 2001 to 2003, a consistent source of through-time information for *E. coli* for the period 1993 to 2003 appears to be lacking. Consequently, it is not possible to associate temporal variability in levels of *E. coli* at recreational beaches with variations in tributary quality. The limited tributary data for 2001 to 2003 is discussed in subsequent sections of the report only in relation to the studies in which these data were collected.

2.4.4 Patterns of Occurrence in *E. coli* in Relation to Precipitation

Elevated levels of *E. coli* are frequently reported after periods of precipitation. The intensity and magnitude of the wet weather events are thought to be a factor determining the degree to which bacteria are washed into watercourses and ultimately delivered to the lake shoreline. Rising flow rates in tributaries, a typical response to precipitation, are likely contributing to greater movement of bacteria within the tributary and larger mixing zones in the lake. Of interest, are agricultural drains and small creeks which may be especially responsive to wet weather events, flowing only at certain times of the year (e.g. spring snow melt) or in direct response to wet weather.

Precipitation data collected at gauging stations at Lucknow, Benmiller and Varna maintained by the MVCA and ABCA were used to examine the correspondence between patterns of precipitation and times when levels of *E. coli* were elevated. Annual precipitation for the spring and summer periods is presented in Figure 18. The average of the daily precipitation data for three gauging stations spread over the lands adjacent to the 10 beaches was used to represent precipitation levels over the shoreline. The range in cumulative precipitation over the summer and late spring periods is appreciable among years with the summers of 1998 and 2000 being the driest and wettest, respectively, and the springs of 2000 and 2001 being the wettest and driest, respectively. The histograms of daily precipitation for the spring and summer periods

illustrate that the extremes of weather, as evidenced by high values, are more evident in some years than others (Figure 18). For example, the springs of 1996 and 2000 stand out as years of very heavy precipitation events, in contrast to the spring of 2001 when daily precipitation averaged across the three stations never exceeded 15 mm/day (Figure 18).

Cumulative precipitation over the spring and summer periods for the years 1993 to 2003 are plotted against median *E. coli* for the study beaches in Figure 19. There appears to be no overall trend among years of higher median *E. coli* as cumulative precipitation increases. However, the spring and summer period of the wettest year (2000) also showed the largest range of *E. coli* median values.

The number of days with accumulative precipitation exceeding 10 mm was tabulated over the summer periods for each year, as a measure frequency of wet weather events among years. This was calculated with the hypothesis that more frequent rain storm events in a year might contribute to higher overall levels of *E. coli* in the beach monitoring program. Wet weather events as inferred from the 10 mm/day criterion ranged from 1 to 11 times per year but the number of events did not appear to be correlated with median levels of *E. coli* (Figure 19).

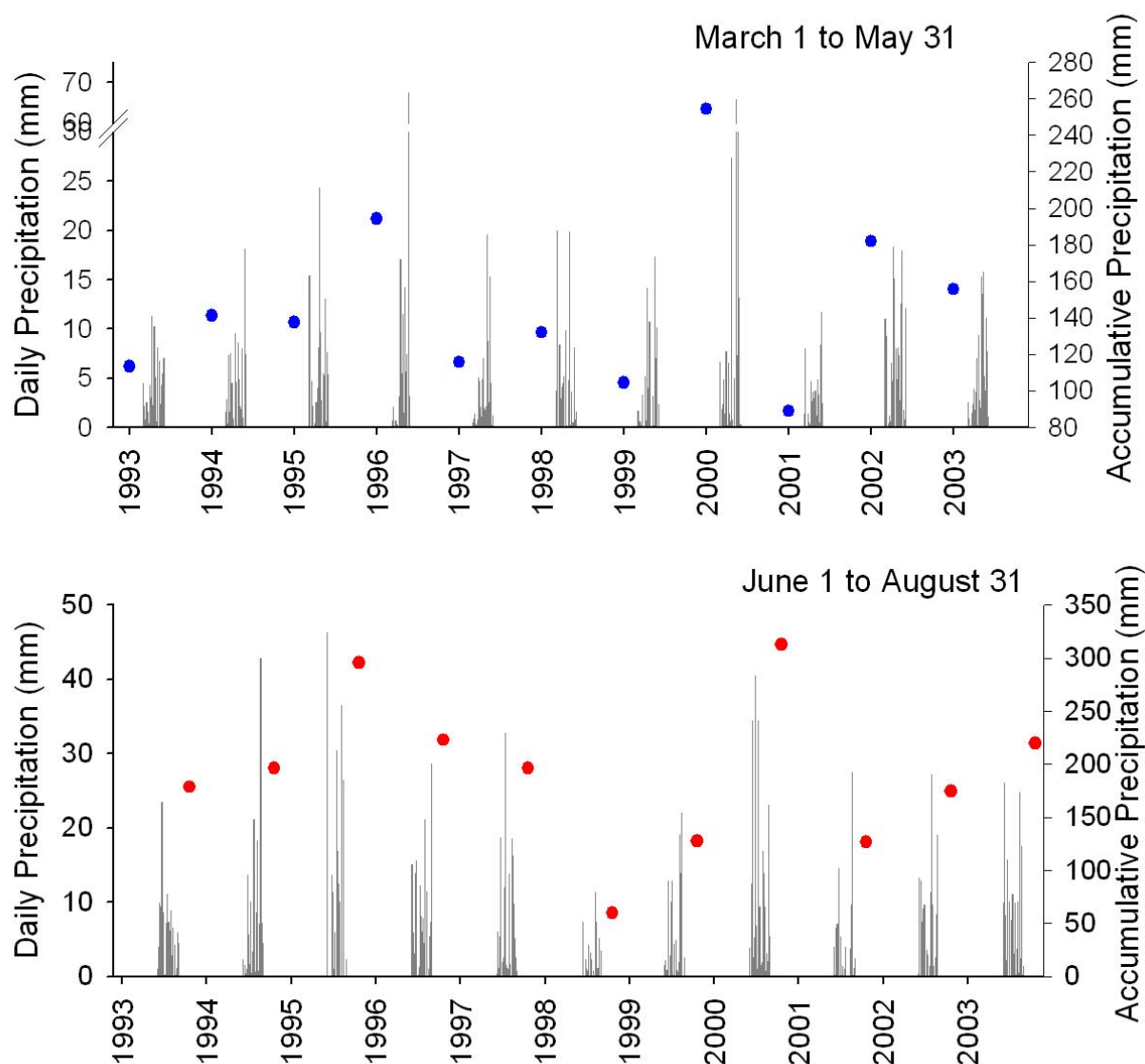


Figure 18: Accumulate daily precipitation as the average of measurements at the Lucknow, Benmiller and Varna gauges for the March 1 to May 31 (top panel) and June 1 to August 31 (bottom panel) periods are given by the bars. The circles give the accumulative rainfall for the periods as an average of the three datasets.

The geometric means of *E. coli* in individual sample sets for selected beaches is plotted against accumulative precipitation for the day preceding sampling and the day of sampling in Figures 20 to 22. Again, daily average precipitation for the Lucknow, Benmiller and Varna gauges was used to represent cumulative rainfall. Figures 20, 21 and 22 organize the data into beaches directly adjacent to the mouths of tributaries, those most distant from the mouths of tributaries and the three Goderich beaches, respectively.

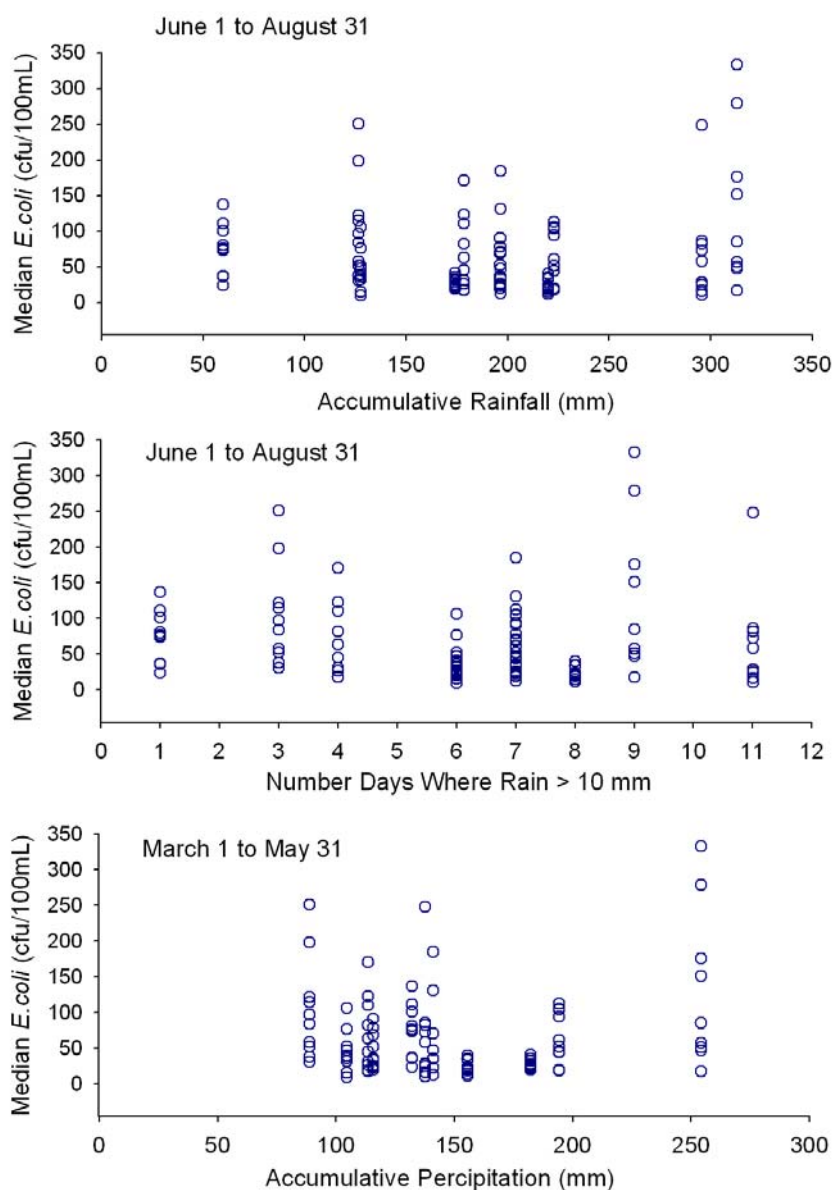


Figure 19: Annual median *E. coli* at beaches is plotted against accumulative precipitation as the average of measurements at the Lucknow, Benmiller and Varna gauges for the summer and spring period in the top and bottom panels, respectively. Median *E. coli* is plotted against number of days when daily precipitation exceeded 10 mm over the June 1 to August 31 periods in the middle panel.

There is little evidence of a consistent positive relationship between rainfall and levels of *E. coli* at the beaches directly adjacent to the small to moderate size tributaries (Figure 20). At Bayfield Main Beach, elevated levels of *E. coli* commonly coincided with heavy rainfall events. In cases where the geometric means exceeded 100 cfu/100ml there had been some precipitation the day of sampling or the previous day, however, *E. coli* was not necessarily elevated when precipitation events had recently occurred. The results for Port Albert Beach,

while suggesting some of the same features as Bayfield Main Beach, differ in that elevated *E. coli* levels (> 100 cfu/100 mL) often followed or occurred on dry weather days. Overall, levels of *E. coli* were frequently higher at Port Albert Beach than Bayfield Main beach. Results for Ashfield Township Beach are in contrast to the previous results in that rainfall as represented here appears to have little relationship to levels of *E. coli* (Figure 20).

At the three beaches spatially removed from tributaries, there appeared to be little consistent relationship between rainfall, as measured here, and levels of *E. coli* (Figure 21). At Blacks Point and Amberley Beach many of the times when elevated levels of *E. coli* were observed, there appeared to have been no rainfall on the previous or day of sampling. Yet most instances of accumulative rainfall >20 mm were coincident with *E. coli* levels above the PWQO. At Sunset Beach, *E. coli* levels were typically lower than at Amberley or Blacks Point beaches. In most cases, Sunset Beach levels exceeded 100 cfu/100mL when there had been at least some precipitation, although, on many occasions, precipitation, even appreciable precipitation, did not result in elevated (>100 cfu/100mL) *E. coli* levels.

The correlation between rainfall and levels of *E. coli* at the three Goderich beaches is similar among the beaches and again does not suggest a systematic relationship with rainfall (Figure 22). The strongest association with rainfall is suggested at Cove Beach, where the highest levels of *E. coli* were typically coincident with rainfall. However, levels at Cove Beach did not necessarily exceed 100 cfu/100mL at the time of the rainfall event. The degree to which *E. coli* levels were elevated above 100 cfu/100mL in the absence of rainfall was lower at the Cove Beach than at the Main or St. Christophers Beaches where results frequently exceeded 100 cfu/100mL with no rainfall the day of sampling or the previous day.

The diversity in patterns of correspondence between rainfall and levels of *E. coli* among the beaches strongly suggests that responses to rainfall vary among beaches.

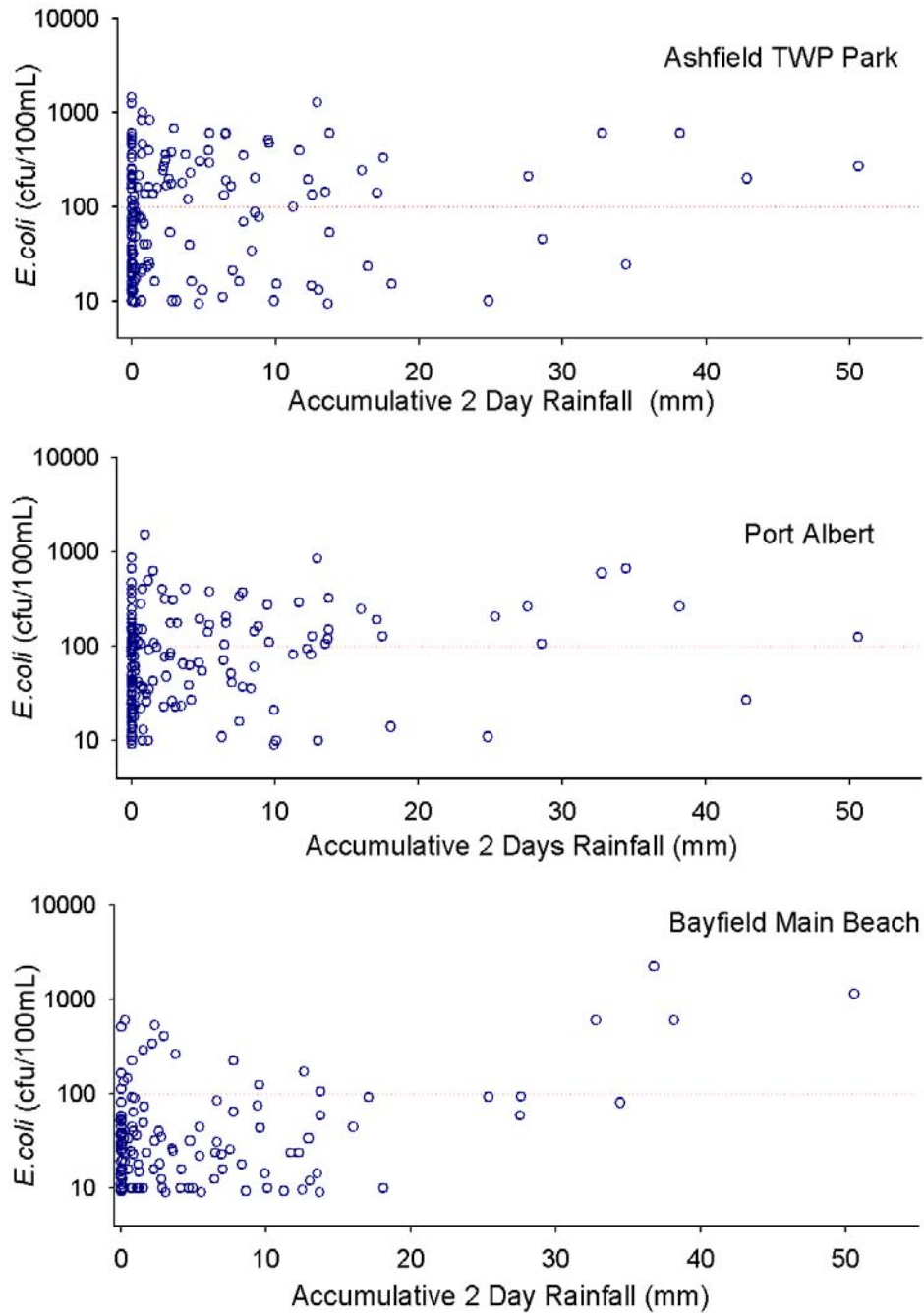


Figure 20: Geometric mean *E.coli* for individual samples sets plotted against two-day accumulated precipitation (as the average of measurements at the Lucknow, Benmiller and Varna gauges) for the Ashfield Township, Port Albert and Bayfield Main beaches (top, middle and bottom panels, respectively). These three beaches are adjacent to the mouths of tributaries.

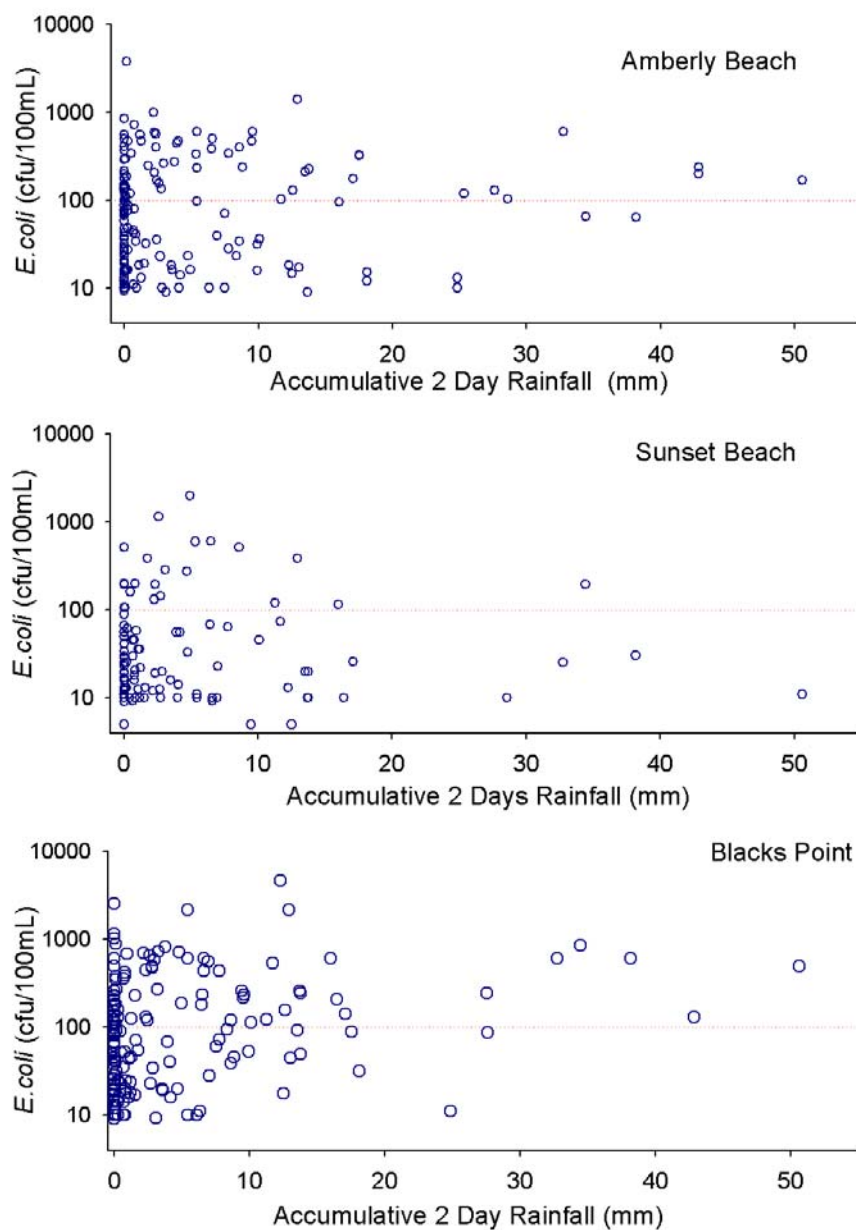


Figure 21: Geometric mean *E.coli* for individual samples sets plotted against two-day accumulate precipitation (as the average of measurements at the Lucknow, Benmiller and Varna gauges) for the Amberley, Sunset and Blacks Point beaches (top, middle and bottom panels, respectively).

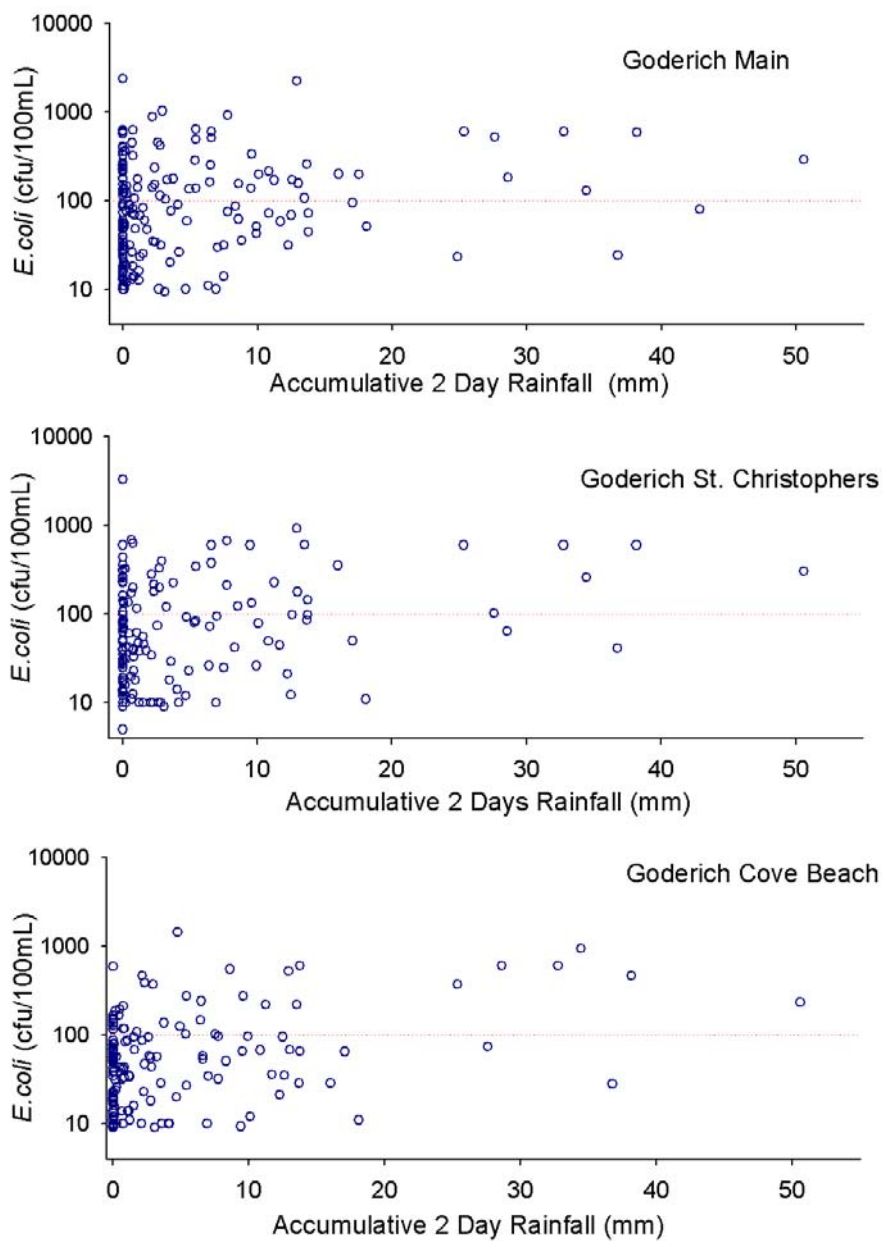


Figure 22: Geometric mean *E.coli* for individual samples sets plotted against two-day accumulate precipitation (as the average of measurements at the Lucknow, Benmiller and Varna gauges) for the Goderich Main, St. Christophers and the Cove beaches (top, middle and bottom panels, respectively).

2.4.5 Patterns of Occurrence in *E. coli* in Relation to Water Level in Lake Huron

There have been dramatic fluctuations in lake levels over the period 1993 to 2003.

Consequently, the susceptibility of shoreline beaches to microbial pollution may have varied over years as a result of changing beach position relative to the waterline, variable water table, variable shore discharge points, or the overall physical extent of the beach. For example, during low water levels, the beach areas are wider and the waters edge further from the developed or forested areas behind the beaches.

Lake levels as measured by Environment Canada at Goderich were relatively high from 1993 to 1998, peaking, in 1997, at approximately 120 cm above chart datum (Figure 23). Lake level then declined to low levels in 1999 to 2003. Lows of >20 cm below chart datum were reached in 2003. Seasonal water level fluctuations are normal and highest levels typically occur during the summer period. Mean water level over the June 1 to August 31 period was calculated on an annual basis to allow interannual seasonal water level comparisons (Figure 23). Water levels during the bathing season were highest in 1997 and lowest in 2001 and 2003.

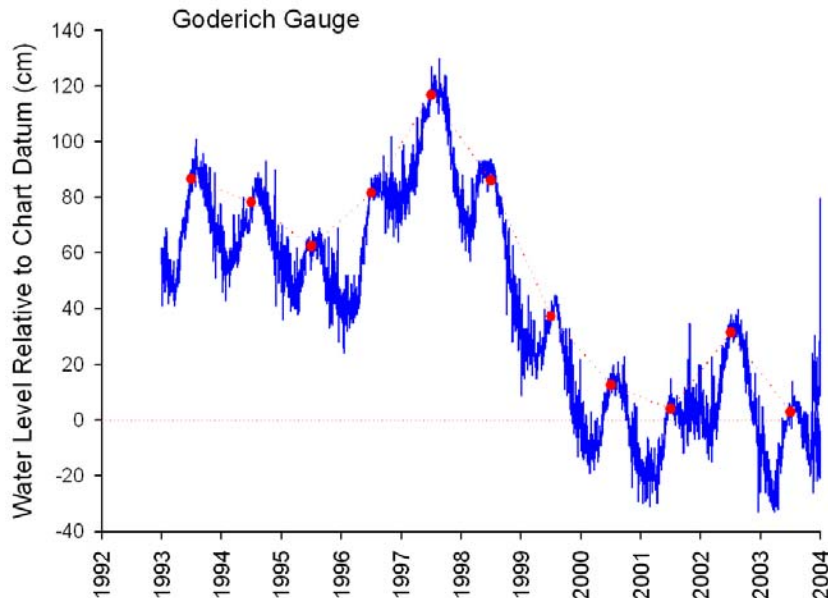


Figure 23: Mean daily water level in Lake Huron relative to chart datum at the Goderich gauging station (blue line). The red circles indicate the mean water level for the June 1 to August 31 periods for each year.

Summer median levels of *E. coli* for sample sets at beaches were plotted against water level features to explore any possible relationships (Figure 24). While no relationships were apparent, the widest ranges of median values were observed during two of the three low water level

years. However, collectively the ranges in median values in a year were unrelated to water level. The tendency for water levels to rise, fall or remain the same over the bathing season was examined by calculating the daily changes in water level over the summer and then averaging the daily changes for that period. Again, there was no relationship between median values and water level fluctuations for the summer period (Figure 24).

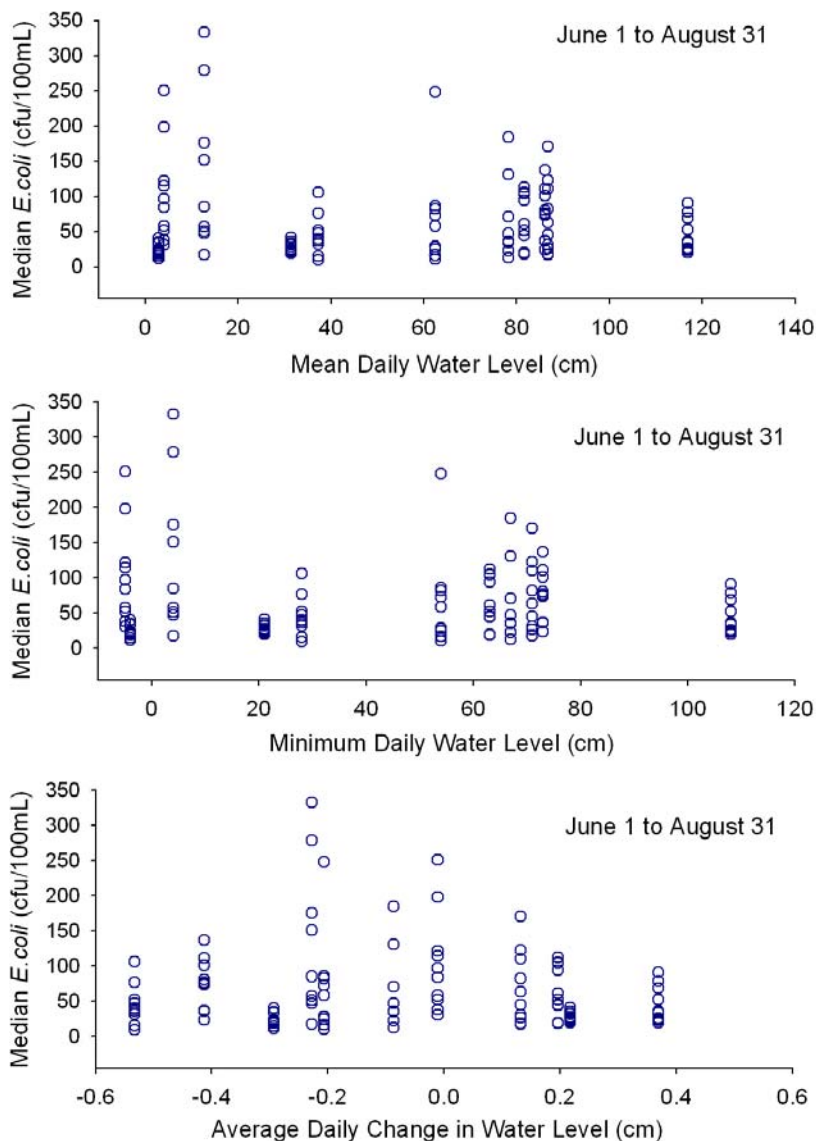


Figure 24: Annual median *E. coli* for beaches plotted against average (top panel) and minimum (middle panel) water level features over the June 1 to August 31 periods for years 1993 to 2003. Annual median *E. coli* is plotted against the average of the one-day water level changes over the period of June 1 to August 31 in the lower panel.

2.4.6 Patterns of Occurrence in *E. coli* in Relation to Water Temperatures of Lake Huron and Air Temperature of the Shoreline

The temperature of the nearshore waters of Lake Huron varies among years in response to weather patterns and during the summer can change dramatically over short periods of time as a consequence of lake-scale circulation, most notably upwelling and downwelling events.

The die-off rates of *E. coli* when in the lake environment may vary appreciably as a function of summer water temperatures, which typically range from $<10^{\circ}\text{C}$ to $>20^{\circ}\text{C}$. A portion of the observed differences in levels of *E. coli* observed over a monitoring season, and between seasons, may indeed result from variable die-off rates in response to lake temperatures. Rapid thermal fluctuations due to upwelling and the associated intrusion of cold hypolimnetic water into the shoreline areas is of particular interest not only for the effect that it has on water temperature itself but also as a mechanism by which lake water and suspended material occurring near the lake bed may be displaced toward shoreline waters.

Daily water temperature data collected at the Goderich Water Treatment Plant (WTP) was examined in relation to levels of *E. coli* among years, to explore possible correlations between fluctuations in levels of *E. coli* and water temperatures. Average water temperature over the summer (June 1 to August 31) ranged from approximately 16 to 20°C (Figure 25). The number of upwelling and downwelling events, which were defined as temperature changes of $>4^{\circ}\text{C}$ between consecutive days, could occur up to 8 times over a summer season. The years 1995 and 1996 had the most frequent mixing events and 2001 the fewest events (Figure 25).

There was little evidence of a relationship between water temperature and levels of *E. coli*. Scatter plots of median *E. coli* against mean water temperature and the 90th percentile of temperature are presented in Figure 26. No consistent relationship was discernible between lake temperature and median *E. coli* on an annual basis. However, there may have been a weak negative relationship between median *E. coli* and frequency of $>4^{\circ}\text{C}$ temperature fluctuations. The range in median values appeared widest in the two years with the least number of temperature fluctuations suggestive of upwelling or downwelling events.

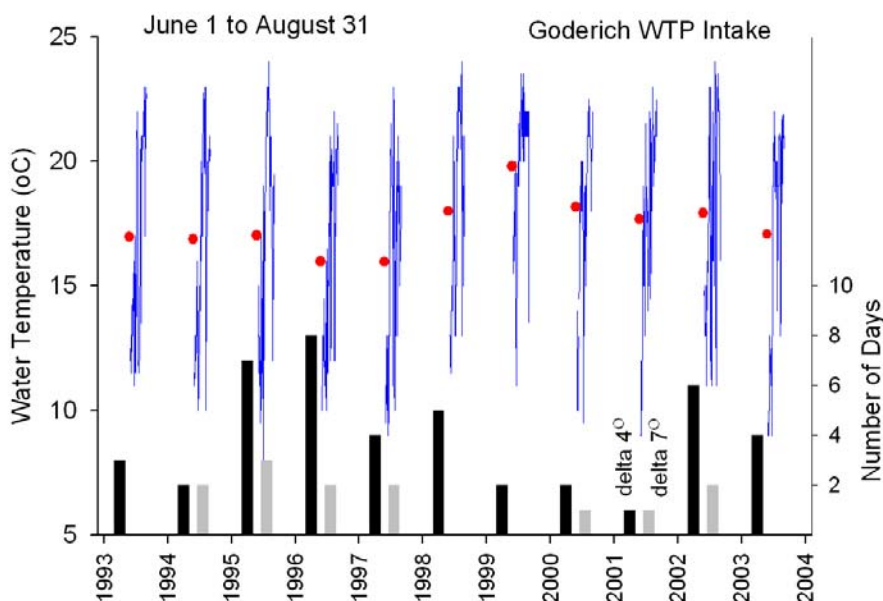


Figure 25: Daily temperature of water at the intake of the Goderich Water Treatment Plant from June 1 to August 31 (blue line). The red circles give the mean temperature over the same period. The bars indicate number of occasions between June 1 and August 31 where the temperature change between consecutive days was greater than 4° (black bar) and 7° (grey bar).

Air temperature measured at the Goderich airport (EC 6122847) near the shores of Lake Huron was also examined to explore the possible correspondence between the frequency of warm weather days (as inferred from mean air temperatures) and levels of occurrence of *E. coli* among years. Daily air temperature was calculated from hourly measurements and is presented along with the average daily air temperature over the June 1 to August 31 period (1994 to 2003; Figure 27). Average summer air temperature ranged from approximately 17.5 to 20°C among years.

There appears to be little relationship between air temperature and levels of *E. coli* when examined either as annual medians or geometric means for datasets (Figure 27). Elevated levels of *E. coli* (GM >100 cfu/100mL) were observed across the full range of daily mean air temperatures (Figure 27; top panel) and do not suggest a tendency for elevated levels to occur on warm days which may be more conducive to beach usage.

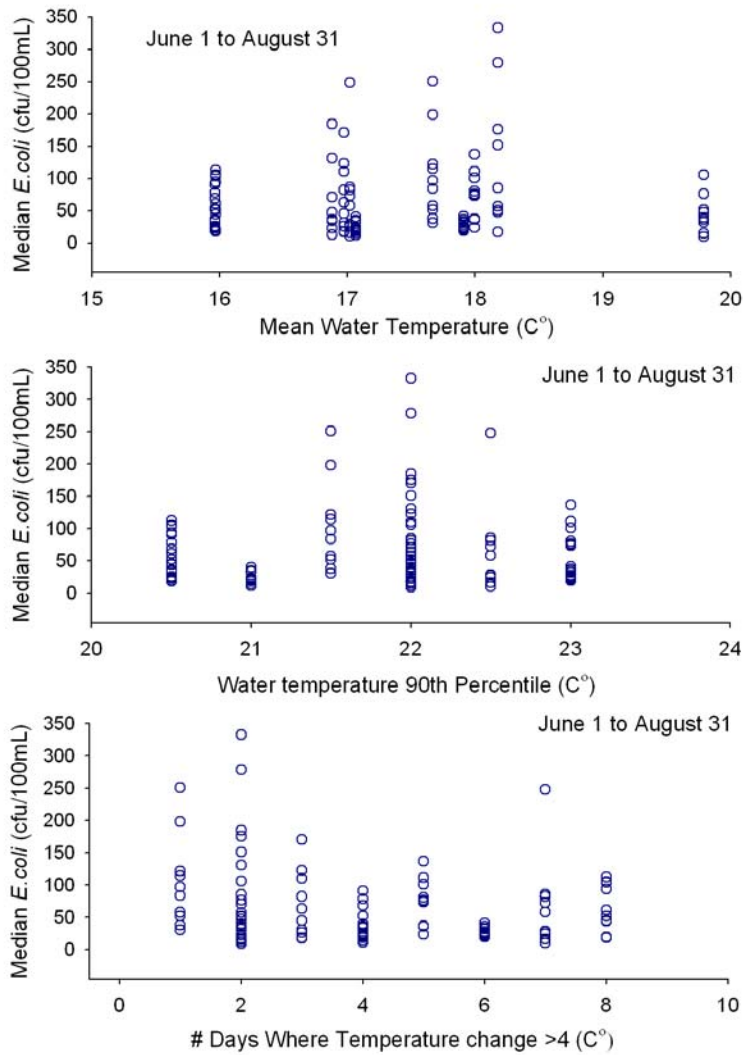


Figure 26: Median *E. coli* at individual beaches is plotted against average daily water temperature and the 90th percentile at the Goderich Water Treatment Plant intake for the June 1 to August 31 period in the top and middle panels, respectively. Median *E. coli* is plotted against number of days when daily temperature fluctuated by >4°C between consecutive days in the bottom panel.

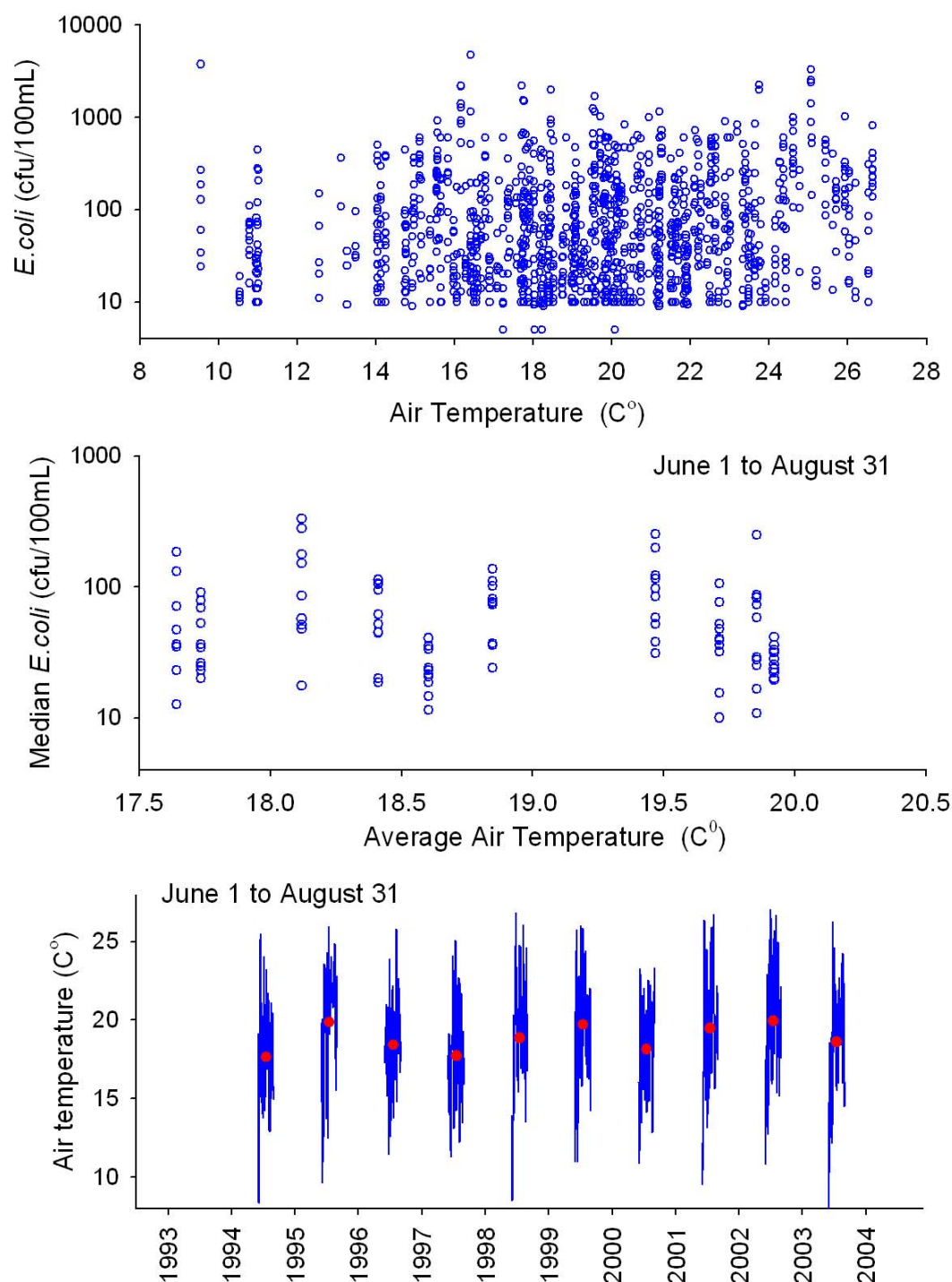


Figure 27: Daily average air temperature at the Goderich Airport (EC station 6122847) from June 1 to August 31 (blue line) along with the mean temperature over the same period (red circles) is given in the lower panel. Median *E. coli* at beaches is plotted against average air temperature for June 1 to August 31 period in the middle panel. Geometric mean of *E. coli* for all data sets collected at the study beaches from 1994 to 2003 is plotted against daily average air temperature at the Goderich Airport in the top panel.

2.4.7 Patterns of Occurrence in *E.coli* in Relation to Turbidity and Suspended solids in the Nearshore of Lake Huron

Periodic wind-induced resuspension of lake sediments contaminated with *E. coli* into the water column and subsequent movement of sediment-laden water onto the lakeshore has been hypothesized as a mechanism contributing to elevated levels of *E. coli* at beaches. If the redistribution of lake sediments is a feature of the contamination of shoreline beaches, then it follows that fluctuations in suspended solids in nearshore waters should correlate with the impact of bottom sediments on beaches. Unfortunately, detailed through-time information on the levels of suspended solids is not readily available. However, daily measures of turbidity were made on the raw water at the Goderich WTP over the period 1993 to 2003 and were used here as a surrogate for suspended solids in the nearshore. It is recognized that turbidity values will be highly variable along the shoreline at any one point in time, however, strong fluctuations in suspended solids as a result of winds should be correlated along the largely open shores of Lake Huron between Bayfield and Amberley Beach.

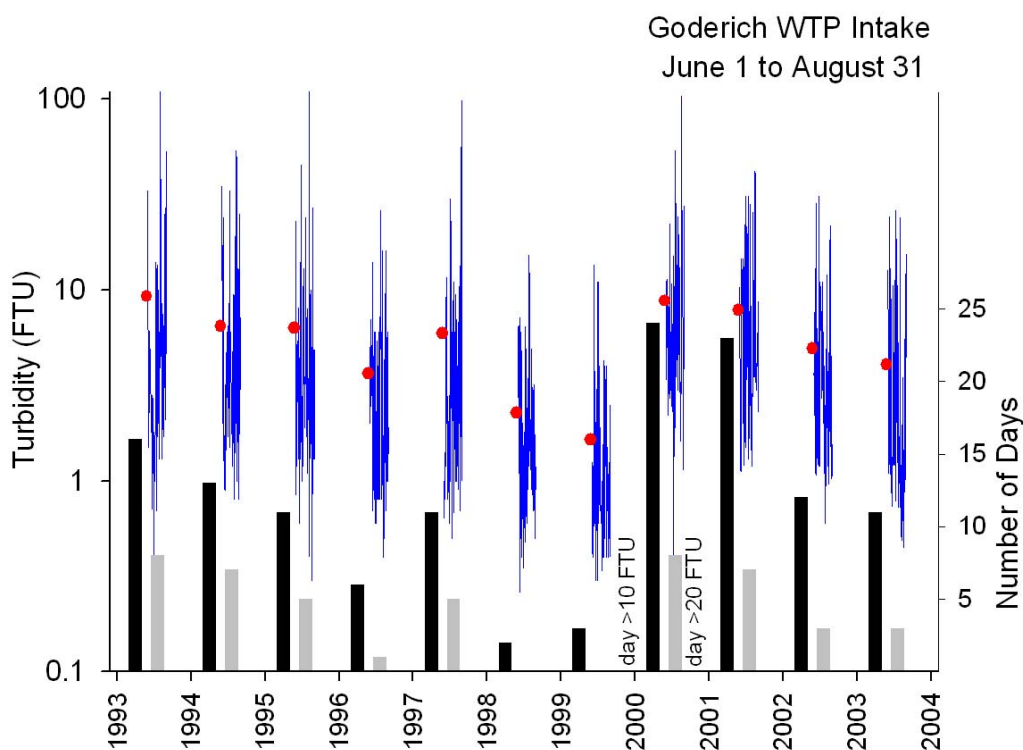


Figure 28: Daily turbidity measurements at the intake of the Goderich WTP from June 1 to August 31 (blue line). The red circles give the mean turbidity over the same period. The bars indicate number of days between June 1 and August 31 when the turbidity levels were greater than 10 FTU (black bar) and 20 FTU (grey bar).

The daily fluctuations in turbidity at the intake of the Goderich WTP can be appreciable with turbidity ranging from levels indicative of clear water and low in solids (turbidity <1 FTU) to murky water approaching 100 FTU (Figure 28). Average turbidity among years over the June 1 to August 31 period has ranged from just under 2 FTU to just over 9 FTU. The frequency of high turbidity events has varied appreciably among years ranging from 2 to 24 days considering the days when turbidity exceeded 10 FTU (Figure 28).

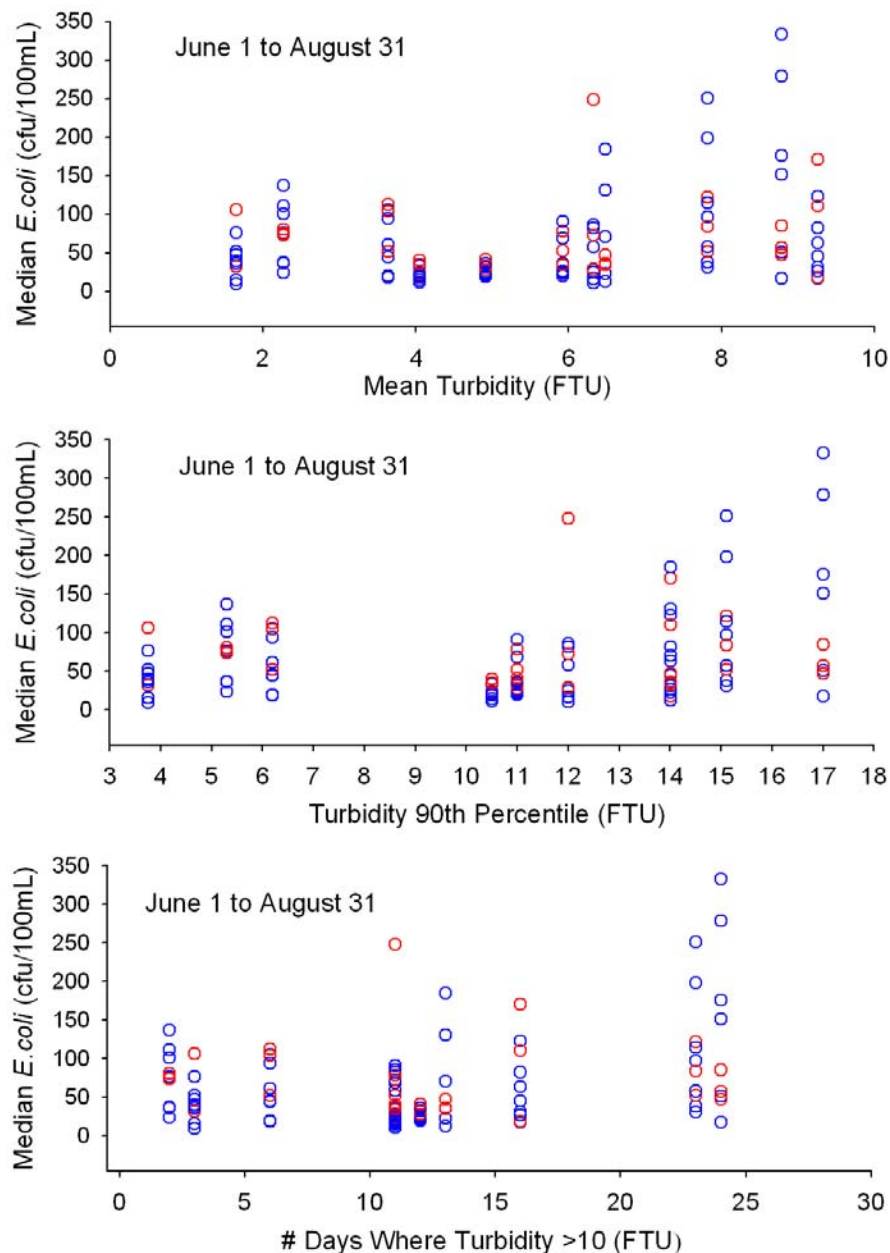


Figure 29: Median *E. coli* at study beaches is plotted against average turbidity and the 90th percentile for at the intake of the Goderich Water Treatment Plant for the June 1 to August 31 period in the top and middle panels, respectively. Median *E. coli* is plotted against number of days where turbidity >10 FTU in the lower panel. Data points for the Goderich beaches are shown in red.

Plots of annual median *E. coli* at beaches as a function of mean turbidity, the 90th percentile and the number of days with turbidity exceeding 10 FTU all suggest a weak positive relationship between levels of *E. coli* and elevated turbidity (Figure 29). The range in median values in years appears to increase with mean turbidity, the 90th percentile and the number of days with elevated turbidity. Also of note is that the two years with most frequent days of high turbidity as inferred from days with >10 FTU, were also among the years with the most frequent occurrence of elevated levels of *E. coli*.

2.4.8 Patterns of Occurrence in *E. coli* in Relation to Wind Events in the Nearshore of Lake Huron

Hourly wind data collected at the Goderich airport was further used to explore the potential relationships between wind, sediment transport and incidents of elevated levels of *E. coli* at recreational beaches. The analysis presented here is at best approximate since wind conditions at the shoreline of the lake can be highly variable among locations being influenced by local conditions. Consequently, the degree to which the data collected at the Goderich Airport reflects conditions in the nearshore of the lake, even in the Goderich area, is uncertain.

Wind stress, a measure of the wind energy operating on the lake surface was calculated for compass directions representing, offshore, onshore, onshore from the NW and onshore from the SW and summed over the period June 1 to August 31 for the years 1994 to 2003. This represented a gross indicator of the frequency of wind disturbances between years. When annual median *E. coli* for beaches were plotted against wind stress there appeared to be a weak positive correlation between wind stress in the onshore directions and the range in annual median values (Figure 30). Of note, the year with atypically high wind stress from the SW corresponded to the year with the widest range in median values.

The detailed *E. coli* monitoring data for the Goderich beaches and Sunset Beach to the north of Goderich were also examined in relation to the Goderich Airport wind data. The geometric means for all datasets collected at the Goderich beaches and Sunset Beach are plotted against wind features in Figures 31 and 32, respectively.

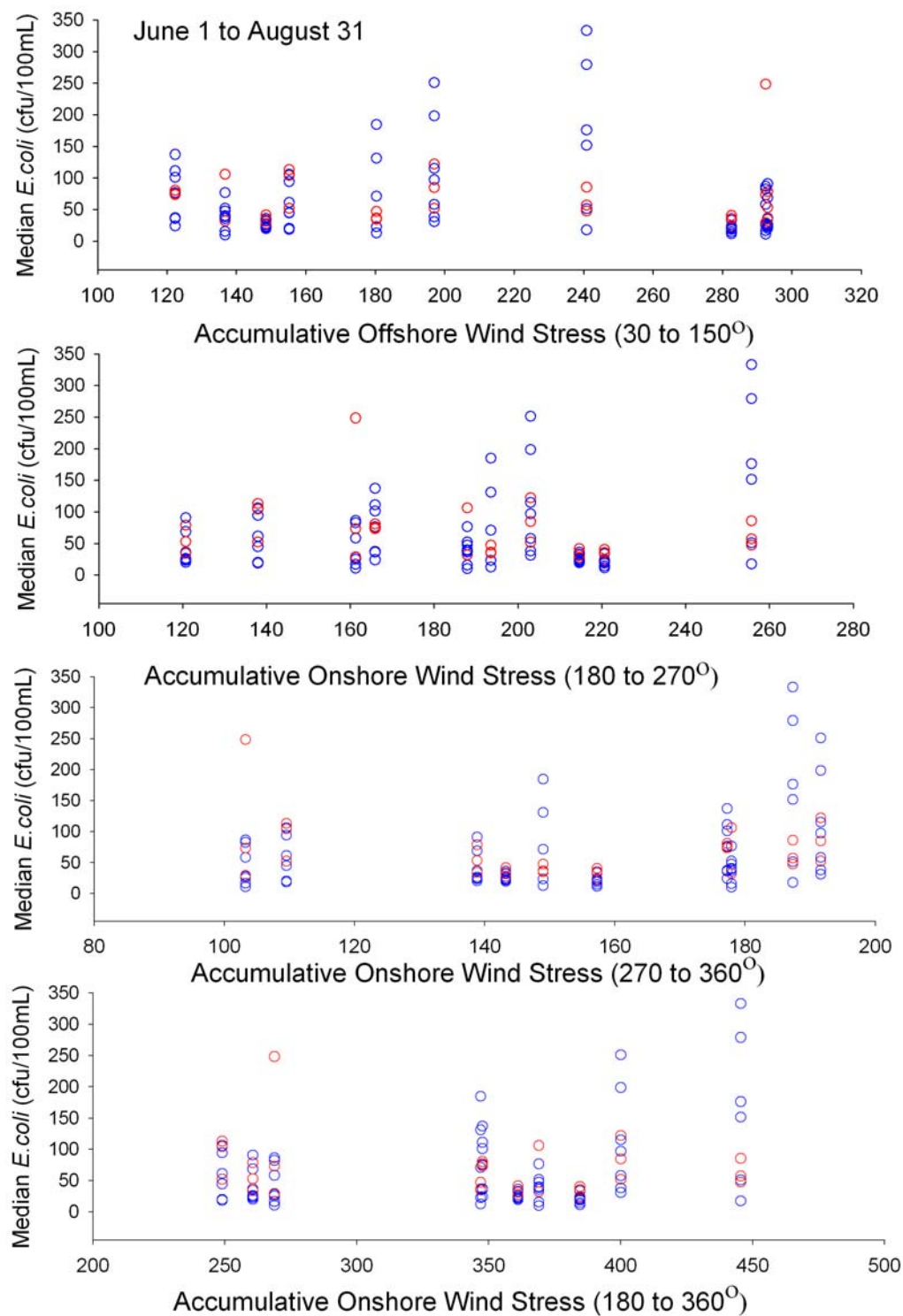


Figure 30: Median *E. coli* at study beaches is plotted against accumulative wind stress (X-axis is dynes/cm²) for offshore (top panel), onshore-SW (2nd from top), onshore-NW (3rd from top), and onshore (bottom panel) directions based on hourly wind speed measurements at Goderich Airport. Data points for the Goderich beaches are shown in red.

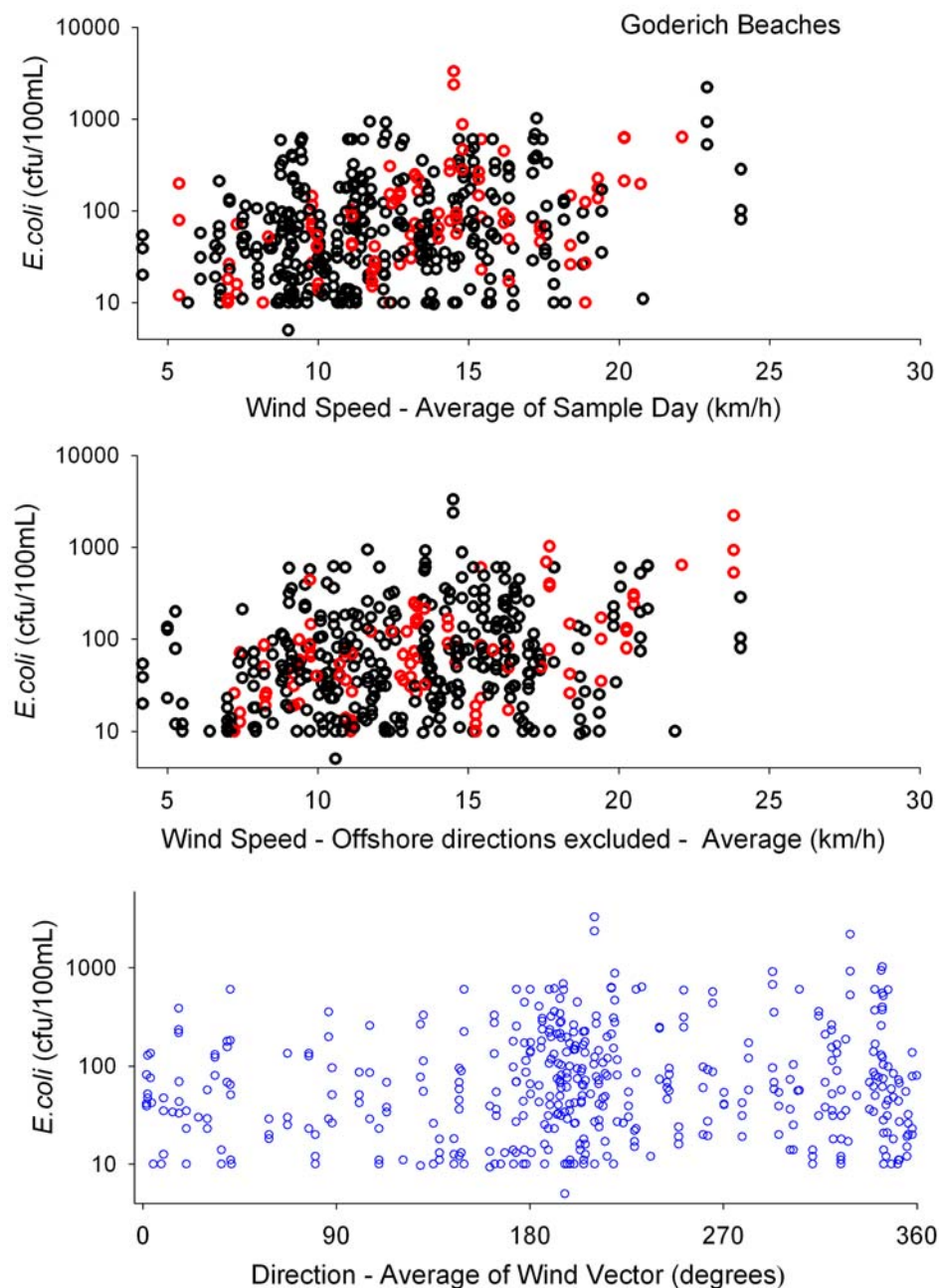


Figure 31: Geometric means of *E. coli* in sample sets collected at Goderich beaches for the years 1994 to 2003 plotted against average wind speed on the day of sampling is presented in the top panel. The red circles indicate datasets collected when the vector averaged wind direction was from the SW (200 to 250 degrees). Other directions are in black. The middle panel provided comparable information to the top panel except that wind data from offshore directions (30 to 150 degrees) have been excluded in the calculation of average wind speed and vector averaged direction. Geometric means of *E. coli* in sample sets are plotted against vector averaged wind direction on the day of sampling in the bottom panel.

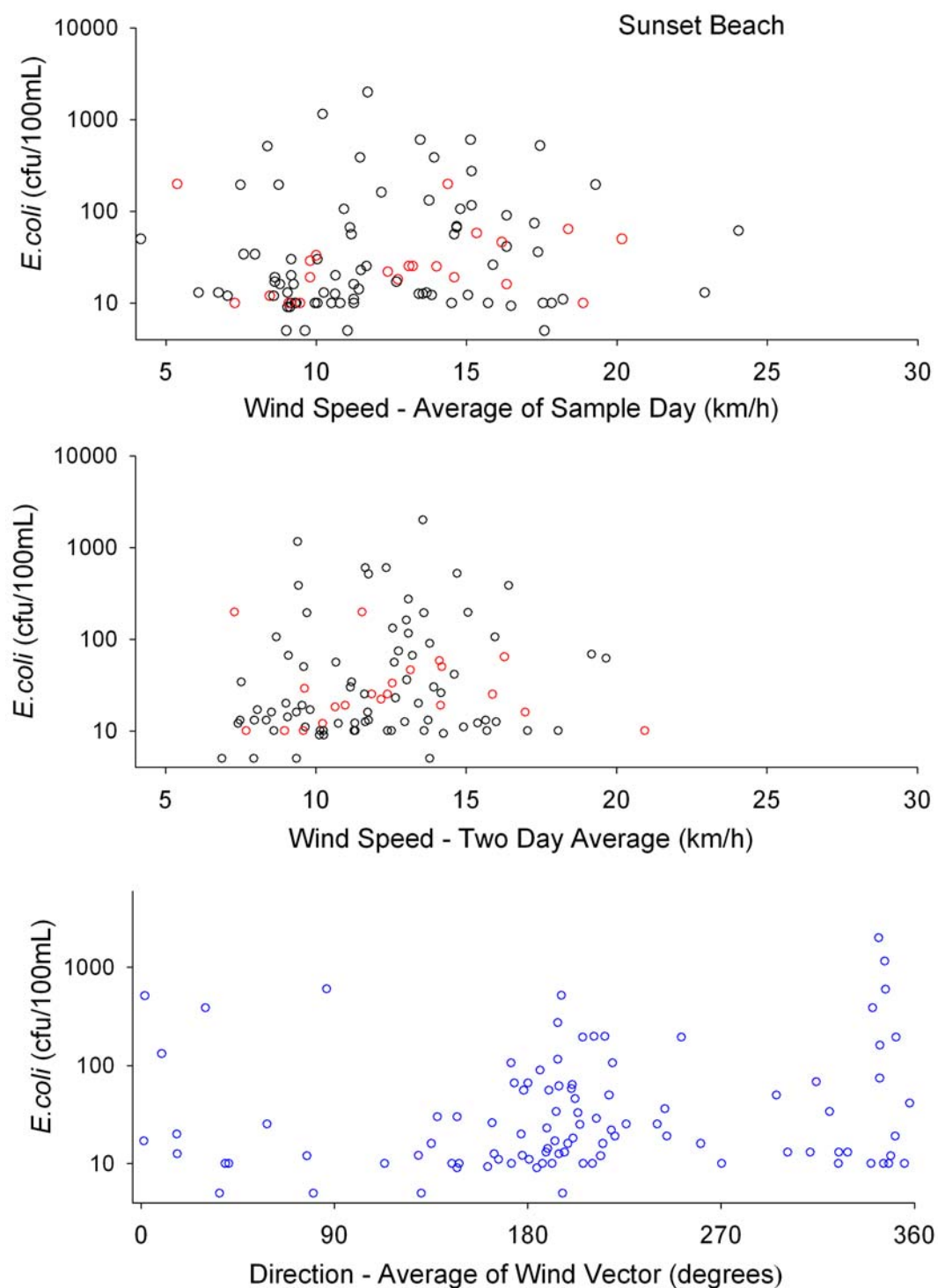


Figure 32: Geometric means of *E. coli* in sample sets collected at Goderich beaches for the years 1994 to 2003 plotted against average wind speed on the day of sampling is presented in the top panel. The red circles indicate datasets collected when the vector averaged wind direction was from the SW (200 to 250 degrees). The middle panel provided comparable information to the top panel except that wind data from offshore directions (30 to 150 degrees) have been excluded in the calculation of average wind speed and vector averaged direction. Geometric means of *E. coli* in sample sets are plotted against vector averaged wind direction on the day of sampling in the bottom panel.

At the Goderich beaches there appears to be a positive relationship between wind speed on the day of sampling and geometric means in sample sets. The degree to which geometric means exceed 100 cfu/100 mL appears to be higher at wind speeds approaching 10 km/h and higher. However, this may be a result of comparatively fewer data for days when winds were <10 km/h. Nonetheless there appears to be a positive slope between geometric means and wind speed for the dataset as a whole and in particular for the geometric means for samples collected on days when the vector averaged wind direction was from the SW. It is important to note that there were numerous occasions when wind speed over the day ranged from 10 to 20 km/h and *E. coli* levels were <100 cfu/100mL, however, in most instances when wind speed averaged >20km/h the geometric means exceeded 100 cfu/100 mL (Figure 31).

The results for Sunset beach are also suggestive of an influence of wind on *E. coli* levels. The numbers of geometric means exceeding 100 cfu/100 mL do not appear to vary with wind speed, however, there appears to be a weak trend of increasing geometric means with wind speed (Figure 32). Similar to the Goderich beaches, a positive relationship between *E. coli* medians and SW wind speeds (daily or 20 day averages) was apparent.

2.4.9 Patterns of Occurrence in *E. coli* in Relation to Water Quality in the Nearshore of Lake Huron

Water quality in the anthropologically perturbed tributaries of the SE shores of Lake Huron is fundamentally different from the oligotrophic waters of Lake Huron into which these tributaries drain. Tributary water is typically enriched in nutrients, ions, suspended solids and organic material (both suspended and dissolved) compared to Lake Huron water. The quality of water in the nearshore of the lake is affected by the loading of these tributaries which contributes to the typically high level of temporal variability in the chemistry of nearshore waters. Several of the water quality features of the nearshore that are, in part, affected by tributaries are known to affect die-off rates of *E. coli* and consequently may affect the levels of *E. coli* observed at the shoreline. In particular, nutrient levels, organic material and light-attenuating particles may affect die-off rates of *E. coli* in lake water, and these are nearshore features commonly affected by tributary inputs.

Over the years, a number of selected raw water parameters have been monitored at the Goderich WTP as part of a Great Lakes nutrient monitoring program by MOE and data is available for the years 1993 to 2003. These data provide insight on the seasonal and interannual variability in water quality that occurs in the nearshore waters of Lake Huron in the Goderich area. The data can be used to provide two perspectives on water quality which are relevant to the annual fluctuations in levels of *E. coli* at recreational beaches. First, the data provide information on the variability in water quality between 1993 and 2003. This variability may potentially correlate to *E. coli* fluctuations observed in the beach monitoring program because die-off rates are known to be related to water-quality parameters. Secondly, features of variability in water quality reflect the degree of the watershed influence on the nearshore waters. It is possible that fluctuations in the levels of *E. coli*, in part, reflect variability in the loading from tributaries and may correlate with water quality features reflecting tributary influence. Water quality is often variable along even limited areas of the shoreline and consequently chemical concentration data collected at one location will not be representative of other locations. However, large-scale patterns of temporal variability are likely to be similar along adjacent shorelines with similar physical conditions. An assumption here is that the water chemistry data collected at the Goderich intake will reflect patterns in temporal variability over the shoreline where the ten study beaches occur.

The fluctuations in nitrates over the March 1 to August 31 period for the years 1993 to 2003 are shown in Figure 33 and can be used as an indicator of watershed influence on the nearshore. Seasonally, concentrations range from highs in early spring in excess of 2 mg/L and decline to <0.5 mg/L at the end of the summer. The high spring concentrations are a result of land-derived nitrate inputs to the lake. The decline over the spring to summer period reflects declining loading as tributary flows decline over late-spring to summer and biological utilization of nitrates in the lake.

Annual medians of *E. coli* for the study beaches are plotted against mean concentrations of nitrates for the March 1 to May 31 and June 1 to August 31 periods in Figure 33. There does not appear to be any relationship between median levels of *E. coli* during the summer and nitrate levels during either the spring or summer period.

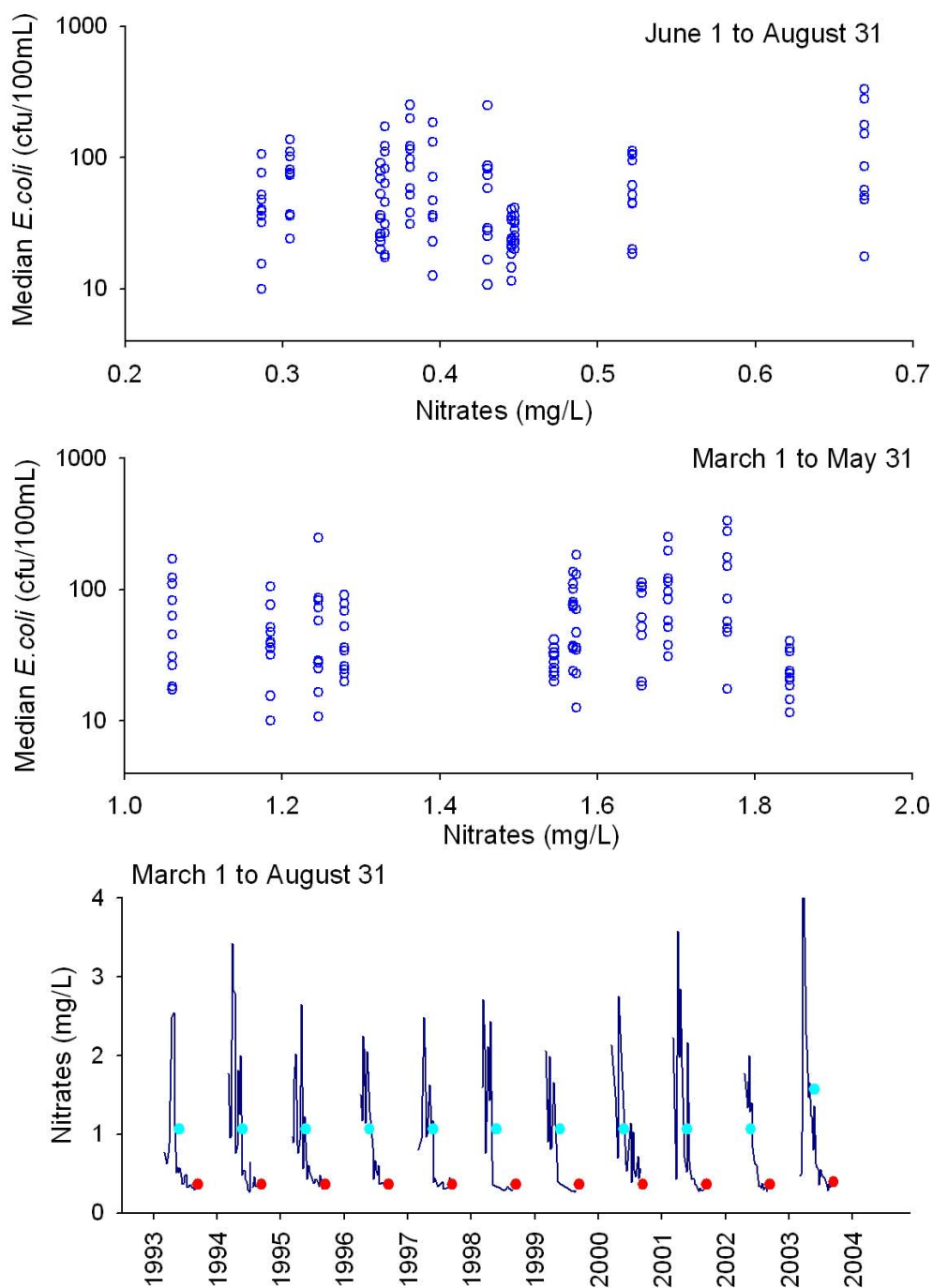


Figure 33: The lower panel presents the concentrations of nitrates (as mg/L N) in raw water collected at approximately weekly intervals at the Goderich Water Treatment Plant intake over the period March 1 to August 31 for year 1993 to 2003 in the MOE Great Lakes Intake Biomonitoring Program. The blue circles are the means for the March 1 to May 31 period and the red circles are the means for the June 1 to August 31 periods. The upper panel provides a scatter plot of median *E. coli* for beaches over the years 1993 to 2003 against mean nitrate for the June 1 to August 31 period. The middle panel provides a scatter plot of median *E. coli* against mean nitrate for the March 1 to May 31 period.

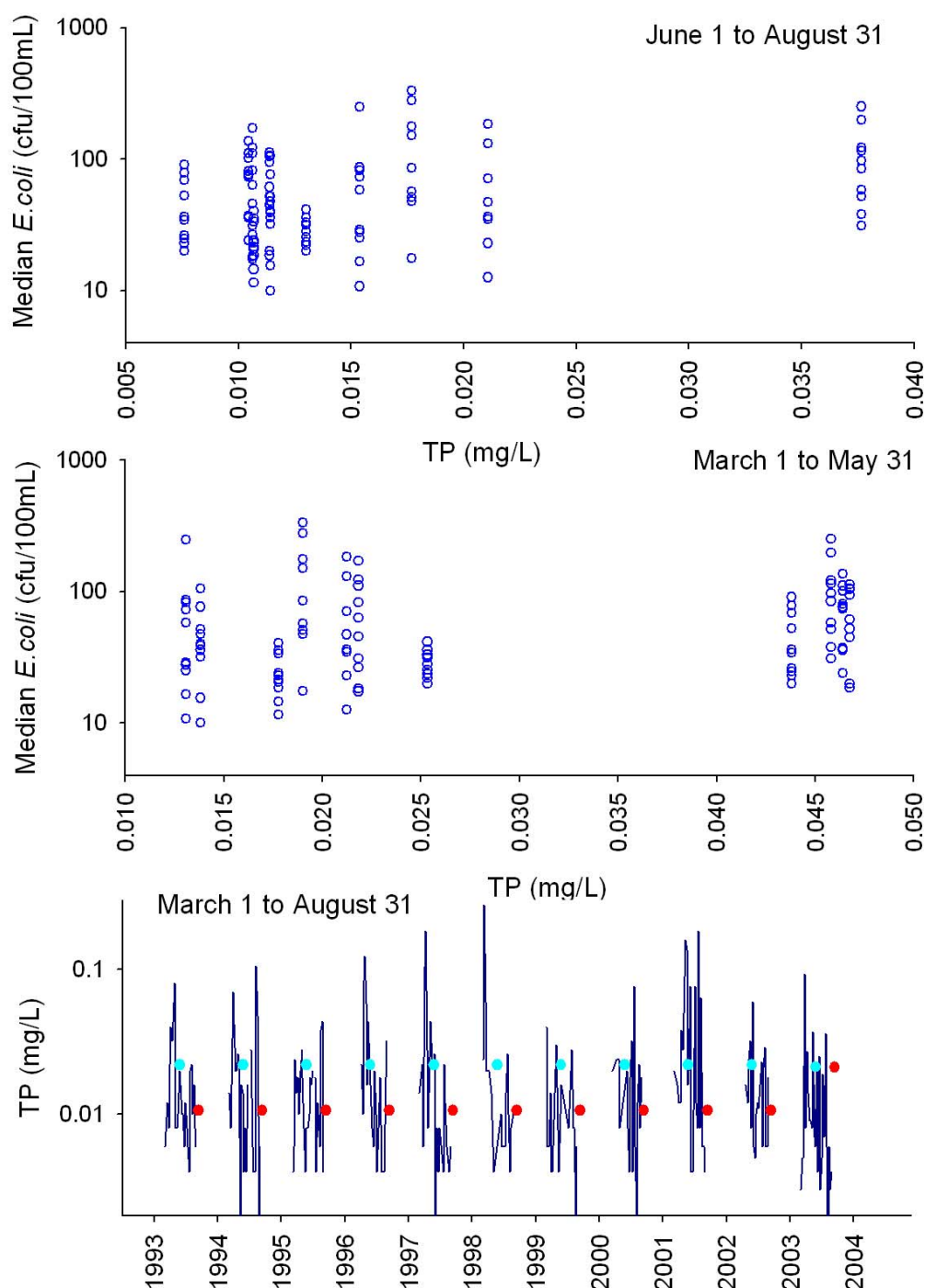


Figure 34: The lower panel presents the concentrations of total phosphorus in raw water collected at approximately weekly intervals at the Goderich Water Treatment Plant intake over the period March 1 to August 31 for year 1993 to 2003 in the MOE Great Lakes Intake Biomonitoring Program. The blue circles are the means for the March 1 to May 31 period and the red circles are the means for the June 1 to August 31 periods. The upper panel provides a scatter plot of median *E. coli* for beaches over the years 1993 to 2003 against mean total phosphorus for the June 1 to August 31 period. The middle panel provides a scatter plot of median *E. coli* against mean total phosphorus for the March 1 to May 31 period.

The fluctuations in total phosphorus (TP) over the March 1 to August 31 period for the years 1993 to 2003 are shown in Figure 34 and can be used as an indicator of both watershed influence on the nearshore waters and the nearshore trophic status (biological and nutrient richness). There are pronounced seasonal changes in concentrations from spring to summer, however, the changes are more erratic than those observed with nitrates. Ambient levels of TP in oligotrophic and nutrient poor waters of Lake Huron are <0.01 mg/L, these levels are more frequently observed in the summer than the spring water intake data. While elevated levels of TP may reflect the input of nutrients to the nearshore of the lake, it is also important to consider that the levels of TP in coastal waters can be elevated because of physical disturbances in the lake and particulate material in samples resuspended from the lake bed.

As with nitrates, there is no apparent relationship between annual median *E. coli* levels, or the ranges in median, and the concentration of TP for the March 1 to May 31 or the June 1 to August 31 periods (Figure 34)

2.4.10 Patterns of Occurrence in *E. coli* in Relation to Physical Character and Land-use in the Areas Draining to Beaches

The land areas draining to the Lake Huron shoreline adjacent to the ten study beaches are predominately rural landscapes dominated by agriculture. However, each of the beach-drainage area associations has unique features which potentially influence the sources of fecal pollutants to the beaches and the mechanisms by which these pollutants are delivered to the beaches. The size and complexity of the drainage areas directly associated with the beaches varies widely from small areas with limited surface water to large tributary systems with many kilometers of rivers. The degree to which the drainage areas support human populations is also variable ranging from small towns to scattered dwellings. In an effort to identify traits of the drainage areas which may be correlated with level of occurrence of *E. coli* at beaches, selected features of the drainage areas were tabulated and plotted against the percentages of sample sets over the 1993 to 2003 period when geometric means exceeded 100 cfu/100 mL. The drainage areas directly associated with beaches for the analysis were arbitrarily defined by the tributaries that contacted the lake within 1 km of any point over the beach area where samples

are normally collected by the HCHU routine beach monitoring program (based on site visits with HCHU staff in May 2004).

Notwithstanding that there are appreciable deficiencies in the land use information used in the analysis, there do not appear to be any obvious correlates between land use features of the drainage areas and frequency in which *E. coli* were elevated in the shoreline waters. The size of the drainage area directly associated with a particular beach did not appear to be related to beach quality (Table 6). Two beaches with a relatively high frequency of elevated levels of *E. coli*, Blacks Point and Amberley Beach, are among the beaches with small drainage areas (<12 km²). These areas contrast sharply in magnitude with the drainage areas of Ashfield Township Beach and Port Albert Beach with moderate drainage areas (100 to 250 km²) and Goderich Main Beach with a massive drainage area (2500 km²). All of these mentioned beaches have a high frequency of elevated levels of *E. coli*.

Plots of percentages of sample sets with geometric means >100 cfu/100 mL against selected features of land use and physical character of drainage areas are given in Figures 35 and 36. The percentages of forested or agricultural land in the drainage area did not appear to be related to the frequency of elevated *E. coli* among the beaches; however, this may not be surprising given the similarity in land use among drainage units. The lack of any correspondence in frequency of sample sets above the PWQO with absolute amounts of land classified as agricultural, or the cumulative watercourse length in the drainage area, is potentially significant and is inconsistent with the general hypothesis that cumulative loading of fecal pollutants to the lakeshore contributes to the degree of impact at beaches.

The delineation and physical characterization of drainage areas to the lake was accomplished using the WRIP (Water Resources Information Project) GIS Toolbox created by staff at the Ontario Ministry of Natural Resources (MNR). The watersheds were delineated using the MNR Provincial DEMs (digital elevation model) which are at a scale of 1:10,000 and have a grid cell size of 10 metres for Southern Ontario. Once the watersheds were delineated, the WRIP GIS Toolbox was also used to characterize the watersheds using landcover datasets.

The assignments of land areas as agricultural, forested and urban/built up in Table 6 and 7 were derived from the MNR Ontario Land Cover data. The MNR data was derived from digital, multispectral LANDSAT Thematic Mapper data recorded on a range of dates between 1986 and 1997, but the majority of the satellite data frames were recorded in the early 1990s. The data

base was produced in nine segments (termed "tiles") under three separate programs of the MNR between 1991 and 1998 (see Spectranalysis Inc. 1999). Some of the small urbanized areas in the drainage areas are not represented in the MNR data. The documentation of the dataset acknowledges some uncertainty in the classification of small urban areas in agricultural settings; quoting from Spectranalysis Inc. (1999) "Practical experience suggests that the accuracy of the original, high-resolution land cover data is 85 percent for agricultural land cover, taking into account the presence of widespread, unavoidable confusion with small towns and roads."

The tile drain information for Huron and Bruce County was provided as GIS layers by the Ontario Ministry of Food and Agriculture (OMAF) and is based on records maintained by OMAF of projects by licensed tile drain contractors extending back to the 1980s.

The approximations of people and dwellings for the drainage areas were derived from 2001 federal census data using GIS. The census data collected at the dissemination area level (i.e. the smallest area level of packaged census information available) was overlaid by the individual drainage areas, the borders of which were used to "clip out" portions of the dissemination areas contained within it. The ratios of respective dissemination areas "clipped" by the drainage area were multiplied by the total numbers for each respective (entire) dissemination area. All the "clipped" information portions multiplied by the respective ratios were then summed to provide a total for an entire drainage area.

Table 6: Landuse features of areas draining in proximity to shoreline of study beaches. Physical features. See text for details of the analysis.

Beach	Drainage Area Contacting Lake 1km N to 1km S of beach (km ²)	Distance to tributary >1km in length (m)	Length of Tributaries (km)	Area Forested (km ²)	Percent of Area Forested (km ²)
Amberley	1.7	690	2.5	0.1	3
Ashfield Township Park	109	100	180	7.9	7.2
Port Albert	251	50	440	60	24
Sunset Beach	15	150	26	2.1	14
Goderich Main Beach	2500	380	3200	362	14
Goderich St.Christophers	2500	650	3200	362	14
Goderich The Cove	1.6	1000	4.3	0.4	24
Blacks Point	11	420	20	1.3	12
Bayfield Main	500	160	440	50	9.9
Bayfield South	500	770	440	50	9.9

Table 7: Land use features of areas draining in proximity to shoreline of study beaches. Anthropogenic features. See text for details of the analysis.

Beach	Drainage Area Contacting Lake 1km N to 1km S of beach (km ²)	Area Urban/Built Up lands (km ²)	Percent of Area as Urban/Built Up lands	Approximate Population Size (2001)	Design Capacity of Sewage Treatment Plants (m ³ /day)	Approximate number of dwellings (2001)
Amberley	1.7	0	0	<100	0	<50
Ashfield Township Park	109	0	0	3400	0	1400
Port Albert	251	0	0	3400	750	1100
Sunset Beach	15	0	0	<250	0	<125
Goderich Main Beach	2500	9.7	0.4	21000	26,012	7700
Goderich St.Christophers	2500	9.7	0.4	21000	26,012	7700
Goderich The Cove	1.6	< 0.01	0.2	<250	9,050	<125
Blacks Point	11	0	0	<100	0	<50
Bayfield Main	500	2.4	0.5	13000	7,576	5300
Bayfield South	500	2.4	0.5	13000	7,576	5300

Table 8: Landuse features of areas draining in proximity to shoreline of study beaches. Agricultural features. See text for details of the analysis.

Beach	Drainage Area Contacting Lake 1km N to 1km S of beach (km ²)	Area Agricultural lands (km ²)	Percent of Area as Agricultural lands	Counties over which drainage area extends	Tile Drains: Percent of drainage area not accounted for	Area Tile Drained (km ²)	Percent of Area Tile Drained (km ²)
Amberley	1.7	1.6	95	Huron, Bruce	0	0.9	56
Ashfield Township Park	109	100	91	Huron, Bruce	0	79	73
Port Albert	251	182	73	Huron, Bruce	0	70.5a	40a
Sunset Beach	15	12	83	Huron	0	8.9	60
Goderich Main Beach	2500	2100	82	Huron, Bruce, Perth, Wellington	30%	590	49b
Goderich St.Christophers	2500	2100	82	Huron, Bruce, Perth, Wellington	30%	590	49b
Goderich The Cove	1.6	1	70	Huron	0	0.1	3
Blacks Point	11	10	87	Huron	0	6.4	58
Bayfield Main	500	440	88	Huron, Perth	21%	230	72
Bayfield South	500	440	88	Huron, Perth	21%	230	72

Note: b-the tile drain area is for the portions of the drainage areas within Huron and Bruce Counties only; the percentage of area tile drained is relative to the portion of the drainage areas in Huron and Bruce Counties only. a- for area accounting for only 98% of drainage area.

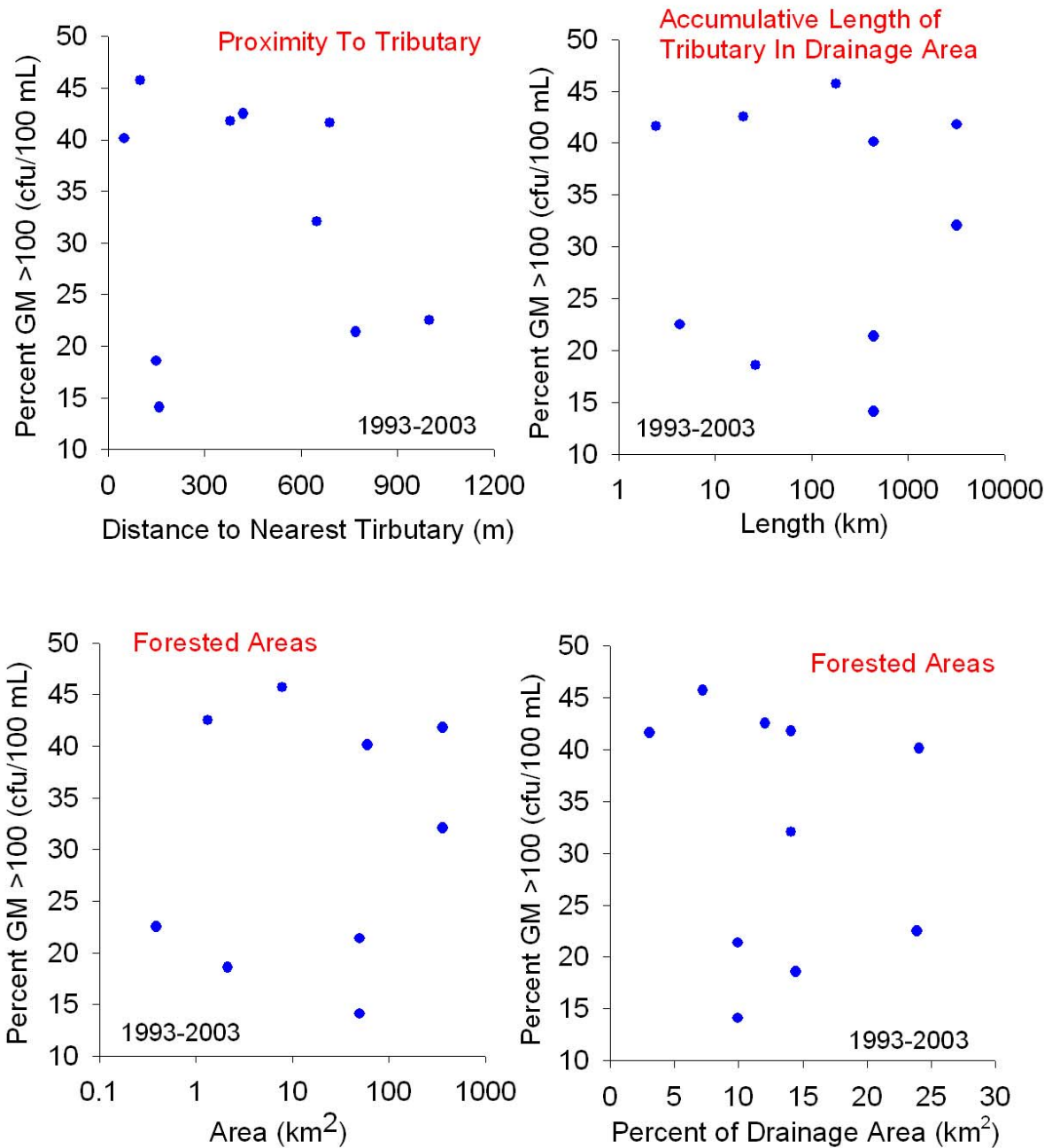


Figure 35: Percent of samples sets collected between 1993 and 2003 when geometric means exceeded 100 cfu/100mL plotted against nearest distance to a tributary > 1 km in length (top left), accumulative length of tributaries in drainage area (top right), area of forested land (bottom left) and percent of land area forested (bottom right).

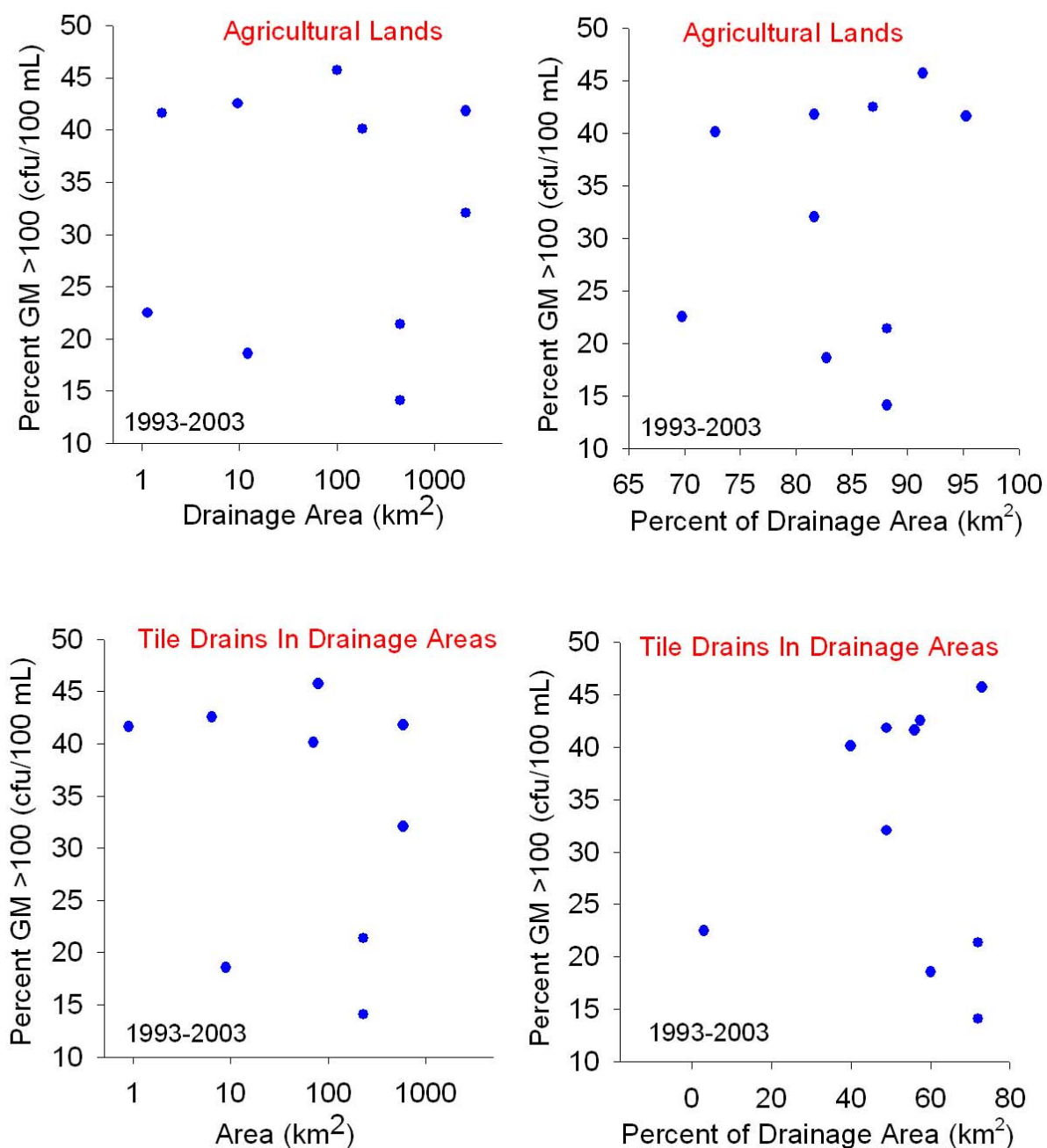


Figure 36: Percent of samples sets collected between 1993 and 2003 where geometric means exceeded 100 cfu/100mL plotted against area of agricultural land (top left), percent of land area as agricultural lands (top right), area of tile drains in Huron County and Bruce County component of drainage area (bottom left) and percent of area tile drained in Huron County and Bruce County component of drainage area (bottom right).

2.5 Monitoring of *E. coli* at a Recreational Beach of Point Farms Provincial Park by the Ontario Ministry of Natural Resources

A beach water quality program was conducted over the bathing season at Point Farms Provincial Park located approximately 6 km north of Goderich on the shores of Lake Huron. Beach monitoring data were obtained from Ministry of Natural Resources for the years 1993 to 2003 and were examined as an additional and independent source of beach quality information from the HCHU dataset. The degree to which levels of *E. coli* varied and were elevated over the 11 year period was examined and the patterns of variability were assessed to provide insight on the possible sources of fecal pollution to the shoreline.

The levels of *E. coli* detected in the beach monitoring program varied appreciably between years. The seasonal geometric means exceeded 100 cfu/100 mL in fewer than 30% of the sample sets for six of the 11 years examined (Figure 37). Only in 1996 did the results for >50% of the sample sets in a season exceed 100 cfu/100 mL. None of the sample sets collected in 2002 and 2003 had geometric means >100 cfu/100 mL. There appeared to be no obvious seasonality in the levels of *E. coli* over the June to early September monitoring period (Figure 37).

Some similarities did exist between high *E. coli* years at Point Farms Beach and high *E. coli* years at the HCHU beaches examined. The years when the frequency of *E. coli* above the PWQO was >30% (1995, 1996, 1998, 2000 & 2001) at the Point Farms Beach corresponded to similar high frequency (>50%) years at many of the HCHU beaches (see Table 5) with the possible exception of 1995. In 1995, only Goderich Main Beach had 50% of sample sets with geometric means >100 cfu/100mL.

The detailed results for the years 1996 and 2001, the years with the highest levels of *E. coli* as inferred from annual medians, are given in Figure 38 along with selected environmental information. The flow in the lower Maitland River is relevant to Point Farms Beach in two ways. Firstly, the mixing plume from the Maitland River in Lake Huron frequently lies along the northern shoreline of river mouth and moves northward in the direction of Point Farms. Secondly, changes in the flow of the lower river will likely be correlated with the degree of water movement from the land to the lake and serves as an indicator of the extent of land-lake interactions.

The MNR dataset confirms the previous observations with the HCHU dataset that periods of wet weather appear to coincide with some of the times when elevated levels of *E. coli* were detected. However, there are other occasions when elevated bacterial levels seemingly occur during dry weather. For example in 1996, there appeared to be a degree of correspondence between wet weather and most samples of >100 cfu/100mL, this is in contrast to data from mid summer 2001, when several samples sets had elevated levels of *E. coli* seemingly in a period of dry weather (Figure 38).

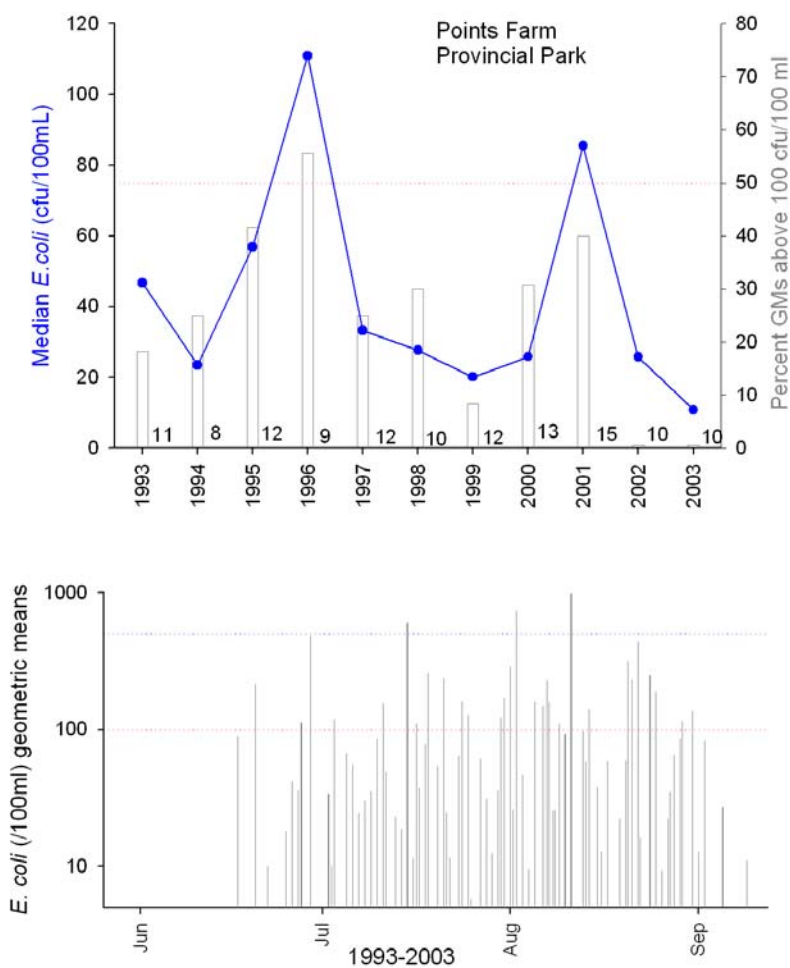


Figure 37: Annual median of geometric means for Point Farms Provincial Park Beach for the years 1993 to 2003 (blue circle) and percentage of sample sets where the geometric mean was above 100 cfu/100ml (bars) are shown in the upper panel. The number of sample sets in a year is shown by the numbers at the bottom of the bars. The lower panel shows the distribution of geometric means over the sampling season as a composite of all years.

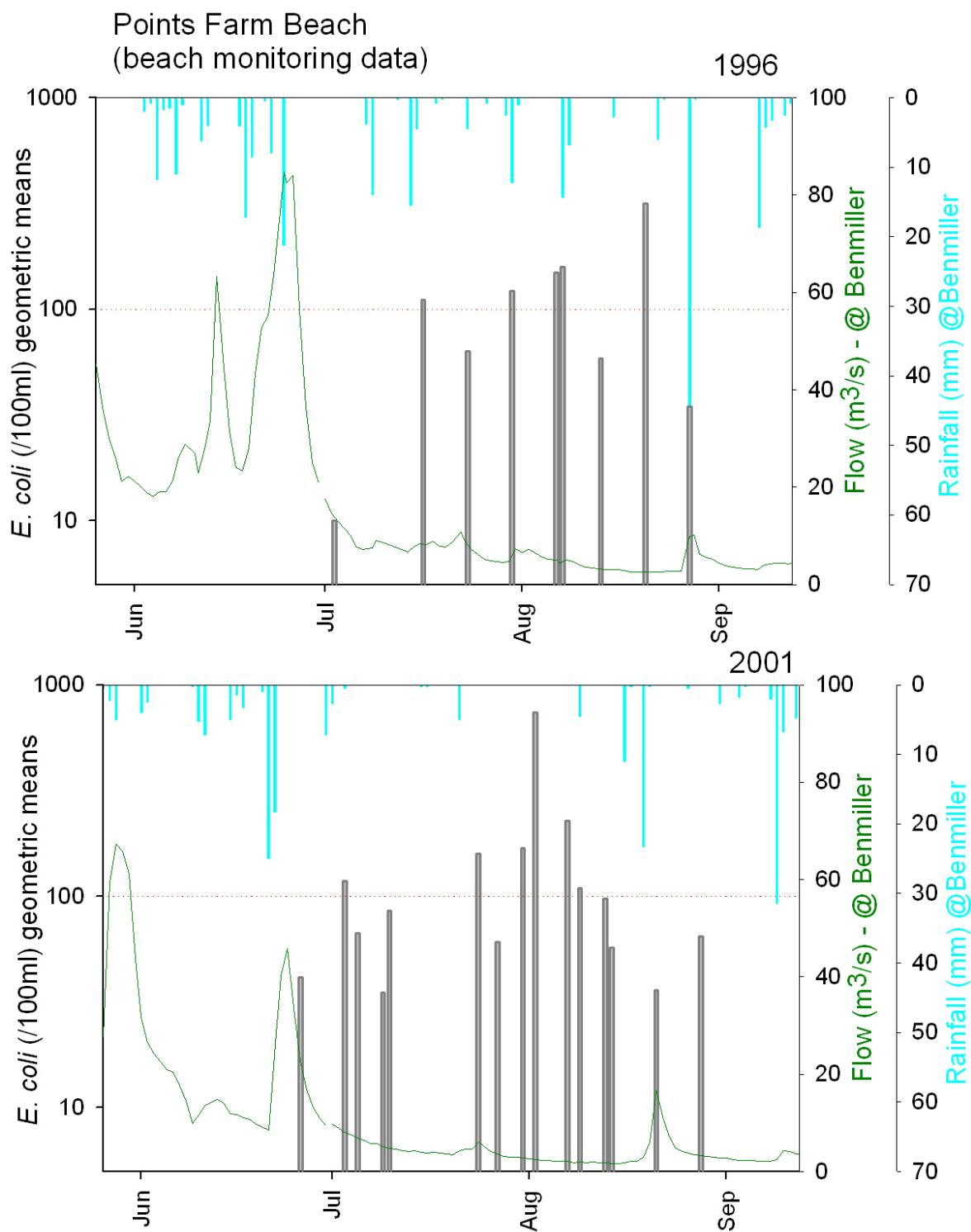


Figure 38: The geometric means of *E.coli* are plotted against time for the years 1996 and 2000 (grey bars) in the top and bottom panels, respectively. Daily rainfall over the period as measured at the Benmiller is given by the blue bars and the flow of the Maitland River at Benmiller is given by the green lines.

2.6 Recent Water Quality Monitoring by Ashfield-Colborne Lakefront Association

The residents of the Lake Huron shores of Huron County through associations of property owners and the local municipalities have undertaken water quality monitoring projects designed in part, to examine microbial pollution of surface waters draining to Lake Huron. The data collected through several of these efforts provide recent and valuable information on the range in levels of *E. coli* in surface waters draining to the lake, particularly for small water courses for which recent information is generally lacking. Selected data from the work of Ashfield-Colborne Lakefront Association and the Township of Huron-Kinloss are presented and discussed in the next two sections. The Blue Water Rate Payers Association also maintains a water quality program and monitoring information, available from their Web site, was examined. However, the locations of these monitoring sites were beyond the geographic area on which the analysis of recent data for this report will focus.

The Ashfield Colborne Lakefront Association (ACLA) is an organization composed of 22 cottage associations in Ashfield Colborne Wawanosh Township located between Goderich and Point Clark. The ACLA has monitored water quality in selected tributaries to Lake Huron since 2001, in response to frustration over ongoing beach posting in the area (ACLA 2003). The Maitland Valley Conservation Authority (MVCA) has collaborated with the ACLA on aspects of the design and interpretation of the ACLA monitoring (Steele 2002). Data and information collected by the ACLA and published on their website ([www. northwesthuron.com](http://www.northwesthuron.com)) for the years 2001, 2002 and 2003 was examined. A subset of the ACLA raw data was used in combination with data from other sources to examine occurrence of *E. coli* in tributaries as an indicator of fecal pollution.

The design of ACLA monitoring has varied among years. In 2001, a suite of downstream sites in tributaries (single site per tributary) were sampled for *E. coli* and selected water column nutrients over the spring to fall period. The objective was to compare streams and assess which ones were having the greatest impact on the lake. The next year, while fewer sites were surveyed multiple locations were sampled within the selected tributaries. In 2003, as in 2001 a range of sites were samples at downstream locations and the geographic range of sites was extended southward to include the Maitland River.

The 2001 to 2003 results for *E. coli* in tributary waters support the contention that levels of *E. coli* are frequently elevated in the waters of the many small to moderate size water courses to the lake. In 2001, the geometric mean of 10 to 15 samples from May to November ranged from just under 200 to approximately 800 cfu/100 mL (Figure 40). Many of the drainage areas of the tributaries are small and all are modest in size compared with the Maitland River (Figure 39). The data suggest that the highest levels of *E. coli* were observed in the smaller tributaries (Figure 40) which would be more sensitive to local influences, pollutant inputs during wet weather and stagnation during dry periods, than the larger tributaries. Attempts were made in 2001 to gauge tributaries and thereby examine the monitoring data as loads delivered to the lake. However, this effort was only partially successful because of several confounding factors (Steele 2002), including the observation that six of the tributaries were dry or did not flow in July and August. On average, site A8 on Nine Mile River, the largest tributary of the set, was estimated to contribute the largest load of *E. coli* to the lake (Steele 2002).

In 2001, *E. coli* levels were both highly variable among sites and over time. The results for Griffins Creek, one of the smaller tributaries is presented in Figure 40 to illustrate the range of variability observed over the sampling period. Levels in most samples ranged from just under 300 to just over 1300 cfu/100mL, however, the range of values was appreciably wider extending from under 100 to over 6000 cfu/100 mL.

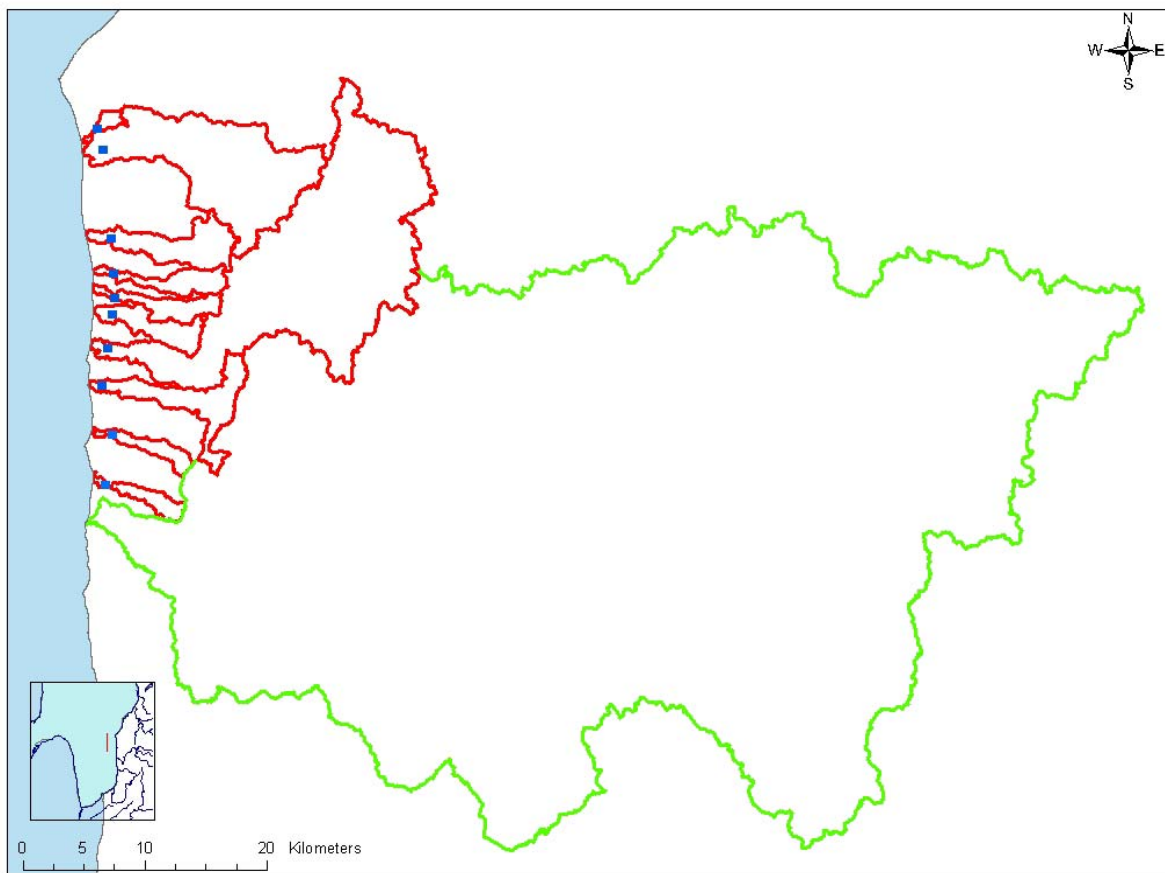


Figure 39: The approximate location of 10 of the 12 ACLA tributary monitoring sites (blue square) is given along with the outline of the drainage area for the tributary (redlines). The drainage area of the Maitland River is shown in green. Inset shows positions of shoreline relative to Lake Huron. In two cases, sites were on the main and side branches of the same tributaries, Eighteen Mile River and Kerry's Creek, respectively. The two sites on the side branches, A1 (Boyd Creek) and A3 (Kintail Creek), are not shown or included in Figure 40.

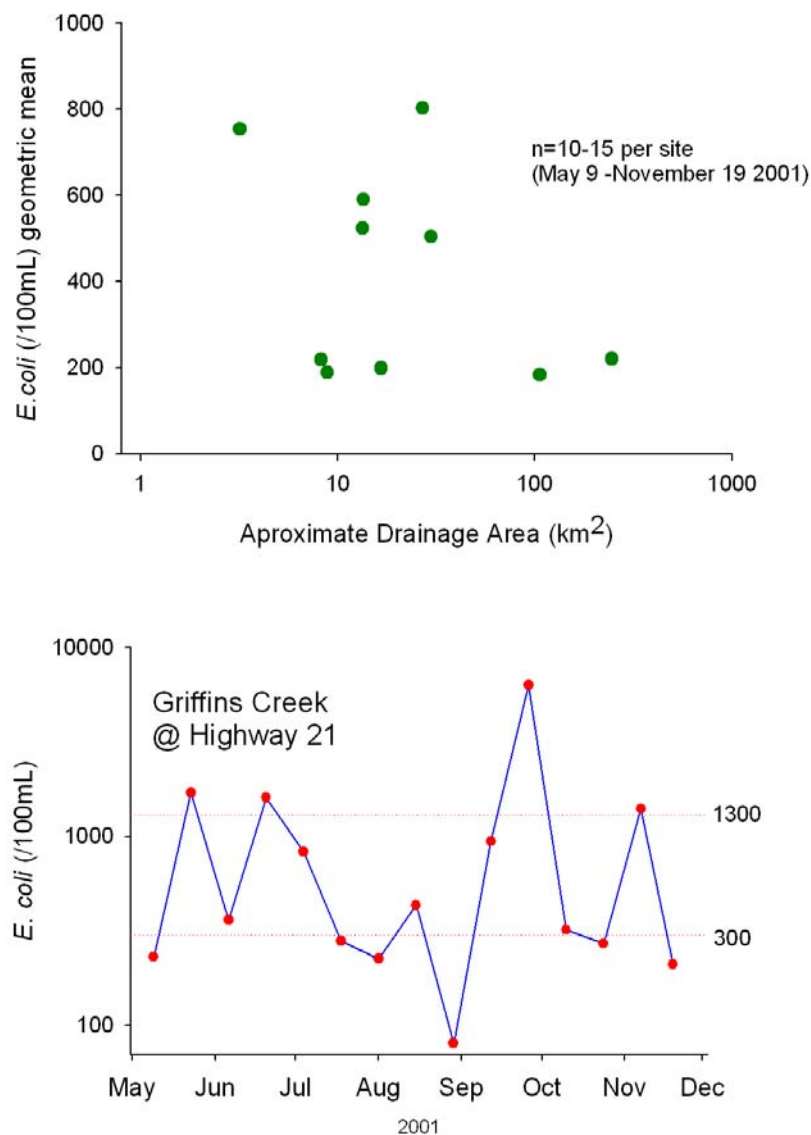


Figure 40: The geometric mean concentrations of *E. coli* in samples collected in 2001 at the 10 of 12 sites monitored by the ACLA is plotted against the drainage area of the tributary on which the site is located (Upper Panel). The levels of *E. coli* in single grab samples from the Griffins Creek site over the 2001 sampling period is shown in the lower panel. The drainage area of Griffins Creek is ~13.4 km². The raw data were taken from Steele 2002. The geometric means for sites A1 and A3 were 200 and 490 cfu/100mL, respectively.

In 2002 the ACLA monitored *E. coli* levels at sites in three tributaries, Nine Mile Creek, Kerrys Creek and Boundary Creek, over the spring to fall period. The objective was to locate specific stream sections of poor water quality in tributaries contributing the greatest *E. coli* load based on the 2001 monitoring data (summary of 2002 results- source unclear). The stations extended from the lakeshore (or just upstream of the lake) to approximately 14 km away and they bracketed most of the drainage areas of Kerrys Creek and Boundary Creek (Figure 41).



Figure 41: The locations of the tributary sites monitored by ACLA in 2002 (red square) are shown on maps of the area which illustrate selected physical and land-use features. The bottom panel shows the sites in relation to water courses and the drainage areas of the tributaries. The locations of dwellings in the drainage area are shown in the top left panel and land use (orange-agricultural; green-forested; blue-water) in the top right panel.

As evident from the 2001 data, elevated levels of *E. coli* were frequently detected in the grab samples of water collected in the tributaries to the lake, however, there was no clear upstream-downstream trend in geometric means of *E. coli* at any of the creeks (Figure 42). It is of note that the geometric means for the stations nearest to the lake were the lowest among stations and the results do not suggest that there is a progressive accumulation of a load of *E. coli* as water moves through the tributary. The levels of *E. coli* vary between the tributaries, with

geometric means and 75th percentiles at Boundary Creek varying over a wider range than those at sites in the Kerry's Creek and Nine Mile Creek. The difference between Boundary Creek and Kerry's Creek is intriguing, in that the nature of the lands draining both creeks appears to be similar (at least a coarse level). The drainage areas are similar and dominated by agricultural lands with limited population densities (as inferred from the limited number of buildings) (Figure 41).

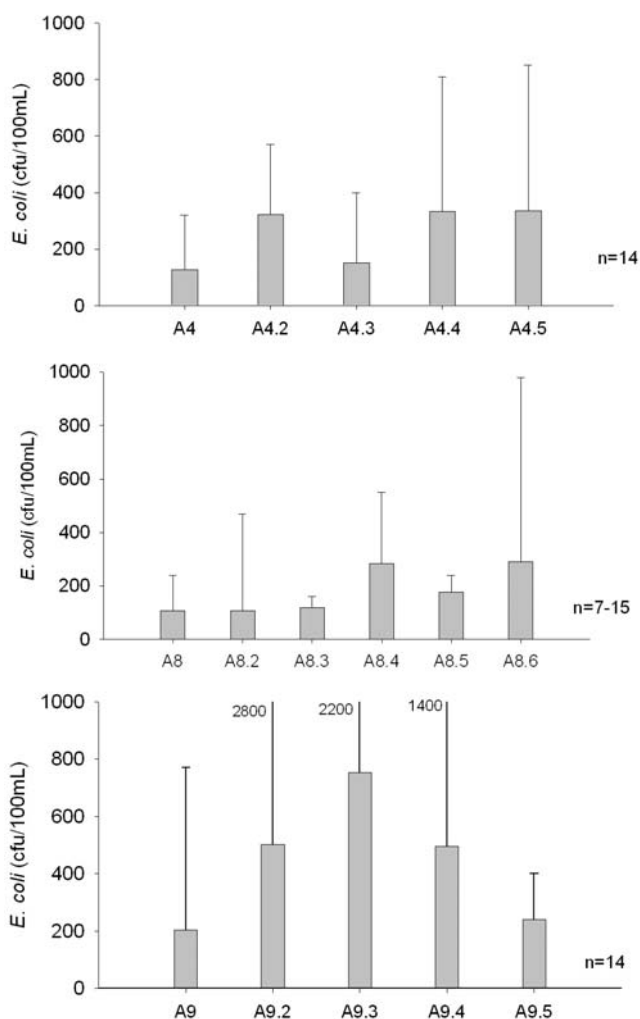


Figure 42: The geometric mean concentrations of *E. coli* (grey bar) and 75th percentile (line bar) for samples collected in 2002 at the sites in three tributaries monitored by the ACLA. The results for Kerry's Creek, Nine Mile River and Boundary Creek are shown in the top, middle, and bottom panels, respectively. The number of samples in the samples sets is given by n.

The mouths of several of the tributaries that ACLA has monitored are directly adjacent, or near, shoreline beaches monitored by the HCHU. The combined ACLA and HCHU data provides a means to explore possible connections between fluctuations in tributary and beach water

quality. The through-time monitoring data collected by ACLA at selected tributaries was plotted on a common time axis with the beach monitoring data collected by HCHU for selected beaches and tributaries. Missing from the information is any measure of tributary discharge and *E. coli* load to the lake. The ACLA results for site A8 (downstream site) on Nine Mile Creek and the HCHU data for Port Albert Beach for 2001, 2002 and 2003 are given in Figure 43. Also, shown are results for *E. coli* sampling at the mouth of Nine Mile Creek conducted by HCHU in 2003.

The limited nature of the data sharply limits the degree to which any tributary-beach interaction can be inferred. There are few occasions when both sets of data overlap directly in time in the years 2001 and 2002, and in 2003 when there is more correspondence among datasets, there were only a few occasions when beach *E. coli* levels were elevated (Figure 43). In the late summer of 2001, when geometric means of *E. coli* were consistently above 100 cfu/100 mL at Port Albert Beach, the levels of *E. coli* in Nine Mile River did not appear to be appreciably elevated and were in a similar range to beach samples (Figure 43). The more extensive dataset in 2003 does not provide clear evidence of a tributary-beach linkage. There are two occasions when river levels of *E. coli* were moderately elevated and corresponded to high beach *E. coli* geometric means (>100 cfu/100 mL), however, there were more instances when this was not the case.

Limited data was also available for two small tributaries monitored by ACLA that discharge to Lake Huron in proximity to beaches monitored by HCHU. Bogies Road Creek discharges to the north edge of Bogies Beach and Allan's Creek discharges a short distance north of Sunset Beach (Figure 44). In 2003, moderately elevated levels of *E. coli* were periodically detected in Bogies Road Creek, however, levels of *E. coli* at the beach over the monitoring period by HCHU remained low (Figure 45). Unfortunately, the highest levels in the creek, up to 42,000 cfu/100 mL, were detected in the fall after the beach monitoring had ended for the season. The comparison of data between Allan's Creek and Sunset Beach for 2003 was similar to that for Bogies Beach and Allan's Creek. Beach results at Sunset Beach in 2001 were more variable than in 2003 but limited data precluded the identification of any relationship (or lack of a relationship) between poor tributary quality and a corresponding effect on beach quality (Figure 45).

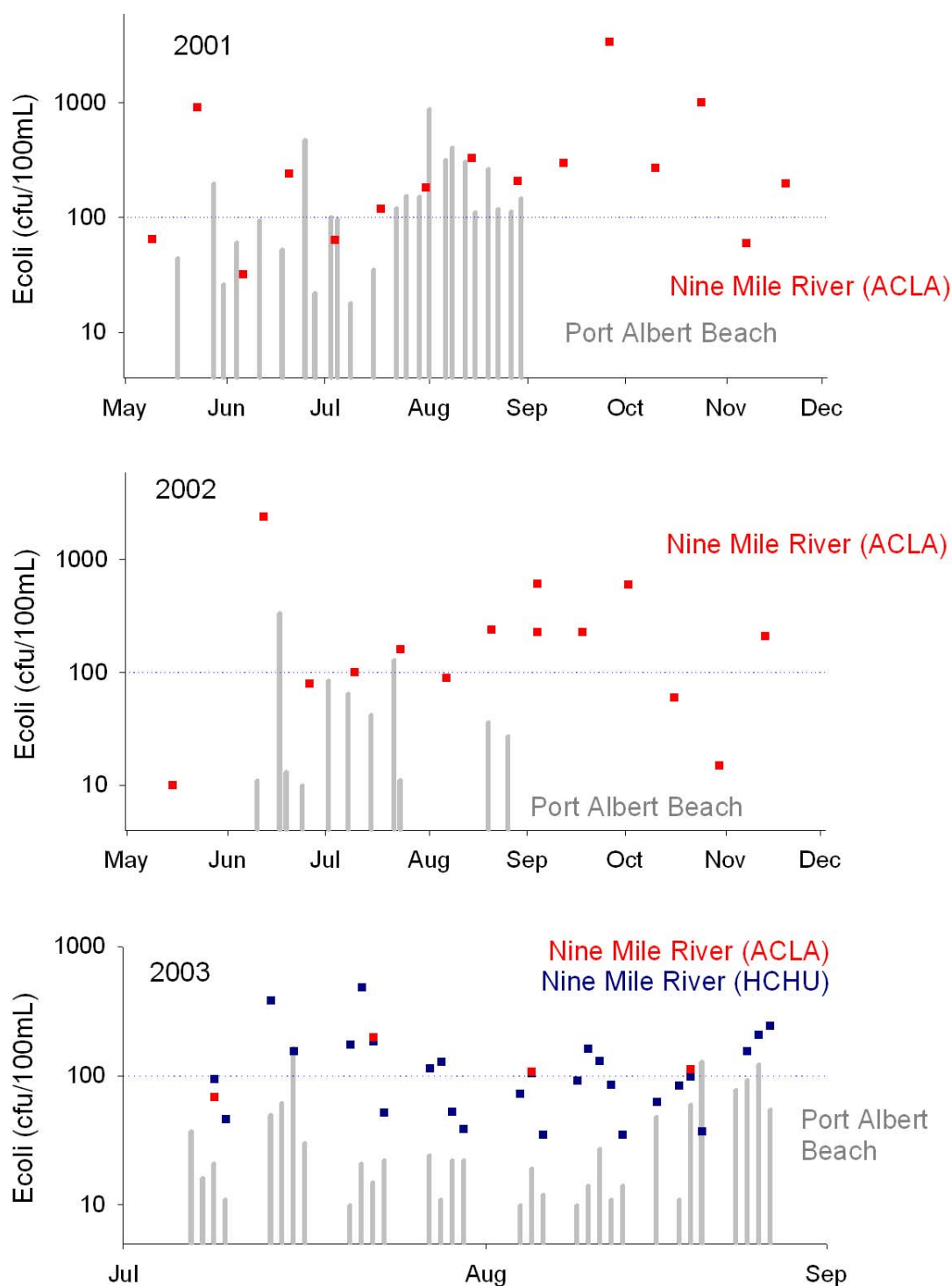


Figure 43: Geometric mean *E. coli* for samples sets collected at Port Albert Beach in 2001 to 2003 by HCHU plotted over time (grey bars). The levels of *E. coli* in single water samples collected near the mouth of Nine Mile River in 2001 to 2003 by the Ashfield Colborne Lakefront Association (red squares) and in 2003 by HCHU (blue squares).



Figure 44: Locations of Bogies Road Creek and Allan's Creek in relationship to Bogies Beach and Sunset Beach.

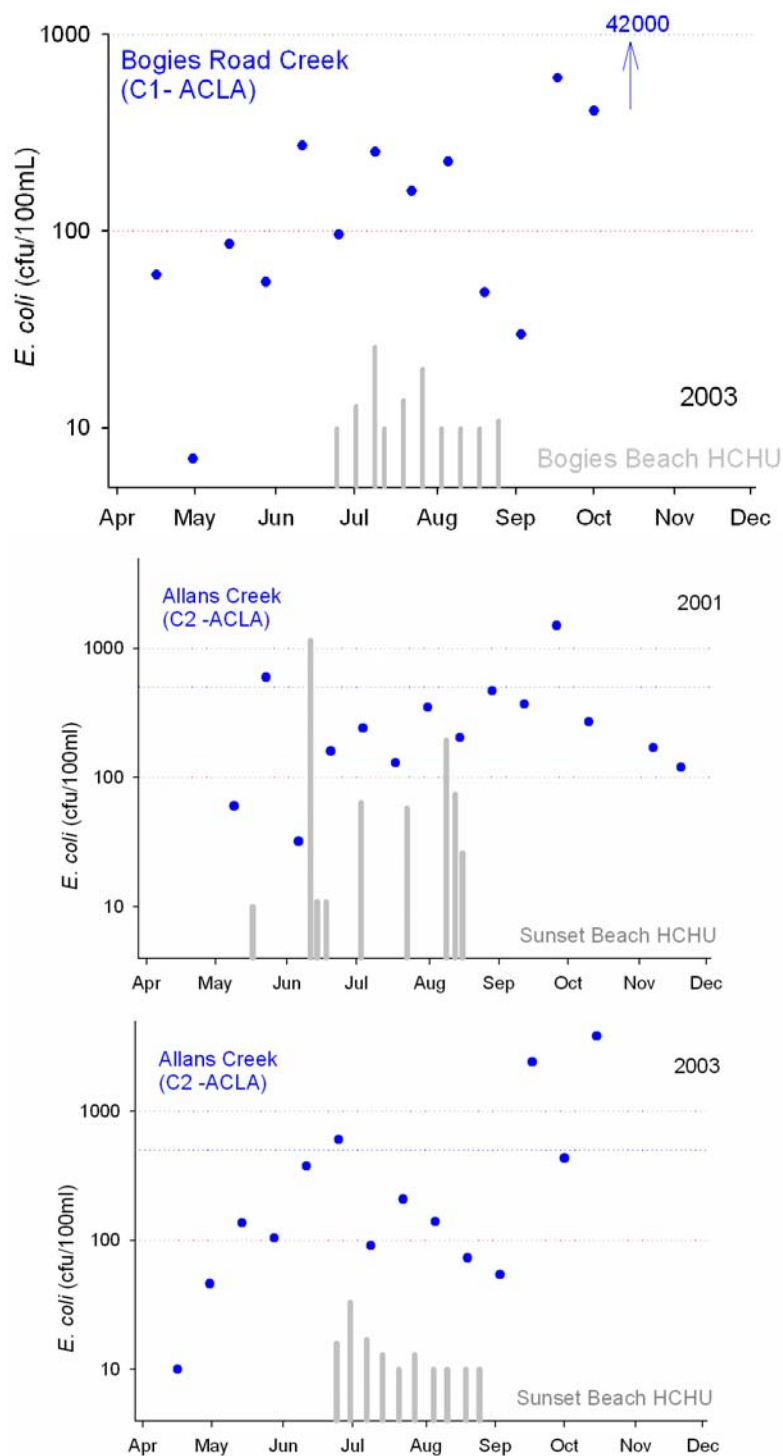


Figure 45: Geometric mean *E. coli* for samples sets collected at Bogies Beach in 2003 (bars-top panel), Sunset Beach in 2001 (bars-middle panel) and Sunset Beach in 2003 (bars-bottom panel) against time. ACLA data for *E. coli* levels in grab water samples from Bogies Road Creek in 2003 (circles-top panel), Allans Creek in 2001 (circles-middle panel) and 2003 (circles-bottom panel) against time.

A component of the 2003 monitoring by the Ashfield-Colborne Lakefront Association was a limited microbial source tracking (MST) study. In September 2003, samples were taken from Amberley Beach, Ashfield Township Park Beach, Port Albert Beach, Eighteen Mile River (at Highway #21), and Nine Mile River (at Highway #21). These samples were shipped to a private laboratory in Florida for MST analysis. The results were reported in a press release in October 2003 by the Ashfield Colborne Lakefront Association (ACLA). They indicated that the source of microbial contamination was of animal origin. However, there are technical and methodological questions which either do not appear to have been considered or are unclear in the reported information which may affect the interpretation of the data.

It is unclear from the ACLA information what was actually done with the samples. The ACLA report states that five 100 mL samples were split into 5 subsamples each and analysis was performed on all 25 samples. The laboratory provided fecal coliform results for 5 samples, and MST results for 5 *E. coli* isolates from each sample. It is more likely that the water samples were analyzed in their entirety for fecal coliforms and that MST analysis was performed on isolates obtained from the samples.

Secondly, it is unclear how the laboratory obtained isolated colonies for MST analysis. The MST technology used by this laboratory was ribotyping. This method involves generating DNA fingerprints using ribosomal genes from *E. coli* isolates, and comparing them to a library or database of DNA fingerprints of *E. coli* isolates from a variety of fecal sources. Accordingly, this technology requires both the isolation of colonies and their subsequent identification to determine which colonies are *E. coli*. The laboratory reported fecal coliform results as “MPN/100 mL”. MPN, or Most Probable Number, is an approach for statistically estimating the true number of bacteria in a sample using broth cultures, rather than solid media that yield isolated colonies which are counted and reported as CFU or Colony Forming Units. The laboratory provided a description of how *E. coli* was identified from fecal coliform colonies on membrane filters on a solid media. According to the laboratory report, the isolated colonies appeared to come from the fecal coliforms that were analyzed by MPN.

There are two possible explanations for this discrepancy. Since isolated colonies are not obtained using MPN methods, additional selective incubation steps may have been performed to transfer broth cultures to plates. Alternatively, the report may incorrectly reflect the methodology that was actually performed. From the reported results, it is unknown how the laboratory obtained *E. coli* isolates from the samples, and whether there was a selective or

enrichment step that may have biased the type of bacteria able to grow under the conditions provided. Consequently, the isolates analyzed by ribotyping may not have been representative of the original water sample.

For any MST method, including ribotyping, adequate representation from each water sample should be evaluated to provide confidence in source classification of waterborne isolates. No MST method correctly classifies isolates 100% of the time, regardless of the quality of the library. It is recommended that 200 isolates obtained from water samples be compared to a source library (Health Canada and Agriculture and Agri-Food Canada 2003). The number of fecal coliforms found in this study ranged from 150 to 1,100 MPN/100 mL. Therefore, 3.3% to 0.5% of the bacteria in the original water samples were analyzed by ribotyping 5 isolates to identify sources.

Finally, the ribotyping method involves a comparison of DNA fingerprints of waterborne *E. coli* isolates to a library or database of DNA fingerprints from a variety of sources. DNA fingerprint libraries need to be fairly large to be representative. It has been estimated that libraries of 20,000 to 40,000 isolates are required to capture the genetic diversity present in *E. coli* (Johnson et al. 2004). Since this is too costly and time consuming to achieve this level of library diversity, a more realistic library of at least 2000 isolates with no less than 500 isolates per source should be used (Microbial Source Tracking (MST) Workshop, Summary of Proceedings, 2003). On a broad scale basis, new isolates have a relatively small chance of being identified by most libraries. To compensate for this limitation, it has been suggested that moderate sized libraries be comprised of isolates taken from sources in smaller, highly confined, geographic areas and used for MST studies in that area only (Johnson et al. 2004). Additionally, the proportion of library isolates from different sources should be representative of the study area.

The size of the library used by the Florida laboratory at the time of the analysis was not reported. However, a study was published by their Senior Research Scientist which described the ability of a library of 3,000 DNA fingerprints generated by the same ribotyping methodology to correctly differentiate between human and animal *E. coli* isolates obtained from various geographic regions in Florida. The library misclassified 21.4% of isolates from nonhuman sources (beef and dairy cattle, poultry and swine), and 15.5% of isolates from human sources (Scott et al. 2003). In the ACLA study, isolates obtained from the southeast shore of Lake Huron were compared to a library generated in the State of Florida. None of the source library isolates were obtained from the study area. The percentage of misclassification is unknown in

the absence of a detailed study, but is likely to be higher than what was reported for the State of Florida.

The credibility of these results is compromised by the uncertainty regarding the cultivation methodology, the MST library and the reporting. These results should be interpreted with caution. Given the extent of agriculture in the region from which the samples came, it is likely that much of the microbial pollution of fecal origin is from animal sources, however, these results neither support this, nor do they eliminate the possibility of additional human sources.

2.7 Recent Water Quality Monitoring by the Township of Huron-Kinloss

The Township of Huron-Kinloss (Bruce County) located on the shores of Lake Huron at the northern edge of Huron County has conducted water quality monitoring programs in support of its planning process and management of environmental concerns. Since 2001 a number of tributary and shoreline locations at the north end of Huron County have been sampled for *E. coli* and selected nutrients (B.M. Ross and Associates 2004a, 2004b). Some of the tributaries monitored by the Township of Huron-Kinloss discharge to Lake Huron within Huron County.

Eighteen Mile River which discharges at the lakeshore adjacent to Ashfield Township Park Beach is one of the tributaries which have been monitored by the Township of Huron-Kinloss (see B.M. Ross and Associates 2004a). Monitoring was conducted over the 2002 and 2003 spring to fall periods at eight sites spread over the tributary (See Figure 46). Three of the sites were located on Boyd Creek, a branch of the main river. Elevated levels of *E. coli* were periodically detected at most of the tributary sites. Geometric means (calculated from raw data in B.M. Ross and Associates 2004a) over the 2002 and 2003 sample set ranged from <100 to >500 cfu/100 mL between sites (Figure 47). The wide 75th percentile bars at most sites indicates that there are wide fluctuations in *E. coli* levels overtime. Overall, the geometric means suggest a trend of decreasing levels of *E. coli* downstream in both the main portion of Eighteen Mile River and Boyd Creek.

Among the tributary sites monitored by the Township of Huron-Kinloss is a small creek immediately north of Eighteen Mile River (see site PR16- Figure 46). Data collected at this site from April to November 2002 and 2003 and June to November 2001 illustrate the difficulty in

assessing the impact of small watercourses on the shoreline (Figure 48). Levels of *E. coli* fluctuated from under 100 cfu/100 mL to approaching 10,000 cfu/100mL (B.M. Ross and Associates 2004a). During the June to August bathing season, levels may be appreciably elevated, however, the volume of water in the tributary and the delivery of water to the lake is likely to decline to low levels. There is little data for PR16 in August and September because the creek was dry or had no flow (Figure 48).

Since Eighteen Mile River is directly adjacent to a beach monitored by the HCHU, the combined Huron-Kinloss and HCHU data was examined to explore possible connections between fluctuations in tributary and beach water quality. The 2002 and 2003 data for sites PR30 and PR27, the downstream sites in the main branch of Eighteen Mile River and Boyd Creek, respectively, are plotted on a common time axis with the HCHU beach monitoring data for the Ashfield Township Park Beach in Figure 49. The tributaries in the areas of PR30 and PR27 were also sampled by the ACLA in 2003 and these data are also provided in Figure 49. The levels of *E. coli* at PR30 and PR27 were frequently elevated over the summer of 2003, with a maximum value of over 1000 at PR30 observed in mid August. Frequent beach sampling in 2003 by HCHU indicated levels of *E. coli* above 100 cfu/100 mL in mid August and for a short period in early to mid July. There was no obvious correspondence between *E. coli* levels in the tributaries and at the beach. Unfortunately, only limited data is available for 2002 with too few instances with both tributary and beach data to justify comment.



Figure 46: The locations of the tributary sites monitored by the Township of Huron-Kinloss on Eighteen Mile River and a small creek north of the river (red squares) are shown on maps of the area which illustrate selected physical and land-use features. The top panel shows the sites in relation to water courses and the drainage areas of the tributaries. The locations of dwellings in the drainage area are shown in the bottom left panel and land use (orange-agricultural; green-forested; blue-water) in the bottom right panel.

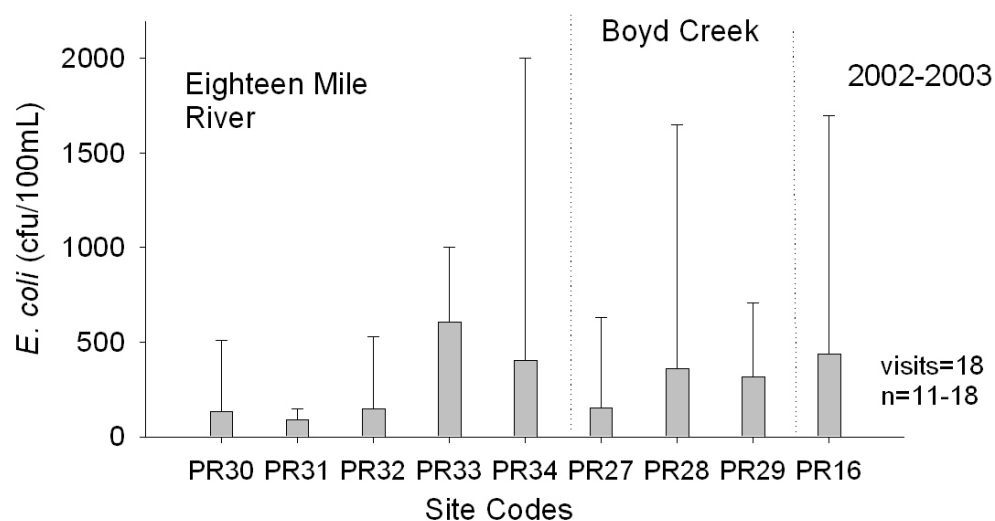


Figure 47: The geometric mean of *E. coli* (grey bar) and 75th percentile (line bar) for 2002 and 2003 at the sites in Eighteen Mile River monitored by the Township of Huron-Kinloss. The number of samples in the samples sets is given by n.

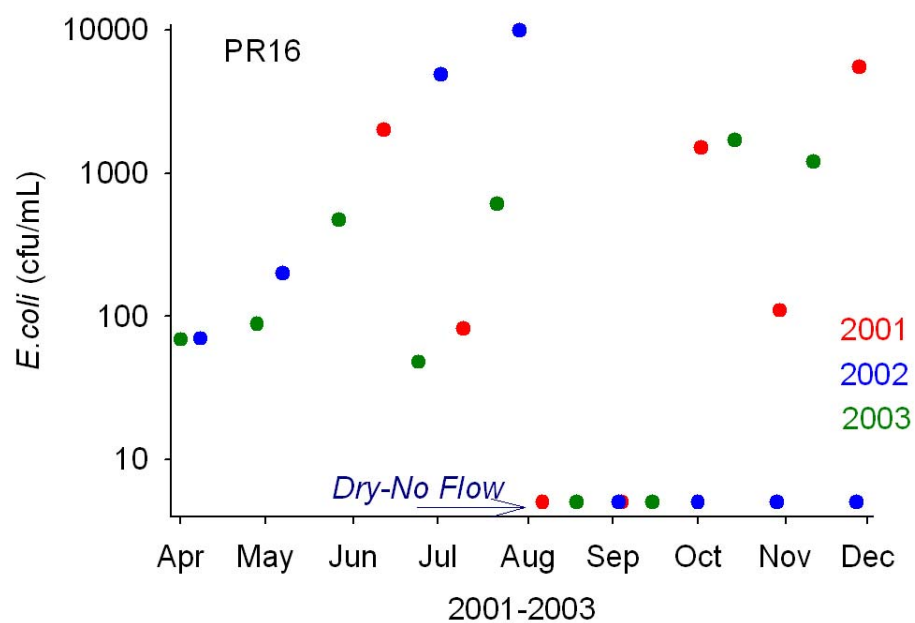


Figure 48: *E. coli* in single water samples collected at site PR16, a small creek north of Eighteen Mile River in 2001 to 2003 are plotted against time of year. The data were collected by the Township of Huron-Kinloss.

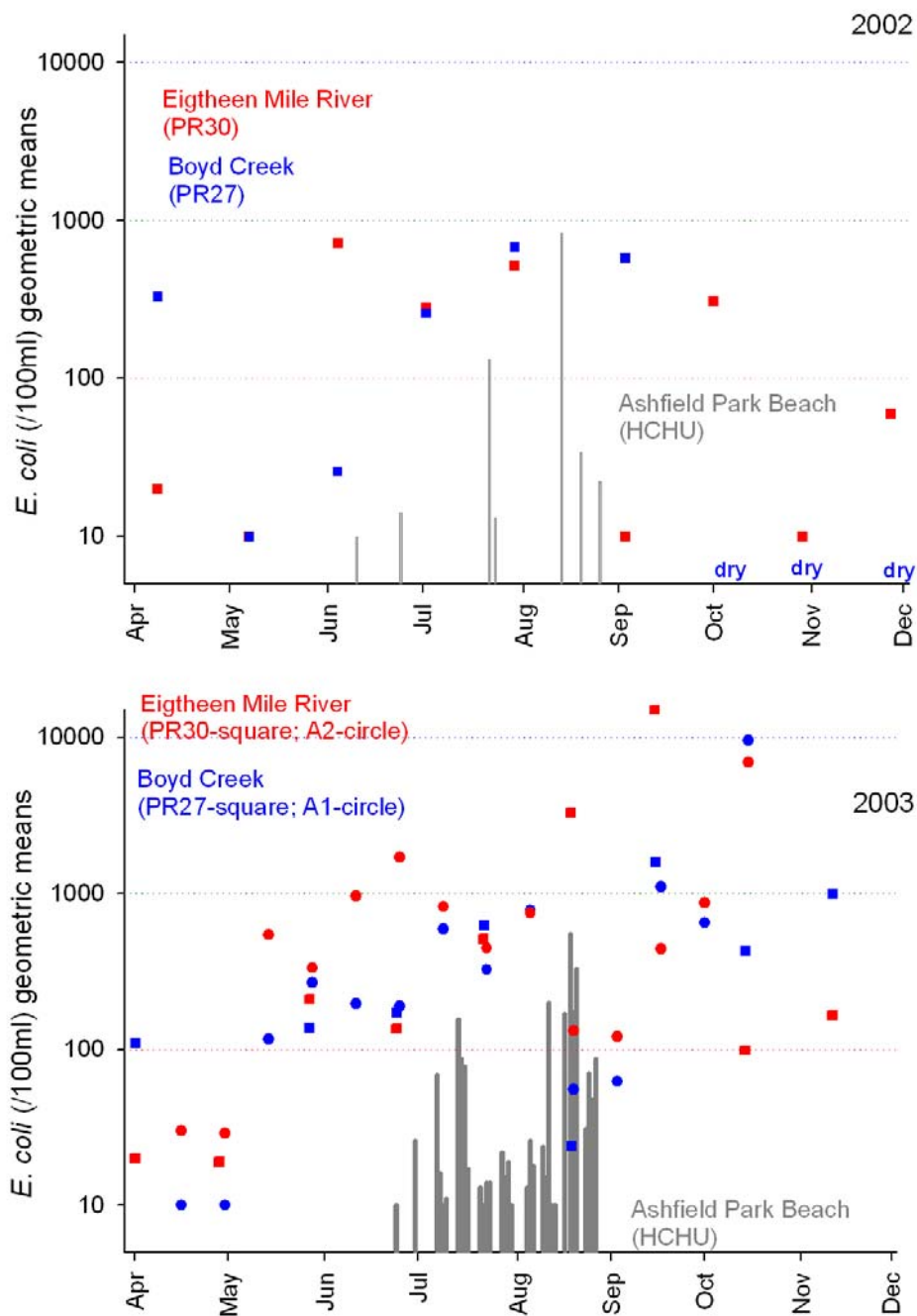


Figure 49: Geometric mean *E. coli* at Ashfield Township Beach in 2002 and 2003 (HCHU data) are plotted against time (grey bars) in the upper and lower panels, respectively. The levels of *E. coli* in single water samples collected near the mouth of Boyd Creek (blue symbols) and main branch of Eighteen Mile River (red symbols) collected in 2002 and 2003 by the Township of Huron-Kinloss (squares) and Ashfield Colborne Lakefront Association (circles) are also shown.

2.8 Recent Monitoring of *E. coli* in the Bayfield River by the Ausable Bayfield Conservation Authority

The water quality program at the Ausable-Bayfield Conservation Authority (ABCA) is partially self-directed and self-funded and also a part of the Provincial Water Quality Monitoring Network (PWQMN). The ABCA currently takes water samples from 18 sites within its jurisdiction on a monthly basis during the open-water season. Nine of the sites are part of the PWQMN. Approximately 40 inorganic chemical indicators are analyzed from these samples. The ABCA funds the microbial (*E. coli*) analyses of the water from these PWQMN sites. Analysis of water from the remaining nine sites within its jurisdiction is for nutrients (nitrogen and phosphorus), total suspended solids and *E. coli*. There are five water quality sites within the Lake Huron Science Committee study area. The sites are in the Bayfield River watershed and represent both main channel (PWQMN sites) and headwater locations (ABCA sites) (Figure 50).

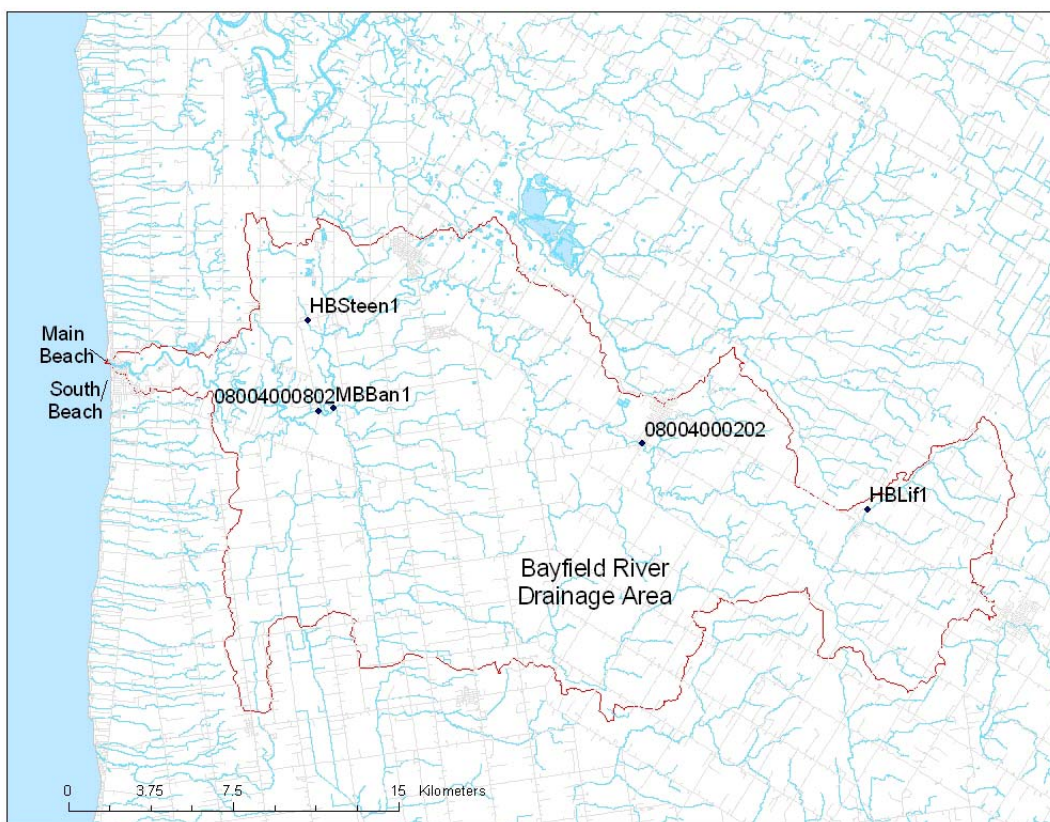


Figure 50: Water Quality monitoring sites on the Bayfield River surveyed in 2003 by Ausable-Bayfield Conservation Authority.

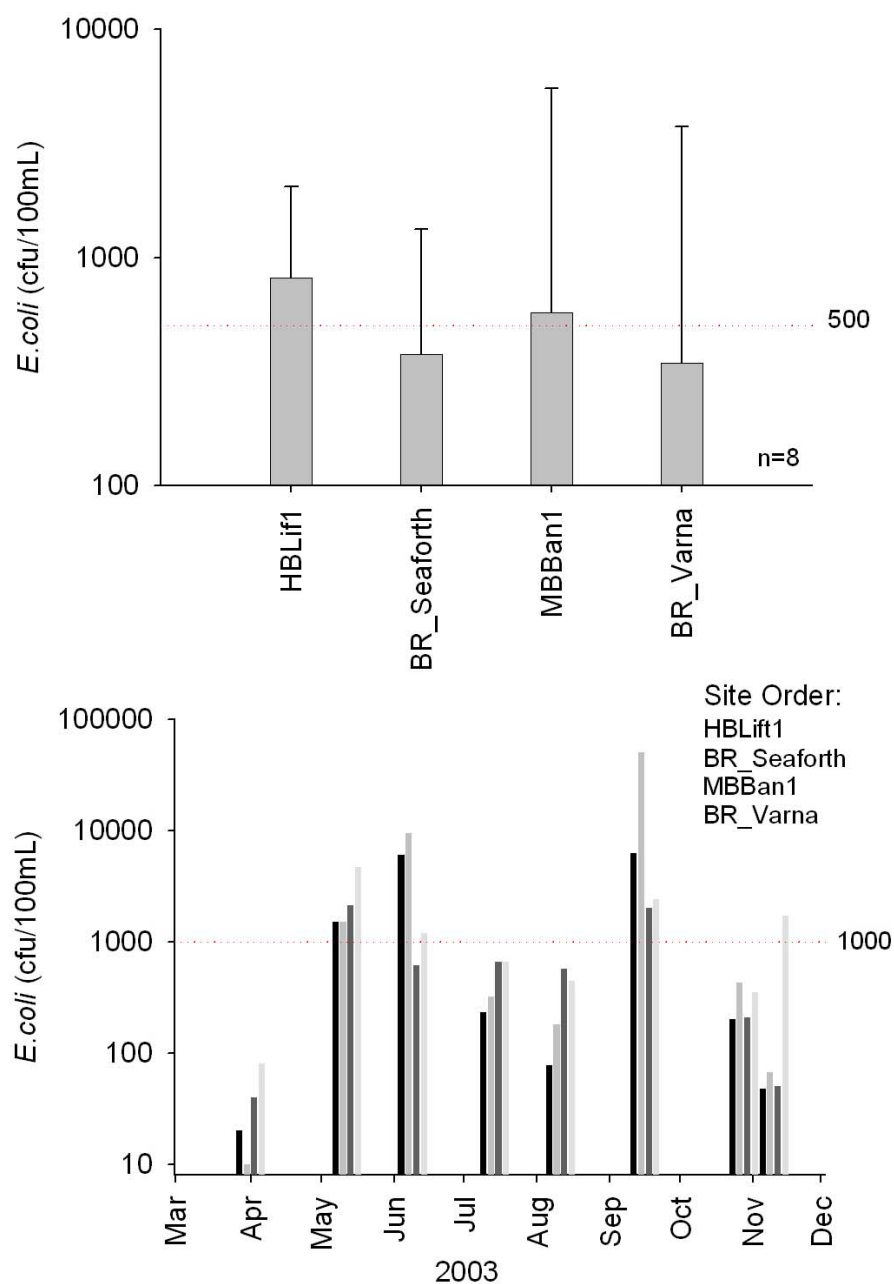


Figure 51: The Upper panel provides the geometric mean levels of *E. coli* at four sites on the Bayfield River over April to November 2003. The bars indicate the 75th percentile. The lower panel provide the individual sampling results. Station BR_Seaforth and BR_Varna are identified as stations 08004000202 and 08004000802 in Figure 50, respectively.

The *E. coli* data collected for the 2003 monitoring period for sites within the Bayfield River drainage area are presented in Figure 51.

There does not appear to be any consistent upstream-downstream trend in the seasonal geometric means between the stations (Figure 51). The geometric means are appreciably

above 100 cfu/100 mL and in excess of 500 cfu/100 mL at the headwater (HBLift1) and downstream main branch (MBBan1) sites. There were appreciable fluctuations in levels at all sites over the monitoring period but no apparent seasonal pattern. Levels were above 1000 cfu/100 mL at all sites in September samples. On the other hand, lowest levels, all under 100 cfu/100 mL, were observed in the April samples. Fluctuations in levels over time appear to be correlated between sites suggesting that system-wide factors (e.g. weather) may dominate over local conditions in determining the timing of loadings of bacteria to the waters of the tributary. Overall, the data indicate that levels of *E. coli* are frequently elevated, sometimes appreciably so, in the Bayfield River.

2.9 University of Guelph - Huron County Surface Water Quality Studies

Beach water quality in Huron County, as a function of rural and urban land-use features as well as human population density, has been examined in two related studies by University of Guelph researchers (Bonte-Gelok 2001; Bonte-Gelok and Joy 1999). Coliform data (total coliform, FC and *E. coli*) were obtained from a variety of sources including the Provincial Water Quality Monitoring Network (PWQMN), wastewater treatment plants (WWTP), landfills, provincial parks, and the local health unit for Huron County. Data from CURB reports, Ontario Ministry of Agriculture, Food and Rural Affairs (OMAFRA) tile and well water studies, as well as the ABCA target sub-basin studies were used in the analysis.

Huron County was divided into 13 sub-basins for the studies, each of which were ranked by their soil drainage class and their human and livestock (poultry, cattle and swine) population densities. Septic system densities for the sub-basins were calculated from sub-basin Census Canada data for human population not serviced by a WWTP and they were ranked. Qualitative (poor, imperfect and well drained) soil drainage classification was performed for each sub-basin. Interestingly, those sub-basins bordering and draining directly into Lake Huron were frequently unranked, presumably due to a lack of sufficient data. Correspondence between fecal coliform and land-use and physical features of sub-basins was statistically analyzed. Relationships between human, swine, cattle and poultry densities, soil drainage class and stream discharge to levels of fecal coliform bacteria (and other water quality features) at selected PWQMN sites within the sub-basins were evaluated using statistical approaches including rank correlation, multiple regression and principal component analysis (PCA). Fecal coliform bacteria were

correlated with human population density and soil drainage class using all three statistical approaches. However, the study acknowledged that other landform qualities needed to be considered (i.e. land slope, depth of overburden, number and proximity of sources to local streams and creeks) for a complete evaluation. Total phosphorus water concentrations were often correlated with human population density and soil drainage classification in addition to stream discharge. Nitrate concentrations were found to correlate strongly with soil drainage class as well as swine and poultry densities.

The spatial association between watershed water quality and the frequency of events where *E. coli* levels were above the PWQO at recreational beaches associated with these watersheds was perplexing. Frequency of elevated levels of bacteria at beaches was highest in the northern section of the county yet the poorest water quality in tributaries occurred in the southern portion of the county. It was noted that the Maitland Valley watershed had the three inland and three beach monitoring sites that exceeded the PWQO the most frequently, yet Provincial Water Quality Monitoring Stations with the highest concentrations of coliforms occurred in the Ausable-Bayfield watershed.

One objective of the studies was to assess the “completeness” of the collected data which covers a span of 25 years. The studies listed key data gaps, which included missing coliform and flow data from PWQMN, WWTPs and landfills, and described them as being a large hindrance to the level of statistical analysis that was possible,

2.10 Lake Huron Nearshore Water Quality Report by Lake Huron Centre for Coastal Conservation

In June 2004, the Lake Huron Centre for Coastal Conservation released a report reviewing the historical quality of Lake Huron’s nearshore waters from Sarnia to Sauble Beach (Lake Huron Centre for Coastal Conservation 2004). It examined water and beach quality data, particularly as it pertained to nutrients and microbial pollution, between 1984 and 2003. The information and data for the review were obtained from reports, studies and monitoring programs conducted by a variety of groups from the study area. These groups include local Health Units, area First Nations, community groups, conservation authorities, the Ontario Ministry of Natural Resources

and the Ontario Ministry of the Environment. This study was commissioned by Environment Canada.

The report found gaps in the spatial and temporal coverage of the available data, making a detailed historical assessment difficult. What was apparent from the data examined was that microbial pollution was a widespread and persistent problem on the southeastern shores of Lake Huron and that nutrient enrichment appeared to be increasing. It concluded that the data needed to be re-evaluated in terms of its overall completeness, and that improved organization and cooperation between agencies and groups conducting monitoring programs would be beneficial for the future. The study also found basic information on septic systems in the study area to be limited and acknowledged that a comprehensive assessment of their potential impact on water quality is required.

The Lake Huron Centre for Coastal Conservation report provides a valuable synopsis of water quality information and issues and it has been used extensively to identify sources of information for the current report.

2.11 NWRI Nearshore and Beach Water Quality Study at selected sites on the SE shores of Lake Huron in 2003

In 2003 staff from Environment Canada's National Water Research Institute conducted an exploratory study of the levels of *E.coli* occurring in selected physical components of the shoreline environment of Lake Huron at sites ranging from Goderich to Kincardine. The levels of *E.coli* in water samples collected across transects extending from the nearshore waters of the lake, through the surf zone, and terminating with interstitial waters at the beach edge were examined in five areas on a single occasion between July and October (Milne and Charlton 2004). It should be noted that the boat-based sampling in the Lake Huron nearshore waters was not concurrent to sample collections in the surf zone and on the beach. Additional samples were also collected at the mouths of several rivers in the same areas where the shoreline transects were evaluated. The work was part of larger study including study areas in Lake

Ontario and Hamilton Harbour. The enumeration of *E.coli* was by a MPN approach using Coliplates™.

A pattern observed at several of the Lake Huron shoreline sites was that levels of *E.coli* were typically elevated and highest in the pore waters collected in shallow pits in the beach sand below the swash zone (<3 m from shoreline) compared to lake water samples. The levels of *E.coli* in the open water of the nearshore were usually low and, at a limited number of sites, increased slightly in the shallow water directly adjacent to the beach but never notably above 100 mpn/100 mL. Elevated levels of *E.coli* were observed in water collected at the mouths of several creeks and rivers. However, the authors note that given the low tributary discharge at the time of sampling, it was difficult to attribute the instances of elevated bacteria at the shoreline to the adjacent tributaries. The authors speculated on the role of the beach and beach sand as storage areas for *E.coli* and their possible role as a vector of lake water contamination at the shoreline. Considerable spatial variability was noted in the levels of *E.coli* in pore waters collected at multiple points along a beach.

Amberly beach was one of the sites visited, however, the authors noted concerns with the possible contamination of the August samples and bad weather prohibited sample collection of that pore water. In October, levels of *E.coli* in pore water samples were <100 mpn/100 mL. Transects sampled at Goderich Cove Beach (July) and Goderich Main Beach (October) showed similar patterns in variability across the gradient. Elevated levels of *E.coli* (up to >2424 mpn/100 mL) were observed in some of the pore water samples. Water collected in the surf zone one meter from the beach edge had a lower range in *E.coli* levels than the pore water samples but some samples exceeded 100 mpn/100 mL. At 10m from shore, bacteria levels were below or slightly above 100 mpn/100 mL PWQO. Birds were observed on the beaches at Goderich and noted as a possible source of fecal material to the beach.

2.12 GAP Study of Microbial Pathogens in Shallow Beach Water in 2003

In a 2003 study, health risks associated with shallow lake water at the edges of beaches on the Great Lakes was investigated by a team led by Gary Palmateer of GAP EnvironMicrobial Services Inc (Palmateer et al. 2004). One of three shoreline beaches investigated was an

unnamed beach in SE Lake Huron. The occurrence of selected bacterial pathogens and indicators (*Escherichia coli*, *Salmonella sp.*, *Shigella sp.*, *Pseudomonas aeruginosa*, *Enterococci*, and *Staphylococcus aureus*) in shallow waters at the beach edges (termed interstitial in the report) was evaluated on multiple occasions. The antibiotic resistance patterns of bacteria isolated from the beaches was investigated and, in field experiments, the die-off rates of bacteria loaded to chambers and deployed in shallow waters were determined.

The Lake Huron sites were described as lacking any immediate discharge from urban or rural watercourses but were within several kilometers of a sewage treatment plant (STP). Seagulls were noted on the beach but beyond limited water chemistry, little background information on the shoreline was provided. It was also noted that the beach was posted as unsafe for swimming for the 2003 recreational season. The water samples for microbial analysis were collected at the edge of the beach on four separate days. A sterile pail was submerged at a depth of 10-20 cm to fill a sterile carboy with the water which was used to recover the target bacteria.

The levels of bacteria recovered from Lake Huron were thought to be relatively low compared to the levels sufficient to cause human infection; however, the data indicated a level of fecal pollution at the beach (Palmateer et al. 2004). The geometric mean concentrations of *E. coli* and *Enterococcus* were 139 and 84 cfu/100 mL, respectively, and the detection of *Salmonella sp.* and *Shigella sp.* confirmed that fecal pollution was contaminating the beach. The concentrations of the six target bacteria examined varied throughout the sampling period which the authors suggested might be due to fluctuating numbers of waterfowl, and variations in wave height. Levels of *E.coli* ranged from 10 to 620 cfu/100 mL among the six sampling periods.

Antibiotic resistance/susceptibility was determined for *E. coli* and *Salmonella sp.* using the Replicator System of Cathra Inc. (Steers et al. 1959). Fifty-three isolates of *E. coli* obtained from the large volume sampling of shallow lake water were tested for antibiotic/antimicrobial resistance to five compounds (ampicillin, carbodox, cephalexin, streptomycin and tylosin).

There were variable levels of resistance to the five compounds tested among *E. coli* isolates. A large number of isolates (25%) were resistant to the antimicrobial tylosin which the authors indicated is used as a feed-additive for swine. This was followed by a relatively common resistance to streptomycin and ampicillin. Both streptomycin and ampicillin are commonly used in animal husbandry and ampicillin is also used by physicians to treat human illnesses

(Palmateer et al. 2004). Multiple resistances were limited to three isolates exhibiting resistance to two compounds and two isolates resistant to three compounds.

Bacterial survival chambers were used to investigate the persistence of bacteria in the shallow beach water. Small volumes (20 mL) of non-sterile lake water were added to the survival chambers which were subsequently inoculated with 5 mL of a mixed target bacteria solution (10^8 cells per mL) and deployed in shallow water. Sampling typically occurred on a weekly basis. The only factor likely excluded from the survivability study was the impact of ultraviolet light as it could not pass through the Plexiglas chambers.

The time required for a 99.9% decrease in survival of the bacteria, or a three logarithm (log) reduction, from the starting concentration was used to describe die-off rate. *E. coli* exhibited a three log decrease in 11 days.

2.13 MOE Studies of Water Quality of the SE Shores of Lake Huron near Maitland and Bayfield Rivers in 2003

In 2003, detailed surveys of water quality were conducted along the shorelines of Lake Huron in the Bayfield, Goderich and Southampton areas, in addition to the lower portions of the major tributaries in these areas, namely the Bayfield, Maitland and Saugeen Rivers, respectively. This study was conducted by the Environmental Monitoring and Reporting Branch (EMRB) of the MOE. The purpose of the work was to characterize environmental conditions and provide an up-to-date baseline of information for an area where there was limited recent monitoring but was the subject of several concerns, including microbial pollution of the shoreline, nuisance algae and a perception of declining water quality in the nearshore. A more specific objective was to examine the impact of major rivers on the environmental conditions in the nearshore. It is known that water quality in tributaries in SE Lake Huron is poor in comparison to the oligotrophic open waters of Lake Huron and it is suspected that major tributaries are one of the primary routes in which land-based pollutants are delivered to the nearshore.

Water quality was evaluated in the nearshore and downstream portions of these tributaries over a spring to fall interval. At several points along the study area, high-resolution spatial surveys

were conducted to evaluate the zones of influence of the rivers on the nearshore of the lake. The intent of the study was to examine water quality in a broad sense and did not specifically target microbial pollution. However, the sampling regime did include the analysis of water samples for the presence of *E.coli*. The bacteria data was collected as an indicator of land-based influence on the nearshore and was to be used in concert with chemical indicators of water quality impacts. The data was not collected in a manner suitable for a direct comparison with human health guidelines (i.e. PWQO) nor was it collected with this purpose in mind. Collectively, the *E.coli* data provided insight on the levels at the downstream end of large tributaries and for areas of the lake impacted by river discharge, in addition to providing information on the background levels of *E.coli* in the nearshore waters of SE Lake Huron. The broader study results described the nature of the effect of the major rivers on the shoreline and demonstrated the contrast in water quality between the rivers and the lake providing concrete examples of the linkage between conditions in the rivers to the lakeshore. Selected results for the Goderich-Maitland River and Bayfield-Bayfield River Study areas are presented in this section with additional material provided in Appendix A. The results of the study are, at the time of this report, in analysis phase and are as yet unpublished.

The nearshore study sites in southeastern Lake Huron are referred to by the names of the adjacent rivers; the Maitland, and Bayfield Rivers (Figure 52). The Maitland and Bayfield Rivers flow through the towns of Goderich and Bayfield, respectively, in Huron County.

Goderich & Bayfield Study Areas - 2003

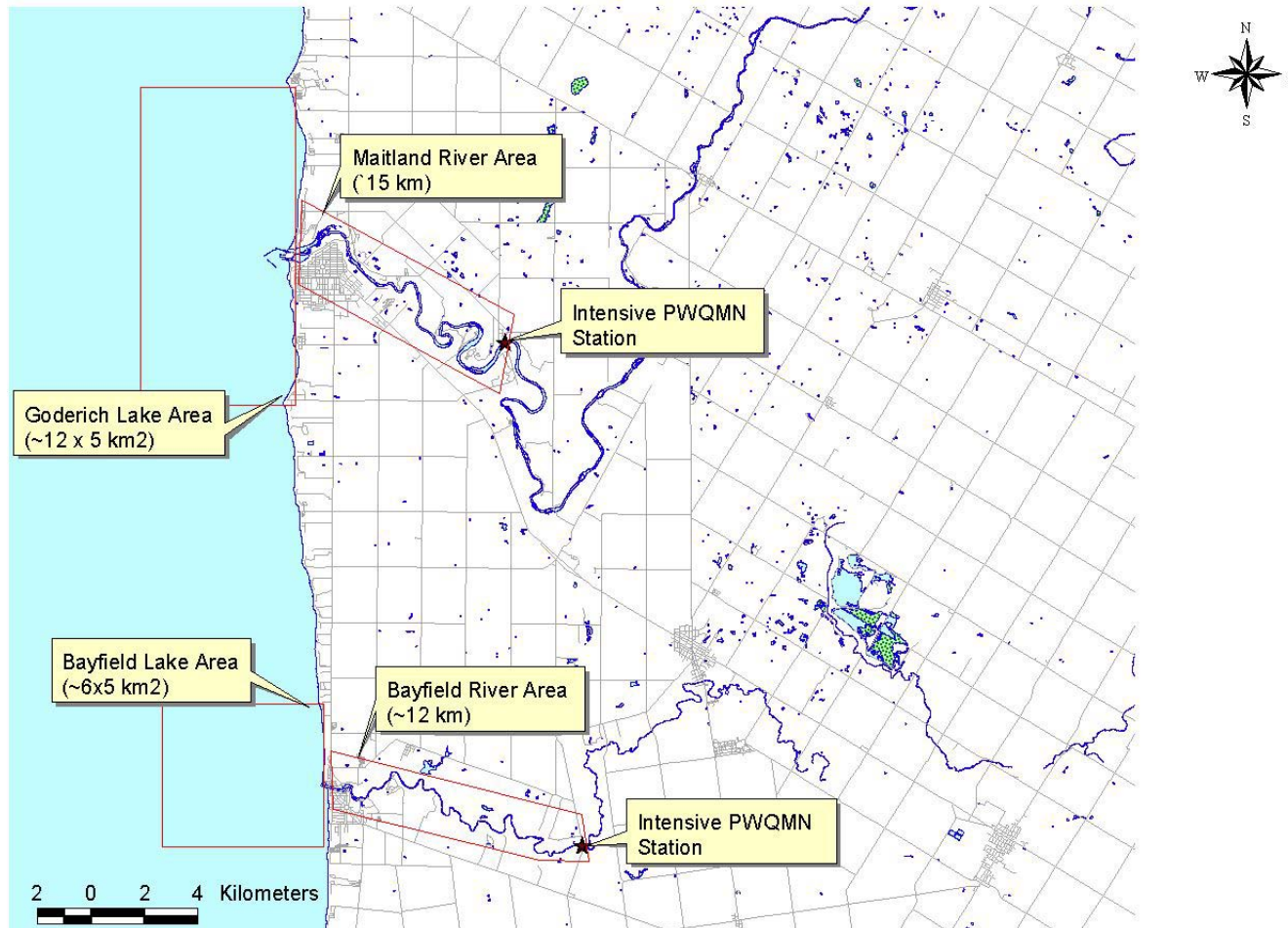


Figure 52: The Goderich-Maitland River and Bayfield-Bayfield River Study areas in southeastern Lake Huron for the 2003 nearshore water quality study.

Water quality at downstream sites on the Maitland and Bayfield Rivers, near Benmiller and Varna, respectively, was monitored from March 2003 to the end of the year. The sites are part of the PWQMN and river flows at these locations are gauged by Environment Canada. The levels of *E.coli* in river water fluctuated appreciably at both sites. In the lower Maitland River the levels of *E.coli* were under 300 cfu/100 mL in most samples with some exceptions (Figure 53). Samples collected in September and October had over 1000 cfu/100 mL and a sample collected in early June had just under 500 cfu/100 mL. Overall, levels of *E.coli* exceeded 100cfu/100 mL in eight of 18 samples. The frequency of sampling was not adequate to effectively capture transient wet weather events therefore the preponderance of short-term elevated levels of *E.coli* levels over the summer is uncertain. Samples were collected over a variety of stream discharge conditions including a number of high flow events. The response of *E.coli* levels to rising flows

appears to be inconsistent with the observed greater response in the fall than spring. The concentration in suspended solids (SS) varied over two orders of magnitude (ca. 1 to >100 mg/L) and appeared well correlated with *E. coli* during the spring but less so later in the year. Total phosphorus concentrations were strongly correlated with SS and also, by extension, somewhat correlated with *E. coli*. Nitrate concentrations were elevated in the river water, however, fluctuations in concentrations did not appear to correspond with fluctuations in levels of *E. coli*.

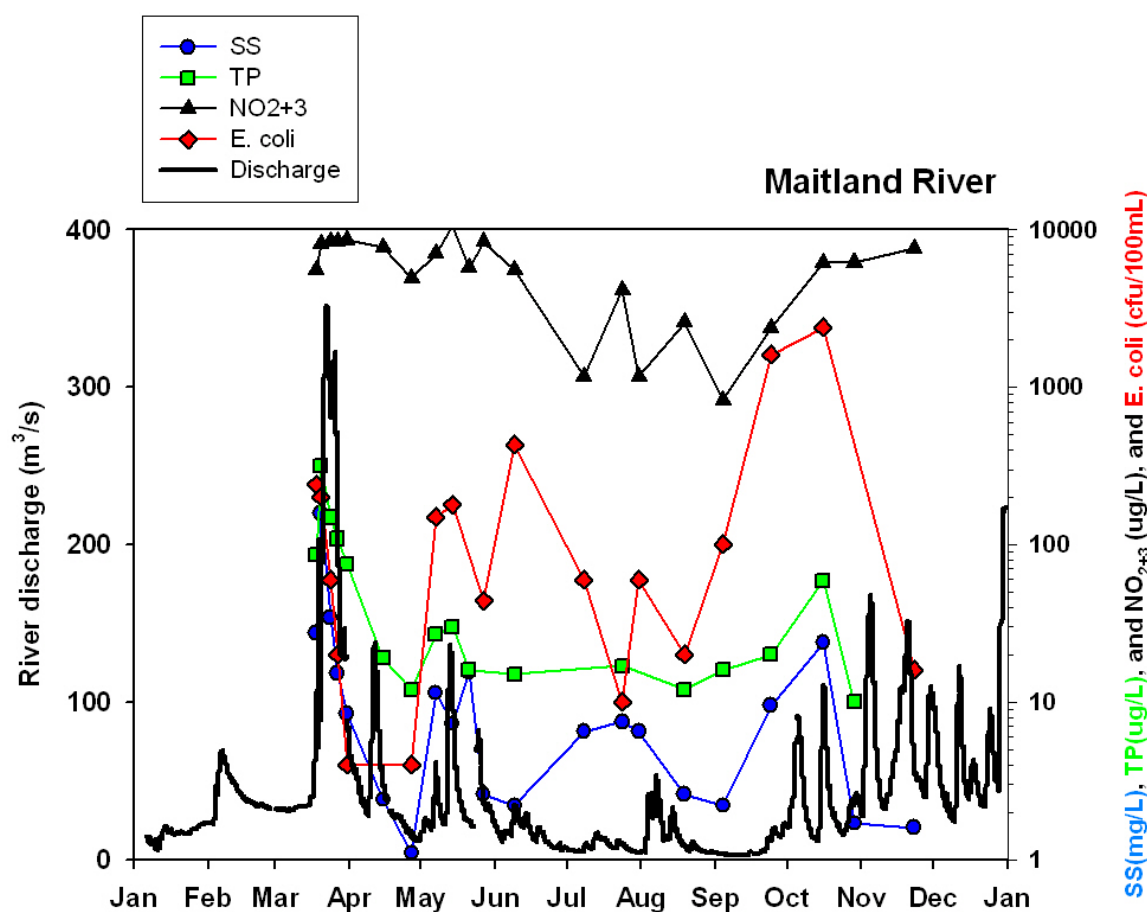


Figure 53: Discharge, suspended solids, total phosphorus, nitrates, and *E. coli* in the Maitland River near Benmiller, upstream from Lake Huron, 2003.

The fluctuation in levels of *E. coli* was more extensive in the lower Bayfield River than the lower Maitland River. The levels of *E. coli* were under 200 cfu/100mL in most samples, however, there were six occasions when levels were appreciably elevated (Figure 54), >1000 cfu/100mL, and

these were collected at different times over the survey season (March, June, August, September and October). The highest level of *E. coli* of ca. 10,000 cfu/100 mL was observed in a sample from late September. Levels exceeded 100cfu/100mL in nine of 20 samples. There appeared to be a stronger correspondence between elevated *E. coli* levels and rising river discharge than observed at the Maitland River site. As noted in the results for the Maitland River site, concentrations in SS and TP appear to be positively correlated with *E. coli* while fluctuations in nitrate concentrations appeared unrelated to levels of *E. coli*.

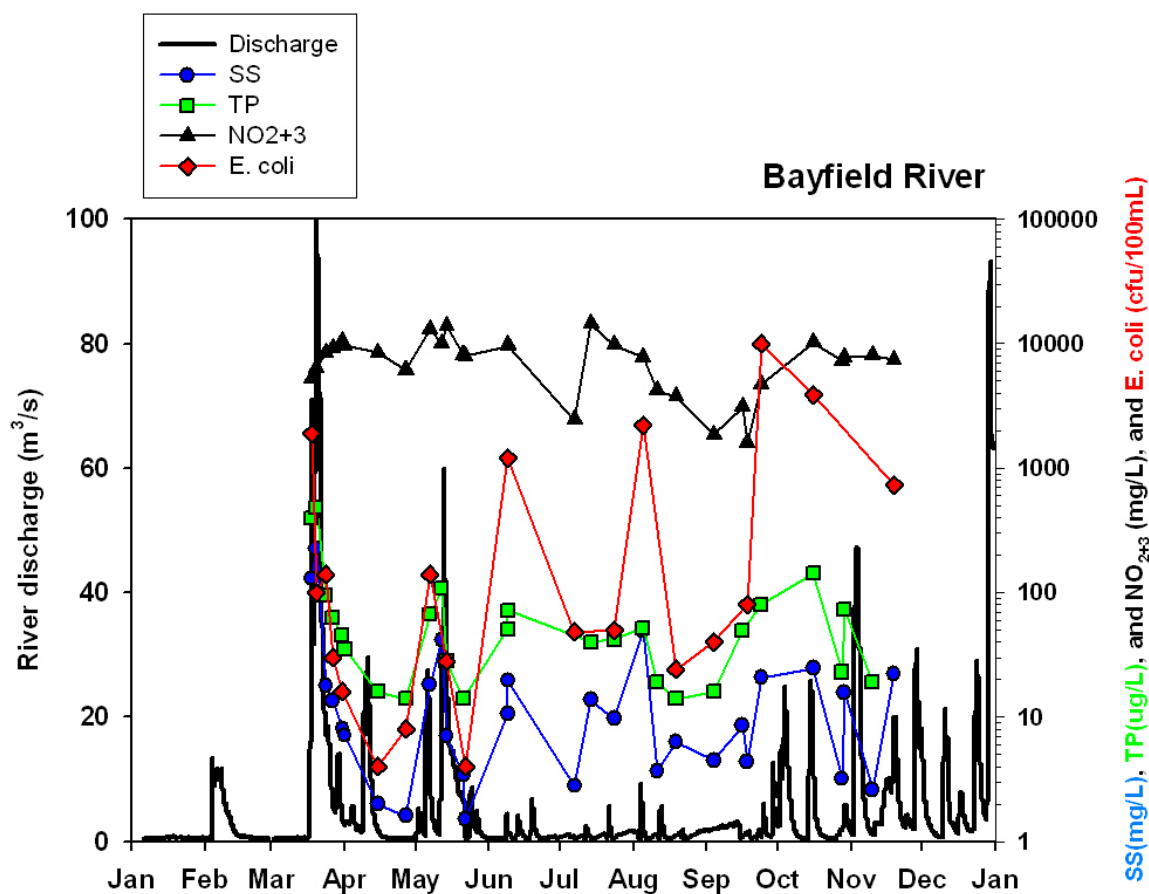


Figure 54: Discharge, suspended solids, total phosphorus, nitrates, and *E. coli* in the Bayfield River, upstream from Lake Huron, 2003.

In contrast to the Maitland and Bayfield rivers, May to November (2003) samples collected in the nearshore waters of Lake Huron adjacent to these river mouths infrequently had elevated levels of *E. coli* (Figure 55). It is important to note that the water samples were collected using a survey vessel at sites near the shoreline but not directly adjacent to the shoreline as would be the case of a beach monitoring program. The minimum water depth of a sample site was ca.

3m. On only three of nine sampling occasions did levels of *E. coli* exceed 100cfu/100 mL in lake water samples and in only three of the 98 samples collected over the nine surveys.

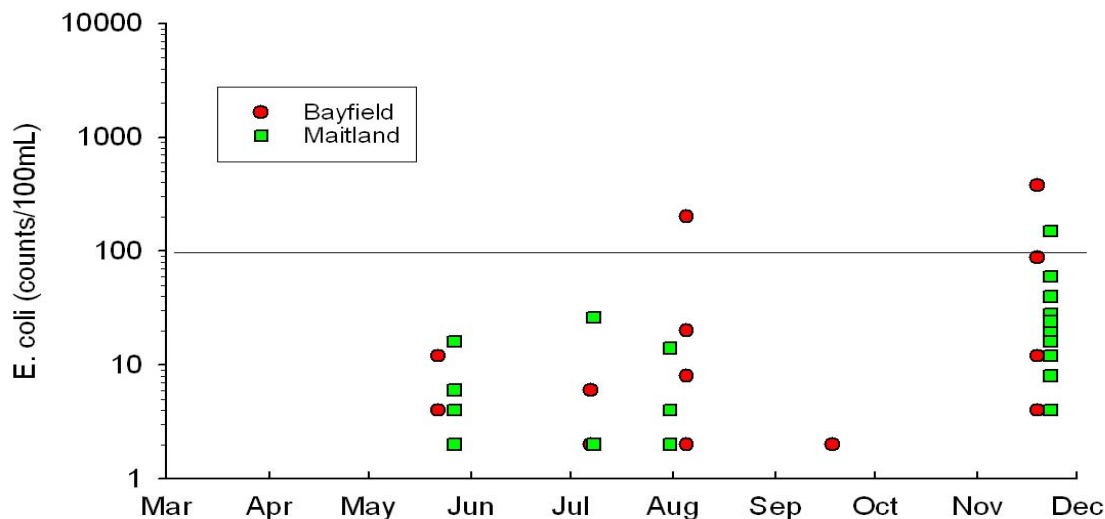


Figure 55: Surface water *E. coli* for three sites along southeastern Lake Huron in 2003, near the Maitland, and Bayfield Rivers.

An element of the 2003 study was a concurrent spatial survey of the water quality parameters in the nearshore of the lake and the downstream portions of tributaries. For the lake portions of the surveys, sensor arrays were tracked over the study areas and the resulting field data interpolated to create spatial maps of the measured water quality features. Discrete water samples for lab-based chemical analysis were also collected at points over the study area. In the tributaries, discrete water samples were collected at points along the river from the PWQMN sites used for long-term monitoring. The lake and river surveys were conducted concurrently to create datasets which could be merged to provide insight on the effect of tributary discharge on water quality features of the nearshore. Water samples for *E. coli* were collected along the shoreward edge of the nearshore and over the tributary portions of the study areas. Four and five surveys were conducted from May to November at the Maitland and Bayfield study areas, respectively. One of the planned Maitland surveys was cancelled because of poor weather.

The results for the May 27th survey over the Maitland study area illustrate the occurrence of *E. coli* in the lake in relation to the mixing areas of the river discharge (Figure 56). Levels of *E. coli* were low in the lower reaches of the river at this time, ranging from 20 to 48 cfu/100 mL. While *E. coli* was present at the mouth of the river (16 cfu/100 mL), this signal was quickly

diluted in the lake to below detection limits (2 cfu/100 mL). Nitrate concentrations (lab data) and conductivity (field data) were elevated in the tributary compared to the lake, consequently, both variables can be used to define mixing areas and plume direction in the lake. On May 27th the river plume was moving primarily to the south (Figure 56). Also of note is that SS concentrations exhibited an onshore-offshore gradient with moderate levels of suspended solids in the waters adjacent to shore.

Three additional surveys were conducted over the Maitland study area with similar results as those observed on May 27th, with respect to the occurrence of *E.coli*. At the time of the surveys, levels of *E.coli* in the lower river were not appreciably elevated and did not exceed 110 cfu/100 mL. Consequently, the effect of river discharge on the levels of *E.coli* in the nearshore waters of Lake Huron was minimal during these surveys. At no time did the levels of *E.coli* exceed 60 cfu/100mL in any of the nearshore water samples. Appendix A provides the results for the July 8, July 31 and November 23 surveys in a form comparable to Figure 56 and the May 27 survey results.

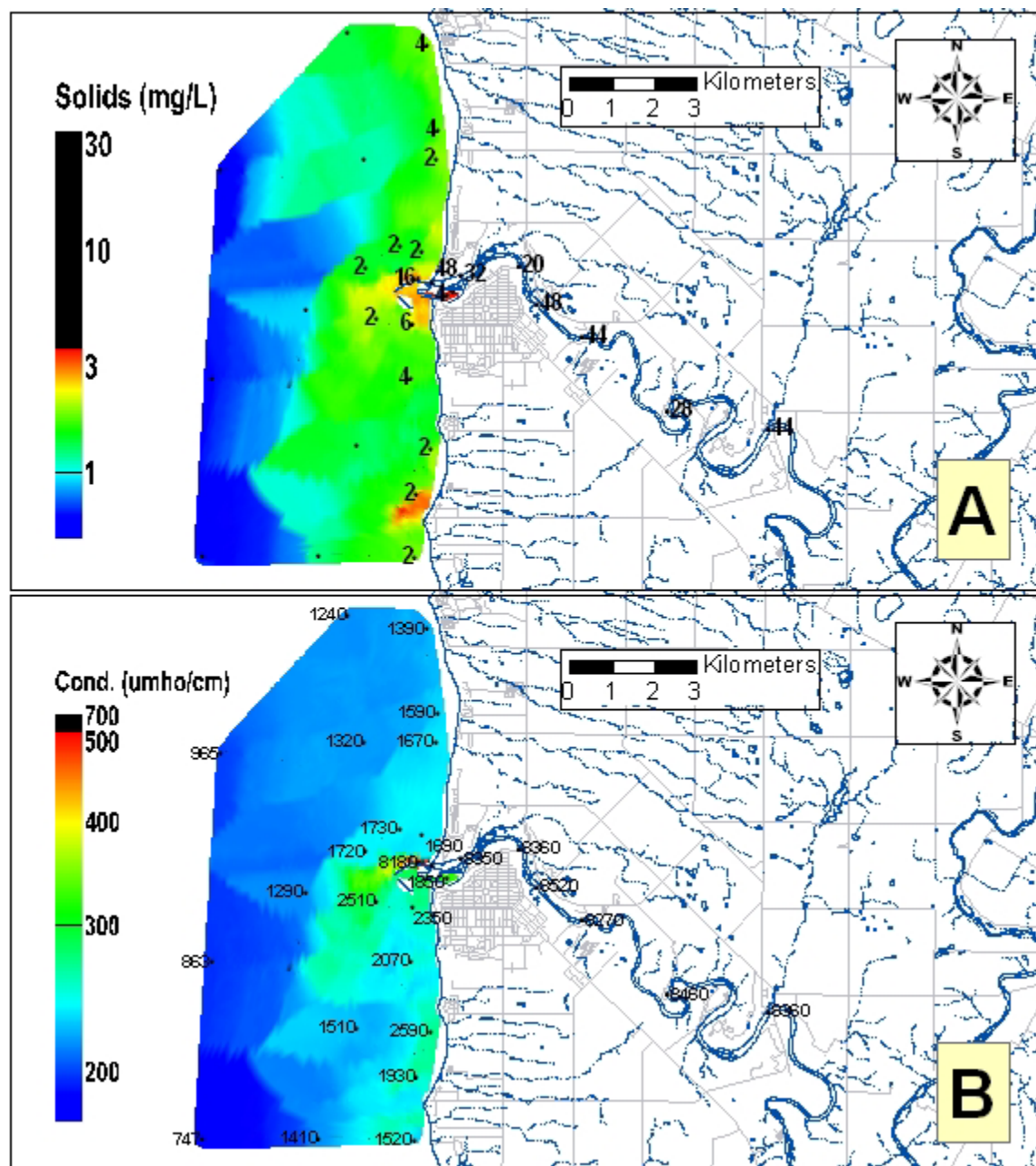


Figure 56: Surface water quality near the Maitland River, May 27, 2003; (A) Suspended solids (color image and scale bar) and *E. coli* (dots are cfu/100 mL), and (B) Conductivity (color image and scale bar) and nitrates (dots are µg/L-N).

The discharge from the Bayfield River is appreciably smaller than that of the Maitland River and it is not surprising that the scale of effect that the Bayfield River had on the nearshore in 2003 was, on average, more limited than that observed in the Maitland study area. Yet, on the first

survey of the season (May 22nd) an appreciable area of the nearshore waters was impacted by land-based inputs to the lake although not necessarily inputs originating from the river itself. Declining gradients in conductivity and nitrates were observed over the first 1-2 km from the shoreline (Figure 57). In contrast levels of *E.coli* along the shoreline were low and did not exceed 12 cfu/100 mL (Figure 57). Similarly, levels of *E. coli* in the lower Bayfield River were also low and did not exceed 20 cfu/100 mL at the time of survey (Figure 57).

A wider range of *E.coli* levels were encountered in the Bayfield River during the spatial surveys then at the Maitland Study area. For examples, in July (7th) and September (18th), low levels of *E.coli* (<100 cfu/100mL) were observed in the Bayfield River and levels in the nearshore did not exceed 6 cfu/100mL (see Appendix A), whereas elevated levels were detected in the river in August (5th) with results ranging from 2100 to 3600 cfu/100mL (Figure 58). The mixing gradient of the river discharge plume with the lake was oriented to the NW of the river mouth, however, a modest-sized area directly south of the river mouth, toward the Bayfield Main Beach, was impacted by the river as inferred from spatial patterns in conductivity and SS over the shoreline (Figure 58). Yet, only a single site in direct proximity to the river mouth had elevated *E.coli* (200 cfu/100mL) concentrations. Otherwise levels in the lake never exceeded 20 cfu/100mL. The limited effect of the river on the nearshore during the surveys is likely due to the relatively low river discharge at the time, contributing to a rapid dilution of the river water upon entering the lake. Moderate elevation of *E.coli* was observed in the Bayfield River during the final survey conducted on November 19, when levels ranged from 200 to 730 cfu/100mL among sites. With the exception of a sample collected at the mouth of the Bayfield River and in the mixing plume directly south of the river mouth (380 and 88 cfu/100mL respectively) *E.coli* levels in the nearshore were not elevated (see Appendix A).

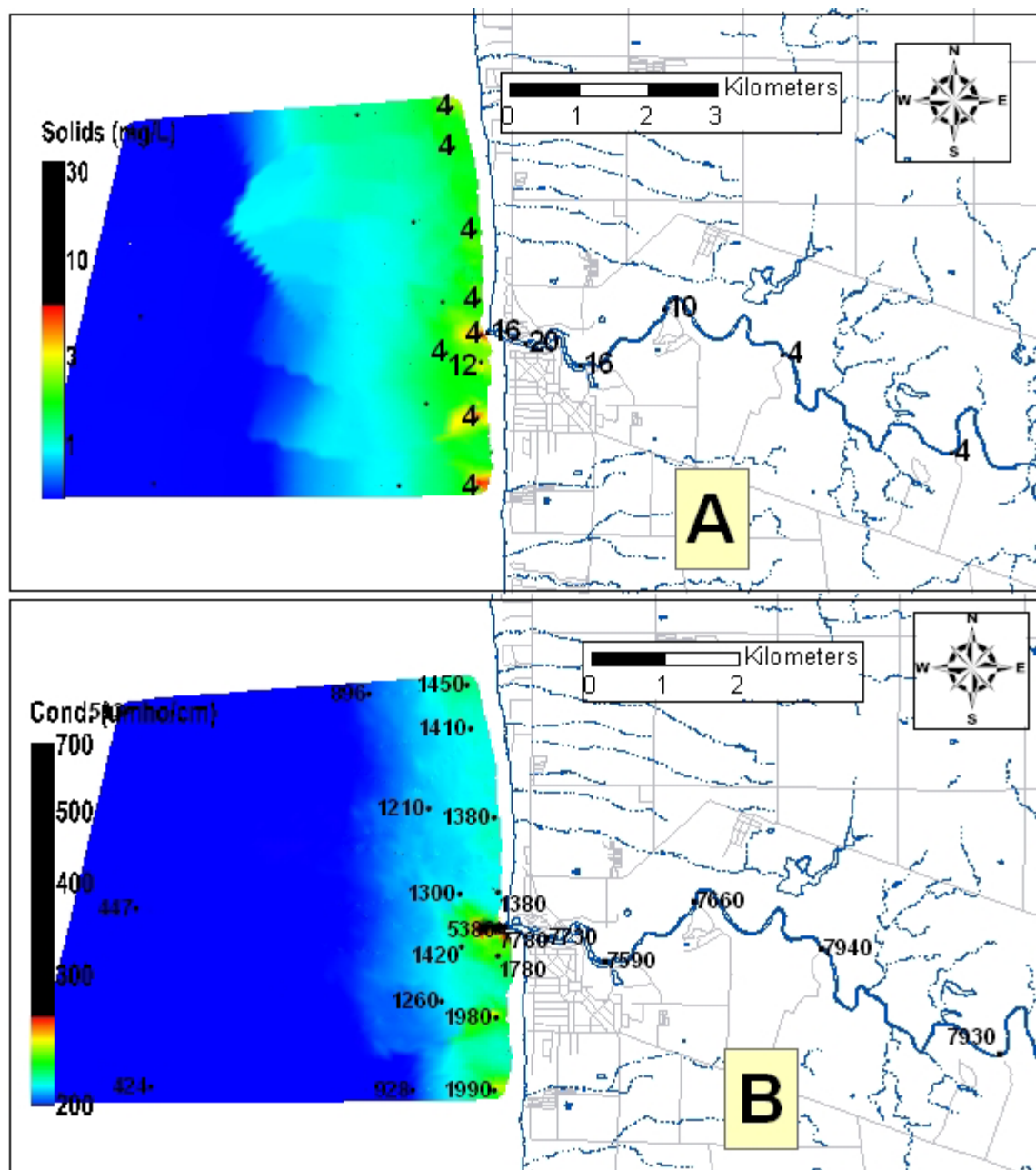


Figure 57: Surface water quality near the Bayfield River, May 22, 2003; (A) Suspended solids (color image and scale bar) and *E. coli* (dots are cfu/100 mL), and (B) Conductivity (color image and scale bar) and nitrates (dots are µg/L-N).

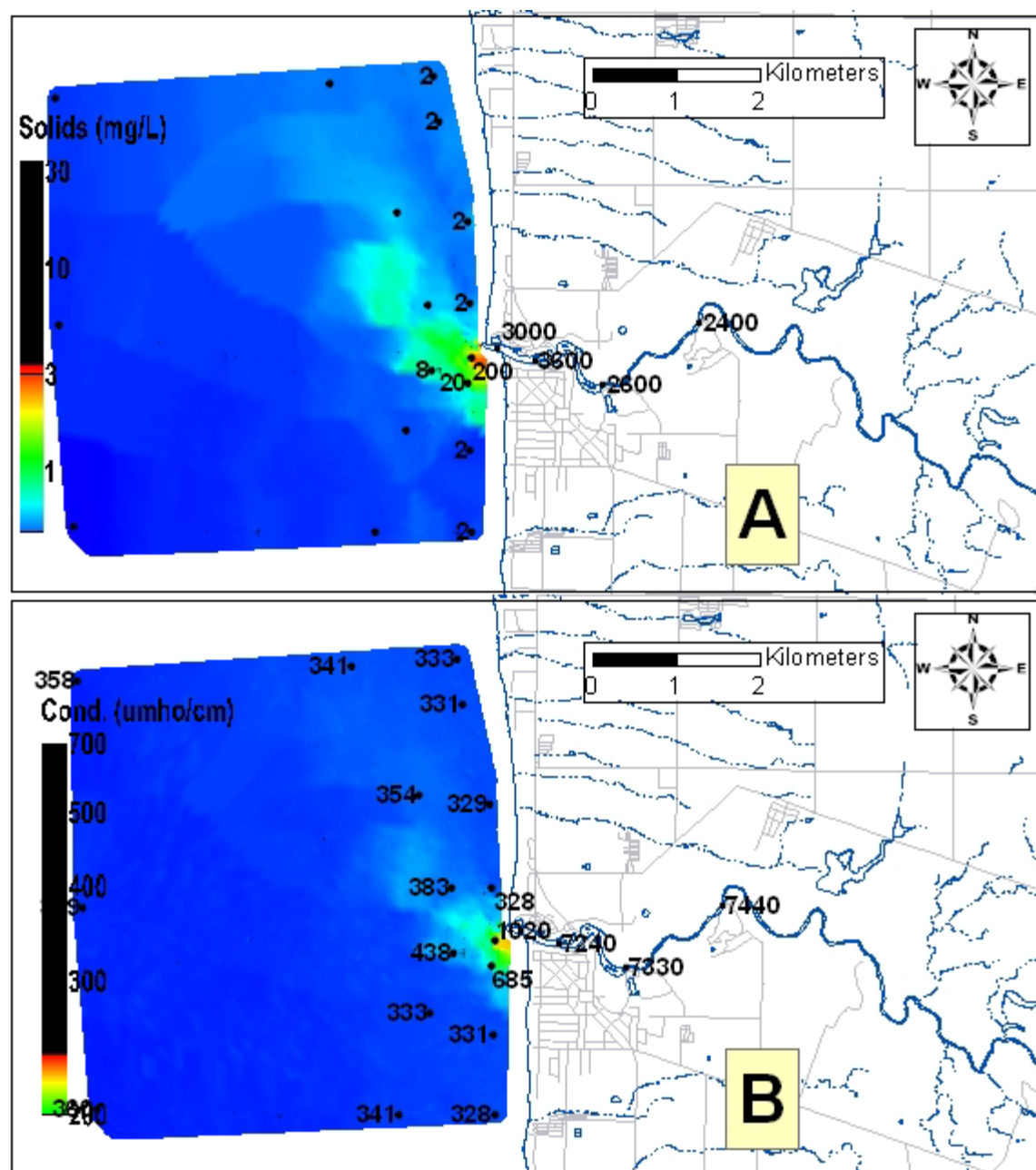


Figure 58: Surface water quality near the Bayfield River, August 5, 2003; (A) Suspended solids (color image and scale bar) and *E. coli* (dots are cfu/100 mL), and (B) Conductivity (color image and scale bar) and nitrates (dots are µg/L-N). Results for two additional river sites beyond the east edge of the map are not shown

3.0 Physical Character of the Shoreline of Huron County

This next section provides information on selected physical features of the shoreline of Huron County to assist in the interpretation of how fecal pollution generated in the watersheds is potentially delivered to the shores of the lake.

The Lake Huron shores of Huron County are dominated by two parallel landforms that give the coastline its character of sandy beaches backed by clay cliffs and ravines. The immediate shoreline, known as the Huron Fringe, is a narrow band of sand and gravel originating from glaciolacustrine deposits left behind by the retreat of the post glacial ancestral lakes of the present-day Lake Huron (Lake Huron Centre for Coastal Conservation 2004). Behind the Huron Fringe, which effectively comprises the beach areas, is an expansive clay-dominated plain perched above the shores of the lake, known as the Huron Slopes physiographic unit. The slope of the plain forming the Huron Slopes is generally gentle and in the direction of the lake. Where the plain meets the sandy shore of the lake, which is typically at a lower elevation, the clay-dominated soils of the St. Joseph Till, the primary geologic unit behind the beaches (Figure 59), have eroded and formed ravines in the cliffs through which many small tributaries now flow. The landforms behind the Huron Slopes are more varied consisting of moraines, drumlins, till plains and swamps (Singer et al. 2003).

Permeability of the shoreline sands and gravel deposits is high in comparison with the adjacent silt-clay plain (St. Joseph Till) where surface moisture movement through the soil is impeded and consequently, more prone to surface transport. The low permeability (Figure 60) of the soils and gentle slopes over much of the land draining to the lake accounts for the extensive use of tile drainage in cultivated lands. The tile drains serve to channel surface water away from agricultural fields and into drains and creeks.

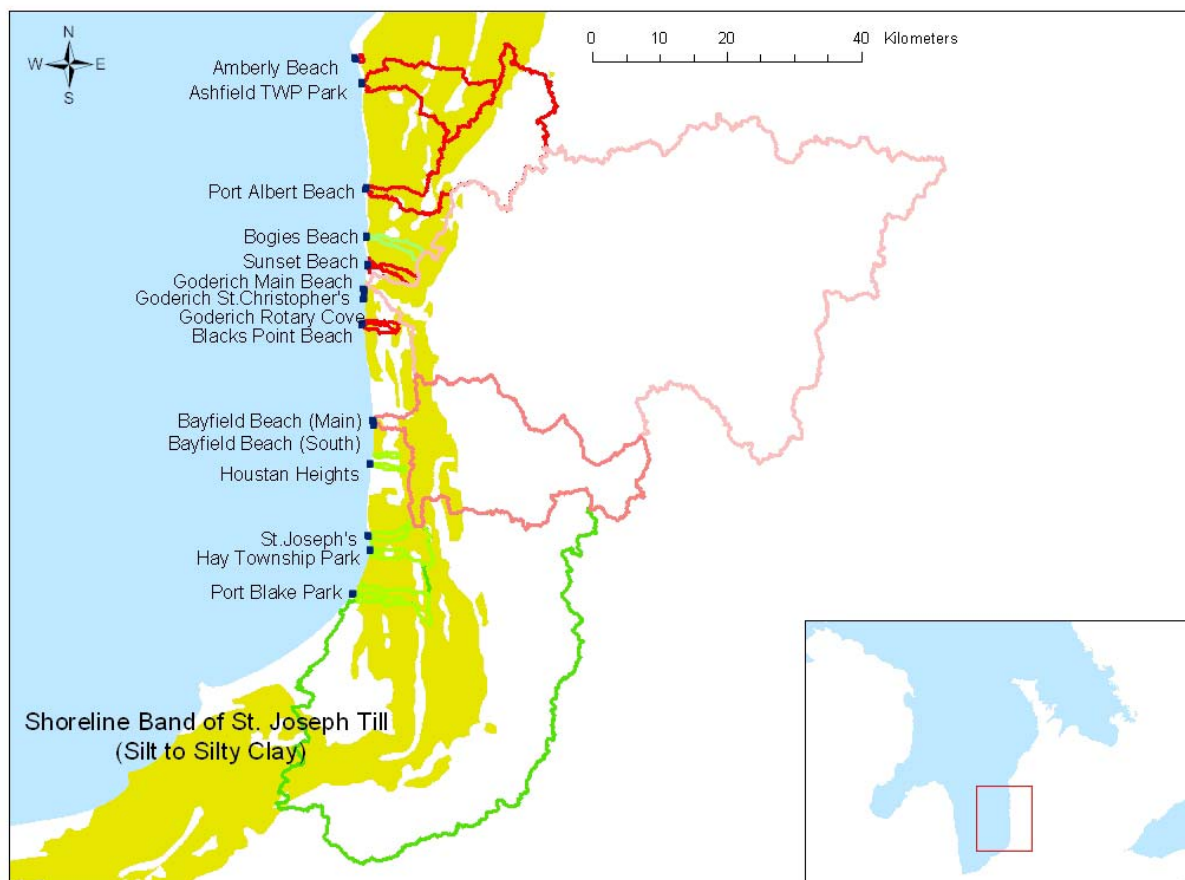


Figure 59: Map of the shoreline of Huron County showing the position of the St. Joseph Till, a clay-dominated band of soil running parallel with the Lake Huron shoreline. The information was taken from the Quaternary Geology of Ontario Dataset (1:1,000,000) of the Ministry of Northern Development & Mines.

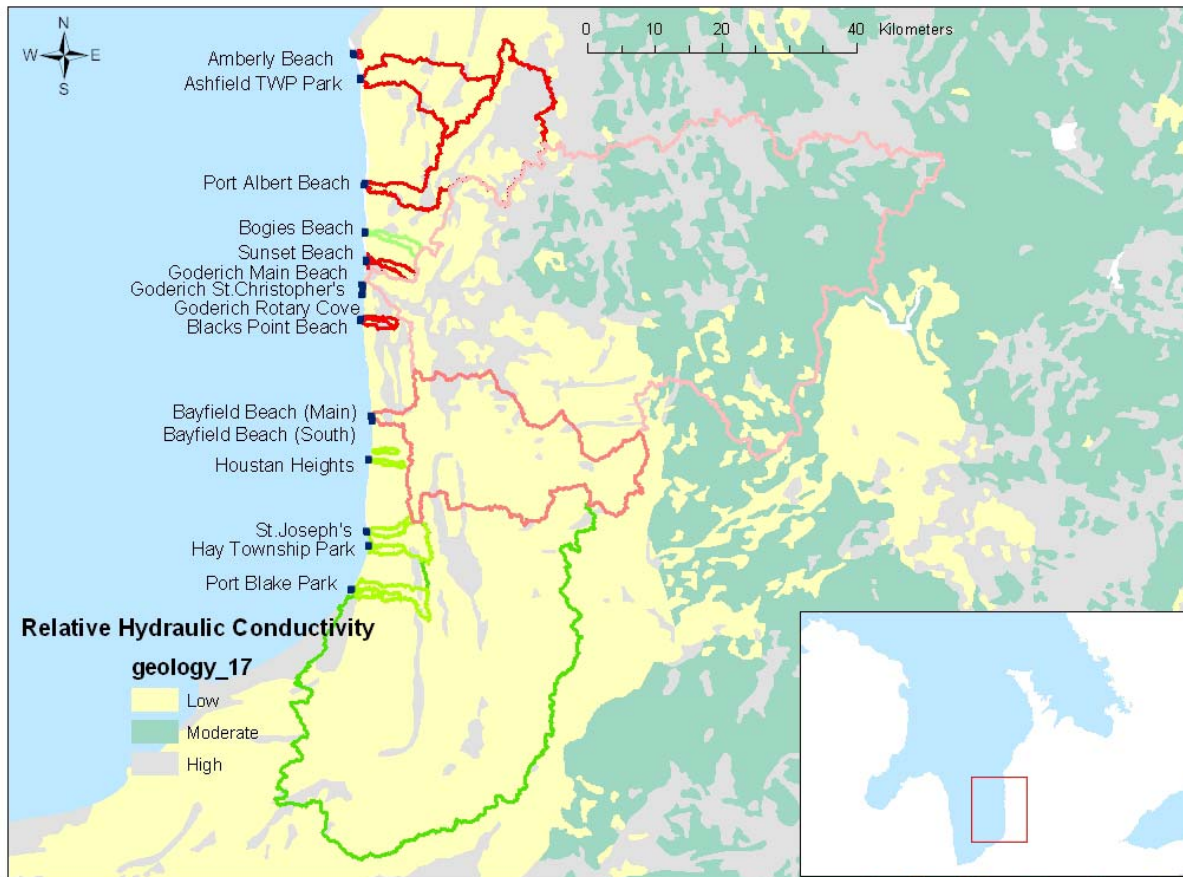


Figure 60: Map of the shoreline of Huron County showing the areas of high to low porosity over the land adjacent to the Lake Huron shoreline of Huron County. The information was taken from the Quaternary Geology of Ontario Dataset (1:1,000,000) of the Ministry of Northern Development & Mines.

A prominent feature of the shoreline of Huron County is the many tributaries running perpendicular to the shoreline which drain the lands adjacent to the lake. The size of the tributaries is highly variable ranging from small creeks to large river systems. Approximately 124 water courses, directly discharging to the lake, are depicted on the 1:10,000 Ministry of Natural Resources Ontario Base Maps (OBM) along the shores of Huron County. Many of these creeks and rivers extend less than 10 km from the shores of the lake, however, a limited number of others, such as the Maitland and Bayfield Rivers, extend many kilometers back from the lakeshore.

The size and proximity to the lakeshore, of the land areas drained by the various creeks and rivers is highly variable along the shores of the lake. In the majority of cases the drainage area is limited in size and the land is in close proximity to the lake. In a limited number of cases but accounting for the majority of the land area draining to the lake, the drainage area is extensive, parts of which are far removed from the shores of the lake. The frequency distribution of

drainage areas among the 124 watercourses to the shores of Huron County are given in Figure 62. There are 78 (63%) drainage areas that are $<5 \text{ km}^2$ in area. Of the land area (ca. $5,200 \text{ km}^2$) draining into Lake Huron on the shores of Huron County, about 80% is drained by the Maitland, Ausable and Bayfield Rivers.

The characteristics of the effects of a creek or river discharge on the shoreline will be determined by the size of the watercourse. The volume of water transported by a small creek will be limited and the mixing zone in the lake at the river mouth will be localized near the mouth when the creek is flowing. Alternatively, large rivers will transport large volumes of water and likely have large mixing zones within the lake. Likewise the impacts that a tributary will have on water quality along the shoreline will likely vary appreciably between tributaries. Furthermore, the timing of the effects will also likely vary as a function of tributary size. For example, some small tributaries experience periods of low or no flow over the summer, these summer periods of low flow may be punctuated by pulse flows in response to wet weather conditions. In the larger rivers, positive flows are maintained over the summer, albeit with base flow well below the peak flows of the spring.

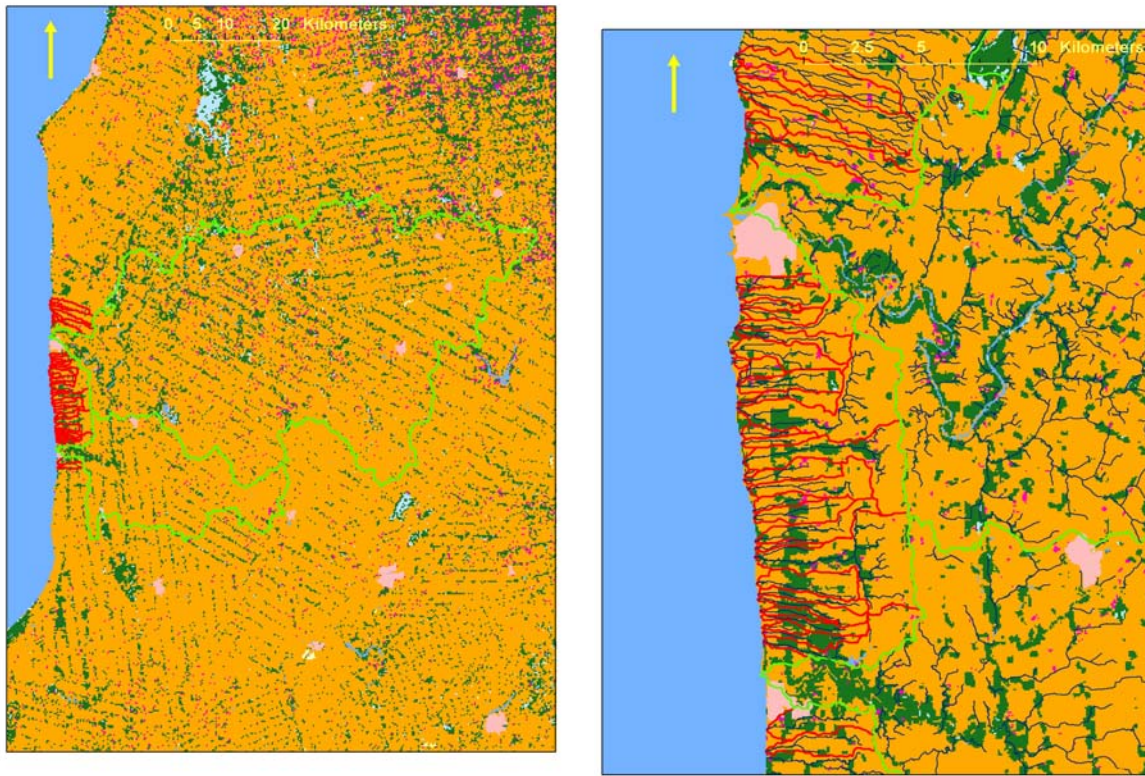


Figure 61. Drainage areas of watercourses to Lake Huron between Bayfield and Point Farms are outlined in green (Bayfield and Maitland Rivers) and red (all others) to illustrate the contrast in size of drainage areas along the shoreline. The left panel show the full extent of the drainage areas and the right panel is an enlargement of the shoreline between Bayfield and Point Farms. The drainage areas are plotted on land-use maps (see 2.4.10). Agricultural lands are shown in orange, forested areas in green, built-up areas in pink and surface water in blue.

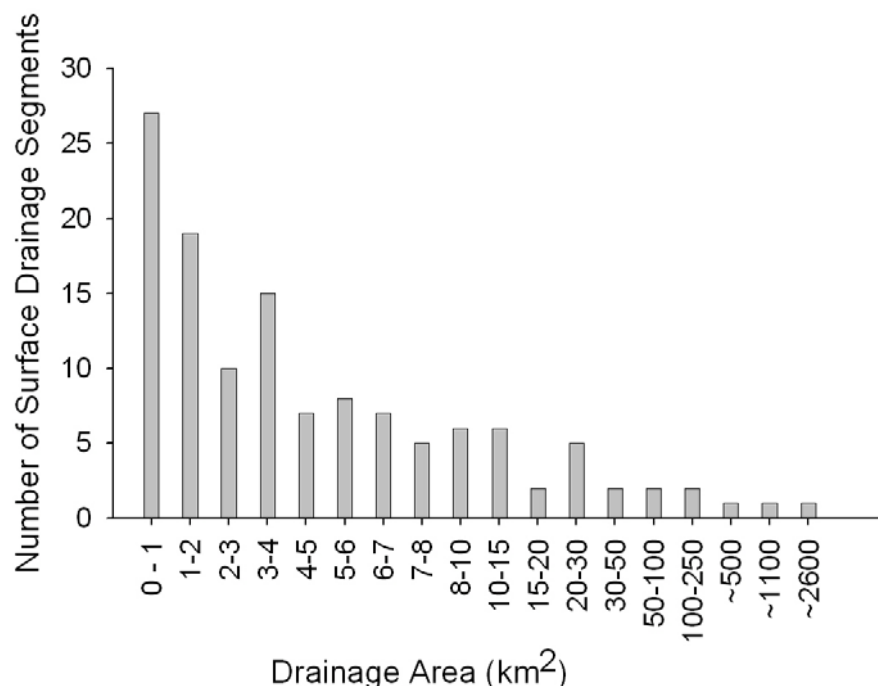


Figure 62. Frequency distribution of drainage area of the 124 watercourses depicted on 1:10,000 OBMs draining to Lake Huron along the shores of Huron County.

4.0 Limnology of the Shoreline of Huron County

Knowledge of the physical and chemical features of the nearshore lake environment and of the physical process which drive the flux of materials at the shores of the lake is essential to understanding microbial pollution of beaches on the shores of the Great Lakes. Information on the physical and chemical characteristics of the lake waters, lake bed and shoreline is needed to interpret how microbial pollutants will persist in the environment. The evaluation of the transport of pollutants from sources to sinks, and from sinks to areas of impact is essential if the sources driving adverse water quality conditions are to be determined. This next section provides brief background on selected features of the nearshore lake environment which are potentially related to how microbial pollution is manifested at shoreline beaches.

4.1 Trophic status and water quality

The trophic status of lakes is generally determined using measures of phytoplankton productivity, nutrient concentration, and light transparency in the epilimnion, and/or oxygen depletion in the hypolimnion. Lake Superior, Lake Huron, and Lake Michigan have been ranked, in order, as the most oligotrophic of the Great Lakes, based on offshore data collected from 1967 to 1975 (Chapra and Dobson 1981). Of all the Great Lakes, including Lake Michigan, Lake Erie, and Lake Ontario, Lake Huron had among the highest secchi disk depths (8 – 10 m), and the lowest concentrations of chlorophyll-a (1 – 2 ug/L) and total phosphorus (TP) (around 5 ug/L), during that time interval. Compared to the smaller Great Lakes, Lake Huron has not been as sensitive to anthropogenic eutrophication due to its larger volume and the relatively reduced urbanization of its watershed. The offshore waters of Lake Huron remained oligotrophic during the eutrophication of Lake Erie, which accelerated from the 1930s through the 1960s. During this time, in Lake Erie, suspended solids increased 36% (140 – 190 ug/L) and chloride increased 243% (7 – 24 ug/L) (Beeton 1965), whereas, in Lake Huron, suspended solids increased 9% (110 – 120 ug/L) and chloride increased 75% (4 – 7 ug/L).

Water quality is generally more variable in nearshore areas along southeastern Lake Huron compared to offshore waters. Nicholls et al. (2001) summarized nearshore municipal water intake data for the Great Lakes from 1976 to 1999. At Goderich, median monthly TP concentrations ranged from 11 – 40 ug/L. Variability in water quality decreased further along shore to the south. Median monthly TP was 9 – 20 ug/L at Grand Bend, and only 2 – 7 ug/L at Lambton. This spatial variability in water quality may have reflected the water intake depths. Resuspension of bottom sediments is expected to be greater at shallower depths due to increased wave energy. Distance from shore can also determine the proximity to tributary plumes, which can strongly influence nearshore water quality in the Great Lakes (Howell and Hobson 2003). The water intakes at Goderich, Grand Bend, and Lambton were at progressively greater depths (4.9, 7.9, and 11.9 m, respectively), although the distances from shore were variable (488, 2530, 101 m, respectively). Regardless, it was clear that nearshore TP was much higher and more variable compared to offshore data from other studies (Dobson 1971; Moll et al. 1980; Beeton and Saylor 1995).

4.2 Thermal regime

Lake Huron is a dimictic lake, that is, the lake mixes completely twice annually (spring and fall) and the entire water column is thermally stratified in the summer. Mean water temperature remains below 4 °C in April and May, increasing to a maximum of around 20 °C in July – September, subsequently decreasing back to around 4 °C through December (Dobson 1971). Miller and Saylor (1981) determined the winter thermal structure of Lake Huron in 1974/5. The decomposition of stratification and eventual mixing of surface and lower waters was observed in late fall and early winter. In early December, the end of the stratified period was initiated by several pulses of relatively warm surface water (> 5 °C) to the hypolimnion on a daily time scale. Following these events, temperatures at the 50 m and 158 m depth contours converged around 5 °C, allowing for water column mixing. The shallow south-central basin and nearshore areas (i.e. < 50 m depth) reached the winter minimum temperature near 0.2 °C by late February and remained isothermal. The deeper north central basin reached the winter minimum temperature of 1.5 °C by early April and was weakly and inversely stratified during the winter. The mean monthly isotherms generally coincided with the lake bathymetry. The deeper central regions were warmer and surrounded by relatively cooler, less dense water. For example, in February, the central portion of the lake was 3 °C and the nearshore was 0.5 to 1.0 °C.

Seasonal thermal stratification in the Great Lakes is characterized by the formation of thermal fronts. Warm fronts, referred to as thermal bars, occur in spring when nearshore waters warm faster and are separated from cooler offshore waters by a 4 °C front. Thermal bars are most common in May, in Lake Huron. Cold fronts occur in the fall, when nearshore waters cool faster than warmer offshore waters. Cold fronts are less frequent because of increased wind-driven mixing. Ullman et al. (1998) examined thermal fronts in the Great Lakes using National Oceanic and Atmospheric Administration satellite imagery collected from 1985 to 1995. In Lake Huron, thermal bars occurred most frequently around the southwest and southeast nearshore waters at depths < 50 m. Thermal fronts also occurred, to a lesser extent, along the bathymetric ridge separating the deeper north basin from the southern basin. Thermal fronts along nearshore areas can reduce the mixing with offshore waters, trapping nutrients and suspended solids from tributary plumes nearshore. Therefore, the timing and duration of these fronts may significantly influence nearshore water quality.

4.3 Offshore and nearshore physical circulation

The main forces causing circulation in the Great Lakes are hydraulic flows, and currents driven by wind stress and spatial gradients in density and temperature (Beeton and Saylor, 1995). In Lake Huron, hydraulic currents are created by inflows from the St. Mary's River from Lake Superior and the Straits of Mackinac from Lake Michigan. The only outflow is through the St. Clair River. Surface currents (up to several tens of cm/s) are generated by wind stress on the lake surface, and average seasonal currents are in the order of a few cm/s (Beletsky et al. 1999). Horizontal convection or density-driven currents occur in Lake Huron as a result of spatial temperature gradients. Internal circulation caused by seiches can result from steady winds, although, in Lake Huron, seiches are generally small in magnitude because of the lake's great depth, this is in contrast to large seiches commonly observed in Lake Erie (Beeton and Saylor, 1995).

Early studies on the circulation patterns of the Great Lakes were based on surface trajectories of floating bottles (Harrington, 1894) or current measurements taken from synoptic daily cruises (Ayers et al., 1965). More recently, Beletsky et al. (1999) used data from moored current meters (Sloss and Saylor 1975) for a more accurate description of mean Lake Huron circulation patterns. Offshore mean circulation patterns of the central basin, west of the Bruce Peninsula, were characterized by a cyclonic gyre (counter-clockwise). On average, summer circulation patterns were similar to winter patterns, with currents moving southward along the west coast, turning northward up the east coast. The minimum, maximum, and mean current speeds in summer (0.4, 4.6, and 1.3 cm/s respectively) were generally less than in winter (0.2, 7.9, and 2.6 cm/s, respectively).

Closer to shore, nearshore currents are complicated by local winds, bathymetry, and geographic features of the shoreline. Wind-induced nearshore currents are created by: 1) oblique waves breaking along shore creating alongshore currents, and 2) nearshore cell circulation (i.e. rip currents) created by alongshore currents in opposition or running against shoreline features (Komar 1976).

Information on water current directions and speeds can inform on the potential of material (e.g. sediment, nutrient etc.) transport. In southeastern Lake Huron, Lawrence and Davidson-Arnott (1997) calculated sediment transport along the coast between Point Clark and Goderich based on back-calculated wave patterns using wind speeds collected during the openwater seasons from 1952 – 1987. Average surface currents along southeastern Lake Huron flow northward

and nearshore transport is expected to follow the current direction (Beletsky et al. 1999). However, Lawrence et al. suggested that net sediment transport was typically southward, with northward reversals in lee areas of the shoreline. In fact, the calculated areas of net deposition vs. areas of net erosion appeared interspersed along the coastline. Annual wave energy flux at the shoreline increased from Point Clark south to Goderich, but beach width and annual recession rates did not correspond with wave energy. Nearshore sediment transport did not directly follow patterns in wave energy, which can be high enough to create erosion in certain areas. Sediment transport was dependent on the supply of erodable sediments from shorelines or tributaries, along with wave energy.

Patterns in sediment transport affect the variability in nearshore water quality, and it is important to understand processes governing this phenomenon. Resuspension is one part of sediment transport, which includes erosion, transport, and deposition. Nearshore wave energy, required for sediment transport, is mediated by bathymetry and shoreline geographic features. The supply of erodable materials may also limit resuspension. The grain size of sediment particles will determine how long those materials remain in the water column (i.e. deposition), and by extension the distance they can travel with a given wave energy. Ultimately, sediment quality will determine how water quality is affected by sediment particles and associated nutrients.

4.4 Beach processes

The ecology of beach areas relies heavily on the interactions between the land and the water at the shoreline. Nutrient delivery, nearshore currents, sediment deposition and erosion are some of the factors that can influence the microfauna of beaches. Bacteria may survive in the interstitial waters of sandy beaches based on a supply of nutrients delivered by waves onshore. Their occurrence in bathing waters of particular beaches may be due to their longshore transport by nearshore currents. Bacterial survival and transport are both linked to the adhesion of bacteria to sediment particles; consequently, deposition and erosion rates of beaches may directly influence the occurrence of bacteria in beach areas. A general understanding of the physical processes occurring in beach areas is essential to achieve a better understanding of the processes governing bacterial behavior in beach habitats.

Areas within the beach profile have been defined and categorized by geomorphologists and oceanographers. Figure 63 below summarizes the classical terminology used to identify the different parts of a beach cross-shore section, as compiled by Komar (1976).

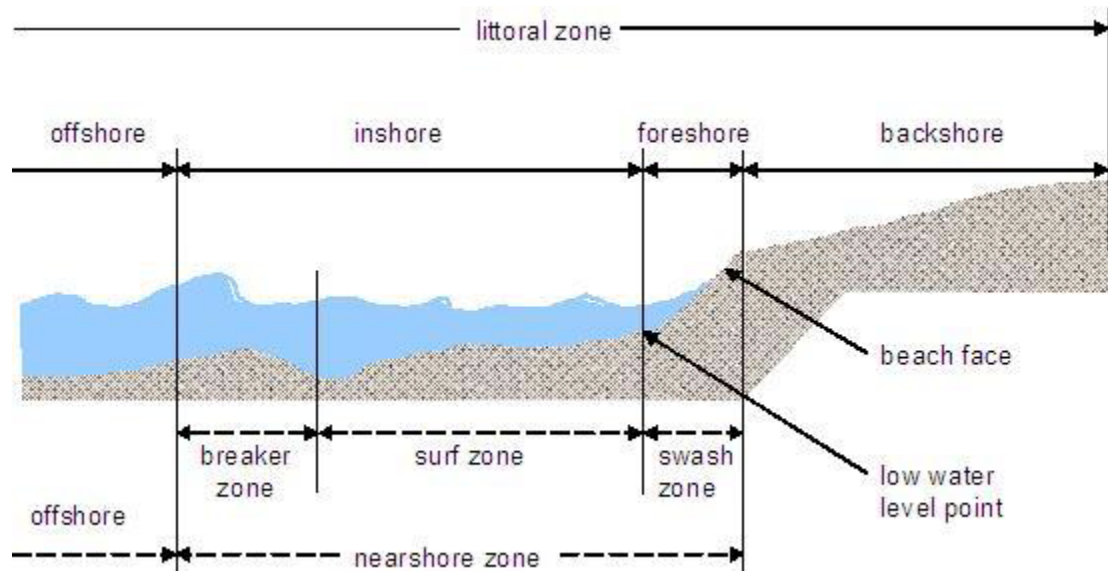


Figure 63: Classical terminology of a typical beach profile. The dashed arrow lines refer to terminology of wave actions and currents in the nearshore region, while the solid arrow lines refer to the physical areas of the beach (based on Komar 1976).

Definitions of the terms in Figure 63 are as follows. The backshore refers to the area of the beach that extends landward from the sloping foreshore to the point of development of vegetation or change in physiography (i.e. cliff, dune field, etc). The foreshore is the sloping portion of the beach profile between either a distinct crest or upper limit of wave swash at high tide, and the low-water mark of the backrush of the wave swash at low tide. The inshore is the area from the foreshore to just beyond the breaker zone. The breaker zone is the portion of the nearshore region where arriving offshore waves lose stability and break. The surf zone is the portion of the nearshore region where translation waves occur following breaking waves, and extends to the swash zone. The swash zone is the portion of the beach face that is alternately being covered by the uprush of the wave swash and exposed by the backrush. The beach face refers to the sloping section of the beach profile which is normally exposed to the action of the wave swash. The littoral zone in geomorphology refers to the zone that extends from the landward end of the backshore to a point in the water where sediment is less actively transported by surface waves. The nearshore zone extends from the shoreline to just beyond the region where the incoming waves break. Finally, the offshore zone is the portion of the profile that extends beyond the breaker zone or inshore area. Note that the definition of the

term nearshore in geomorphology is more exact than in a limnological setting (and as used elsewhere in this report) where nearshore refers to the lake waters extending from the shoreline to an undefined distance or depth offshore.

The prevailing dynamics of beach processes rely predominantly on wind conditions. Winds transfer their energy to the water, which in turn carries it to the shoreline in the form of waves. Winds also transfer their energy directly by blowing over the foreshore and backshore, moving sand and exposed sediment around. Areas beyond the backshore, sand dunes in particular, are also affected by the wind and contribute to sand and sediment movement. Wind is a major contributor to the prevalence and action of longshore currents in the Great Lakes; these in turn have a role in the movement of sediments in nearshore areas. However, most research has focused on the dynamics of wave energy and its impact on sediment transport in nearshore areas. Particularly, the swash zone has been of interest to researchers from various backgrounds due to its influence on beach morphology, coastal stability and microbial ecology. Wave dynamics in the surf zone and its effects on sediment transport in the nearshore has also been extensively studied. The following are discussions on the dynamics of nearshore wave and current action as well as swash zone and backshore dune areas with respect to sediment behavior and transport.

Nearshore wave and current action:

Wave-induced longshore currents are the main cause of sediment movement in the nearshore (Komar 1976). Energy from waves approaching the shoreline at a perpendicular angle and the resulting backwash from the shoreline will support and suspend sand and sediment from the bottom. Typically, the sand and sediment will move back and forth with no net movement. However, when a longshore current is transposed onto this scenario, the sand and sediment will experience drift in the direction of the longshore current. The longshore current does not need to expend any energy in causing the sand or sediment to suspend itself in the water column, as the shoreline-approaching waves and backwash have already done so, thus even relatively small longshore currents may cause appreciable sand and sediment movement.

Another aspect of sediment transport outside the transition zone (i.e. the high energy zone where the incoming waves meet the backwash causing suspension of sand and sediment) is the movement of the bed sediment. The bed sediment comprises the concentrated sediment that moves on or in close proximity to the bottom. The sediment is less suspended in the water

column and tends to move along the bottom as a sheet, as the energy of breaking waves is absorbed by the overlying water column causing minimal suspension. Since bed sediment movement occurs in the surf zone and beyond where the longshore currents are generally stronger, it is assumed that bed sediment transport dominates over suspended sediment transport.

The effect of wave action on grain size distribution of sediment in nearshore areas also needs to be considered when examining microbial delivery to beach areas (see 2.2 for discussion of grain size effects on particle-bacteria associations). The coarsest grain sizes occur at the plunge point of incoming breaking waves in the surf zone and foreshore area. Grain sizes become finer with increasing distance in either direction from the plunge point. The distribution of grain sizes in the nearshore beach sediment closely reflects the energy level of the wave processes.

Swash zone dynamics:

Although the swash zone may be a relatively small part of the overall beach profile, there are many complex forces at play in this zone that have a bearing on the net movement of sediment along and at the beach. One major consideration is the hydrodynamic interplay between the swash wave action and the beach water table. The position of the beach water table will influence the infiltration/exfiltration rates in the swash zone (Horn 2002). Low water tables will result in larger vadose zones, which will allow for the infiltration of incoming swash waves. The infiltration or percolation of water into the beach vadose zone allows for the landward net deposition of sediment. Alternatively, high water tables cause exfiltration effects in which erosion of the beach face predominates, resulting in the seaward net deposition of sediment. The detailed kinetics of these mechanisms is not well understood, but the transition from deposition to erosion also seems to be affected by beach grain size, with finer grain-sized beaches more susceptible to erosion (Butt *et al.* 2001).

Longshore sediment transport in the swash zone may account for as much as 50% of the total longshore sediment transport (Elfrink and Baldock 2002). When waves approach the shoreline at oblique angles, sand and sediment are subject to “swash transport” (Komar 1976). This type of transport follows a zigzag motion along the shoreline. The incoming wave swash drives the sand up the beach face at an oblique angle and the return gravity flow (backwash) brings it back

to its original level in water column, causing a net transport along the shoreline. Figure 64 below illustrates this dynamic transport mechanism. The factors most affecting this type of sediment transport are wave energy and beach face steepness. As both increase, the nearshore range affected by this type of transport also increases.

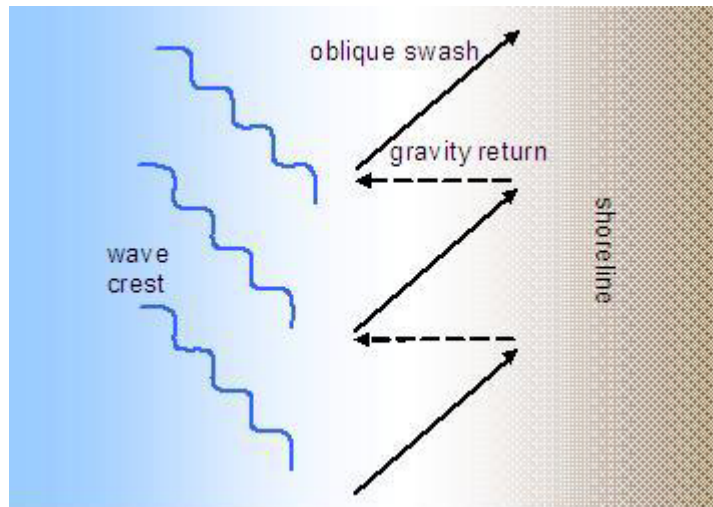


Figure 64: Zigzag movement of sand and sediment in “swash transport”.

5.0 Communal Sewage Works as Sources of *E. coli* to the shoreline of Lake Huron

There is a diversity of potential sources of fecal pollution over developed lands. The remaining sections of the report will provide overviews of the key suspected sources of fecal pollution to the lands draining to Lake Huron over the study area. Depending on availability of information for the study area and the state of understanding as a source of fecal pollution, information will be presented on the occurrence of the sources over the landscape and on the associated mechanisms of delivery of fecal pollutants to surface waters. This initial section will discuss sewage works as sources of fecal pollution.

The overall level of urbanization over the lands draining to the Lake Huron shores of Huron County is low, nevertheless, there are a number of developed and urbanized areas where municipal or communal systems are used to collect and treat sewage and waste water. The effluent discharges from these facilities represent potentially significant sources of microbial loading to the environment.

Information was collected on the effluent flow rates and bacterial (*E. coli*) concentrations for nine sewage treatment plants (STPs) in Huron County in the Maitland, Bayfield and Nine Mile River watersheds within 90 km (river flow distance) of the Lake Huron shoreline (Figure 65). Other STPs further upstream and in Perth County were not investigated in this first phase of the study. STPs typically provide primary treatment (settling of solids), secondary treatment (biological degradation of organic matter and nitrogen), and disinfection (by chlorine or ultra-violet light) of wastewater to achieve an *E. coli* compliance limit of 200 cfu/100 mL. All STPs in Huron County now disinfect their effluent year-round although seasonal disinfection may still be the legal requirement in some of the sewage works approvals issued under the *Ontario Water Resources Act* (section 53).

STPs can also deliver microbial pollutants to the environment in sewage treatment bypasses which occur during periods of heavy rain or snow melt. In these circumstances, wet weather flows enter the sanitary sewage collection system by infiltration or by storm sewers connected to sanitary sewers, known as combined sewer overflows (CSOs). When these CSOs exceed the hydraulic capacity of the STP, the plant operators discharge the excess flow directly to the lake or river. A discharge of untreated sewage is called a primary bypass, while discharge of sewage which has had primary treatment is called a secondary bypass. Bypasses may be disinfected although the efficacy (bacteria kill) of this process is not assured because the large flows mean reduced contact with the disinfectant. Sewage pumping station overflows are another source of raw sewage discharge to the environment. A CSO may also discharge untreated sewage to the environment when during an overflow event, sanitary wastewater mixes with stormwater runoff and the stormwater is discharged at the CSO outfall.

Bypass control targets are set out in MOE procedure F-5-5 (Ontario Ministry of the Environment and Energy 1994b). One target is to achieve the body contact recreational water quality objective at beaches impacted by CSOs for at least 95% of the four-month summer period (June through September) for an *average* year of rainfall or runoff. In the absence of site-specific concerns, the effluent quality criterion for disinfected combined sewage is a monthly

geometric mean not exceeding 1000 *E. coli* per 100 mL. During a seven-month period (April through October), 90% of the volume of wet weather flow shall be captured and treated as much as possible.

The information on STPs which follows was gathered from MOE files and inspection reports. STPs with design capacities greater than 10 m³/d are required to obtain sewage works approval from the MOE. The approval documents outline STP monitoring and reporting requirements. An STP with an older approval would have a minimal requirement for monitoring and reporting effluent quality. An STP with a newer approval would have more comprehensive requirements. In 1996, MOE requested all Ontario municipalities to voluntarily conduct bypass monitoring and reporting. Conformity with this request has not been complete in Huron County, as such, the available information is inconsistent.

In addition to STPs, there are other types of communal sewage treatment being used in Huron County such as subsurface (tile beds) and spray irrigation sewage disposal systems which do not have effluent flows amenable to monitoring. A partial listing of the non-STP communal treatment systems identified in the information gathering to date is also provided here for areas draining to the shoreline between Amberley Beach and Bayfield.

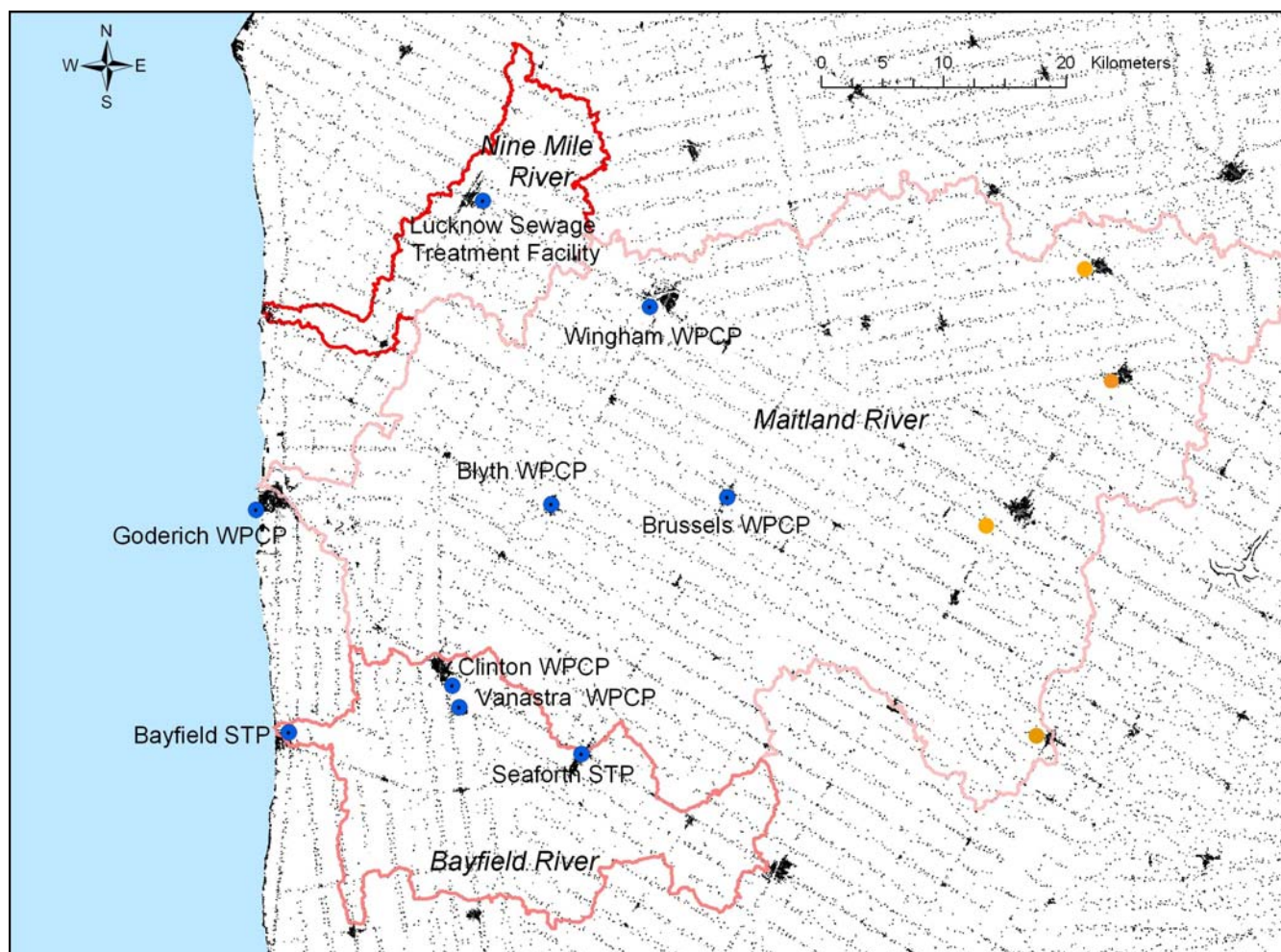


Figure 65: Map of the locations of sewage treatment facilities on lands draining to the Lake Huron shores of Huron County between Amberley and Bayfield (blue and yellow circles). The facilities shown by blue dots are discussed individually in the report. Also shown are the drainage areas for the Nine Mile, Maitland and Bayfield Rivers. The locations of buildings depicted on 1:10,000 OBM maps are indicated by the black and grey (buildings to scale) dots.

5.1 Selected Sewage Treatment Plants

Goderich STP:

The largest community in Huron County is the town of Goderich with an approximate population of 7,000. The sewage treatment plant, which provides secondary treatment of sewage, discharges to the shores of Lake Huron south of Goderich Cove Beach. There is a constructed groin extending approximately 340 m from the shoreline immediately north of the STP discharge

which serves to focus shoreline waters offshore when lake circulation is towards the north (see Figure 66). The Goderich STP is the only treatment facility between Point Clark and Bayfield that discharges directly to the shores of Lake Huron.

Table 9: Design features of the Goderich Sewage Treatment Plant, annual effluent flows and *E. coli* levels in final effluent for the years 1992 to 2002 and mean monthly flows and *E. coli* levels for 2003.

Goderich STP final effluent quality rated capacity: 9050 m ³ /day connected population: 7340 treatment: secondary, chlorine disinfection		Receiver: Lake Huron Distance to Lake Huron: 0 km, outfall at shoreline
YEAR	Average flow (m ³ /day)	<i>E. coli</i> (cfu/100 mL) geometric mean
1992	8700 (max 13,300)	n/a
1993	7440 (max 11,400)	n/a
1994	7490 (max 12,380)	n/a
1998	6993	65
1999	6546	114
2000	7333	43
2001	8710	49
2002	6785	343
Jan 2003	5217	369
Feb	5463	316
Mar	8020	58
Apr	9566	80
May	9720	220
June	8592	905
July	5942	119
Aug	5227	149
Sept	6467	41
Oct	8503	68
Nov	10960	105
Dec	10435	141
2003 annual	7843	163
2000-2003	Dry weather flows, 93 samples	103
2001-2003	Wet weather flows, 9 samples	172

In 2003, bacterial levels exceeding (200 cfu/100 mL) in the final effluent were addressed by increasing chlorine residuals and by including the storm overflow tank as part of the final effluent chlorine contact chamber to increase contact time. Grab samples taken during high flows are now included in the monitoring of final effluent quality which decreases overall effluent quality.

Loading of incompletely treated sewage to the lakeshore during primary and secondary bypass events have occurred in recent years (Table 10). Primary and secondary bypasses discharge via the STP effluent discharge canal to the shores of Lake Huron. Secondary discharges are chlorinated (B.M. Ross and Associates Ltd. 2004c). There were relatively few bypass events in 2002 and 2003 presumably reflecting the improvements in sewer infrastructure in recent years.

Table 10: Number of days experiencing primary and secondary bypasses and the volumes of the bypasses at Goderich Sewage Treatment Plant from 1996 to 1993 (B.M. Ross and Associates Ltd. 2004c)

Year	Primary		Secondary	
	Number of days	Volume (m ³)	Number of days	Volume (m ³)
1996	38	26,594	50	17,446
1997	16	23,147	26	22,318
1998	17	21,002	26	28,404
1999	7	6,979	10	10,119
2000	17	33,688	25	22,987
2001	7	19,846	10	30,615
2002	1	298	0	0
2003	2	665	1	157

Currently, the town is nearing completion of its sewer separation program to eliminate CSOs. From 1990 through 2003, the length of combined sewers has been reduced from 28.6 km to 0.6 km, which now represents 1% of the sewer system. All of the remaining combined sewers are routed to the sewage treatment plant (B.M. Ross and Associates Ltd. 2004c).

There are a number of locations where points of inter-connection exist between the sanitary sewer and storm sewer system. The purpose of these overflow points is to allow relief of the sanitary or combined sewers during periods of high flow. As part of a testing program to examine the functionality of the interconnections, it was found that direction of flow was from the sanitary system to the storm system at one of 16 connections and possibly in that direction for an additional three connection points (B.M. Ross and Associates Ltd. 2004c)

Due to the high concentrations of *E.coli* (presumably other microorganisms) in primary and secondary bypasses such bypasses represent a potential source of impact on recreational water quality at nearby beaches. Mean levels of *E.coli* in primary and secondary bypasses

were 9.3×10^5 and 1.4×10^5 cfu/100 mL, respectively, over samples collected between 1998 and 2003 (B.M. Ross and Associates Ltd. 2004c).

As part of a pollution control plan by the town of Goderich to meet requirements of MOE procedure F-5-5 an analysis was undertaken to examine magnitude of sources of *E. coli* to Lake Huron in the Goderich area including, WPCP effluent, WPCP bypasses, storm sewers and the Maitland River (B.M. Ross and Associates Ltd. 2004c). Contribution from the 3.5 km² of area drained by three storm sewers to the lake were estimated based on limited data on *E. coli* levels collected between May 30 and June 26, 2003. It was assumed that 40% of rainfall would be discharged at storm sewer. Not surprisingly, the rough “order of magnitude” analysis suggested that the load from the Maitland River far outweighed other sources. Considering the years 2000 to 2002, estimated loads from plant effluent, primary bypasses, secondary bypasses and storm sewers (directly to lake) were of the same order of magnitude with the exceptions of the primary bypasses in 2000 which was an order of magnitude higher than the other sources.

Wingham STP:

The STP for the community of Wingham is one of six STPs in Huron County which discharges to the Maitland River. For these plants the discharge plume of the river mediates the delivery of any potential contaminant load from the STPs to the shoreline of Lake Huron.

The average monthly levels of *E. coli* in the treated effluent in 1999 to 2003 are generally low, however, there were an isolated number of estimates exceeding 1000 cfu/100 mL suggesting that in the past there have been periodic difficulties in achieving proper effluent disinfection.

Based on information from four of the last five years, bypassing of effluent does not appear to be a frequent problem. Over the years 1999 to 2002 there were two bypass events, June 13, 2000 and February 10, 2001, respectively. Both events occurred at times of heavy rains and lasted from 1 to 2.5 hours. The volumes bypassed were 90 and 600 m³, in 2000 and 2002 respectively.

Table 11: Design features of the Wingham Sewage Treatment Plant, annual effluent flows and *E. coli* levels in final effluent for the years 1999 to 2002.

Wingham STP final effluent quality Rated Capacity: 3400 (m ³ /day) Connected population: 2900 Treatment: secondary, UV disinfection		Receiver: Maitland River Distance to Lake Huron: 80 km
	Average flow (m ³ /day)	<i>E. coli</i> cfu/100mL , geometric mean, biweekly sampling
Jan, Feb, Mar 1999		8, 11, 0
Apr, May, June		3, 2, 38
July, Aug, Sept		60 (max 1820), 22, 9
Oct, Nov, Dec		47, 40, 1400
1999 annual	2212 (max 5950)	31
Jan, Feb, Mar 2000		2, 157 (max 12,400), 11,024
Apr, May, June		597, 297 (max 4400, UV system problem), 20
July, Aug, Sept		20, n/a, 1
Oct, Nov, Dec		2, 11, 6
2000 annual	2679 (max 9637)	58
Jan 2001	2253	10
Feb	4341	386
Mar	3857	108
Apr	4341	15
May	2156	2
June	2027	3
July	979	31
Aug	1457	1
Sept	2405	2
Oct	3653	2
Nov	3370	1
Dec	3349	2
2001 annual	2849 (max 10,746)	73
2002 Jan to May	3081 (max 9172)	2
Sept		UV bulbs replaced due to final effluent <i>E. coli</i>

Brussels STP:

The Brussels STP discharges to the middle Maitland River some 90 km upstream (river flow path). The final effluent is disinfected for the months of May through October in accordance with the facility's 1980 Certificate of Approval. In 2002 and 2003, during the months when

disinfection was not part of the sewage treatment operation, *E. coli* counts in the effluent reached levels as high as 19,400 cfu/100 ml. Year-round chlorination was approved by the municipality to begin in the fall 2004.

There has been routine reporting of bypasses and overflows since the last inspection in 2001. No bypasses or overflows were reported from August 2001 to February 2004. The last overflow was February 9, 2001, with a duration of 7 hours and an estimated bypass volume of 2000 m³.

Table 12: Design features of the Brussels Sewage Treatment Plant, annual effluent flows and *E.coli* levels in final effluent for the years 2002 to 2003.

Brussels STP final effluent quality Rated Capacity: 880 m3/day Connected population: 1250 Treatment: extended aeration, tertiary filtration, chlorine disinfection			Receiver: Middle Maitland Distance to Lake Huron: 90 km	
	2002		2003	
	Average flow m3/d	<i>E. coli</i> cfu/100mL (single sample)	Average flow m3/d	<i>E. coli</i> cfu/100mL (single sample)
Jan	661	125	499	5100
Feb	1058	2400	537	9100
Mar	836	15000	910	400
Apr	698	12000	691	19400
May	646	6	824	56
June	570	16	681	16
July	504	4	572	2
Aug	527	0	607	1
Sept	472	7	447	5
Oct	503	2	730	10
Nov	477	1200	835	1480
Dec	540	5400	770	5400
average	620	3013	675	3414
1998 to 2003	462-680			
1998 to 2003	910-2180 (daily maxima)			
Feb 10 2004		6600 cfu/100mL (MOE sample)		

Blyth STP:

The Blyth STP discharges to Blyth Brook a tributary of the Maitland River and is 40 km upstream (river flow path) from Lake Huron. It is the STP nearest to the shores of Lake Huron on the Maitland River system.

The year-round effluent *E. coli* limit of 200 cfu/100 mL and biweekly sampling came into effect April 15 2003. The 2003 Annual Report identified that some of the monitoring samples collected in August 2003 exceeded 200 cfu/100 mL and noted that the suspected cause was a low residual chlorine concentration. The geometric mean for August 2003 was 319 cfu/100 mL and a high of 1520 cfu/100mL was observed on August 29. The chlorine dosage was subsequently increased to prevent a similar reoccurrence and compliance with the limit, which was based on the analysis conducted by the operator, was achieved for the remainder of 2003.

From September 2001 to April 2004, two chlorinated bypasses were reported while no overflows were reported during this period. The first bypass was a secondary bypass on February 21, 2002 that lasted 24 hours and discharged 1,346 m³ of bypass effluent. A sample of this effluent contained a high level of *E. coli* (52,000 cfu/100mL). The second bypass, also a secondary bypass, occurred from March 5 to 8, 2004 lasting 71 hours and releasing 4855 m³.

Table 13: Design features of the Blyth Sewage Treatment Plant, annual effluent flows and *E.coli* levels in final effluent for the years 2001 to 2003.

Blyth STP final effluent Rated Capacity: 730 m ³ /day Connected population: 1200 Treatment: extended aeration with tertiary filtration, sodium hypochlorite disinfection				Receiver: Blyth Brook Distance to Lake Huron: 41 km		
	2001		2002		2003	
	Av flow m ³ /d	E. coli cfu/100mL	Av flow m ³ /d	E. coli cfu/100mL	Av flow m ³ /d	E. coli cfu/100mL geomean
Jan	452	53	486	5,600	407	720
Feb	817	Na	620	718	425	108
Mar	588	39,999	562	1	576	600
Apr	578	6	497	700	484	2
May	394	2	459	2	511	20
June	371	8	337	2	442	55
July	344	6	383	2	411	2
Aug	361	8	375	16	379	319
Sept	407	17	365	6	363	20
Oct	541	Na	368	28	458	64
Nov	474	242	396	60	516	1
Dec	424	2	439	2	532	3
Av daily	479		440		459	
Max daily	3388		1944		1212	

Lucknow STP:

The Lucknow STP discharges to an infiltration bed and is not directly connected to any watercourse discharging to the shores of Lake Huron. This STP is located in the drainage area of the Nine Mile River and 27 km from Lake Huron.

There has been routine reporting of bypasses and overflows. At the time of an inspection in March 2004 a bypass was occurring at the plant. The bypass flow was diverted to the emergency lagoon. Observations indicated that the lagoon was permeable and that the effluent was percolating through the bottom of the lagoon. The down gradient flow of surficial ground water is toward the McLeod Drain.

Table 14: Design features of the Lucknow Sewage Treatment Plant, annual effluent flows and *E. coli* levels in final effluent for the years 2001 to 2003.

Lucknow STP Rated capacity: 750 (m ³ /day) Connected population: 1300 Treatment: three aerated facultative lagoons and one emergency lagoon. Effluent from the lagoons is directed to six rapid infiltration basins. Disinfection not required.		Receiver: subsurface discharge to McLeod Drain to Lucknow River Distance to Lake Huron: 27 km
YEAR	Average flow (m ³ /day)	<i>E. coli</i> (cfu/100 mL)
2001	556 (max 3127)	<100
2002	535 (max 1376)	<100
2003	596 (max 2042)	<100

Vanastra STP:

The Vanastra STP is one of four STPs in the drainage area of the Bayfield River. The plant is 18 km from the Lake Huron.

An inspection by the MOE in September 1998 noted the facility had a problem with bypassing with a generally unknown quantity of raw sewage being discharged. Frequent by-passing occurred during rainfall and spring thaw events due to water infiltration into the sanitary sewers. This is indicated by the fact that some of the maximum daily flows (Table 15) are greater than 2.5 times the hydraulic design capacity of the plant. Notes on bypass events for the years 1995 to 1999 are provided in Table 16. The quantities of sewage bypassed generally are within the criteria established by MOE Procedure F-5-5 (Ontario Ministry of the Environment and Energy 1994b).

The facility was issued a Certificate of Approval on March 12, 2003 which requires weekly grab samples for *E. coli* from May 1 through October 31. The facility is over 40 years old, pre-dating the issuance of Certificates of Approval. The site was inspected in June 2003 and assessed for compliance from 2000 through March 2003 using Ministry Guidelines F-5 (Ontario Ministry of the Environment and Energy 1994c) and F-10 (Ontario Ministry of the Environment and Energy 1994d).

Table 15: Design features of the Vanastra Sewage Treatment Plant, annual effluent flows and *E.coli* levels in final effluent for the years 2001 to 2003.

Vanastra STP final effluent quality		Receiver: Grant Creek to Bayfield River
STP rated capacity: 1405 (m ³ /day)		Distance to Lake Huron:
Connected population: 800		18 km
Treatment: secondary with chlorine disinfection.		
YEAR	Average flow (m ³ /day)	E coli (cfu/100 mL) single monthly samples
1998		48 (MOE sample Sept 1)
2000	866 (max 5951)	
2001	940 (max 6190)	
Jan 2002		
Feb	1169 (max 2609)	21
Mar	1070 (max 2588)	8
Apr	995 (max 2254)	6
May	831 (max 1830)	1
June	661 (max 1580)	2
July	566 (max 1588)	1
Aug	493 (max 793)	2
Sept	383 (max 473)	14
Oct	523 (max 1350)	7
Nov	667 (max 1060)	80
Dec	737 (max 2122)	12
2002	760 (max 2609)	
2003		69

Table 16: Notes on bypasses at the Vanastra STP for the years 1995 to 1999.

Year	Notes
1995	363 hours of bypassing reported
1996	no data
1997	12 bypasses; 92 hours total duration
1998	217 hours of bypassing reported
1999	8 bypasses totaling 122 hours duration; volume was 4,200 m ³ ; the largest single event was 3650 m ³ over 51 hours in January ; five of the events were less than 50 m ³ ; abnormally dry year

From 2000 to 2003 fifty-one bypasses were reported. The bypasses are normally chlorinated by virtue of the bypass being combined with final effluent which is then subjected to increased chlorine dosage for the bypass period. No overflows were reported for the period. During the period of 2000 through February 2003 adequate records were maintained related to bypass times and flows, however, the requested analysis of *E. coli* was not consistently conducted. Bypass monitoring was required starting March 12, 2003. A review of the analytical results for bypass events since then indicates that this requirement still is not being consistently met. Of eight bypasses reported in this period, *E. coli* was not analyzed in 4 of these instances.

Table 17: Numbers, durations and flow volumes for bypass events at the Vanastra STP for the years 2000 to 2003.

Year	# of Bypass Events	Average Duration (h)	Average Flow (m ³)
2000	22	21.7	730
2001	6	46	928
2002	13	15.49	443
Up to May 2003	10	21.7	224

Table 18: Numbers, durations and flow volumes and *E.coli* levels for bypass events at the Vanastra STP for individual bypass events in 2002.

2002 bypass reports			
Date	Duration (h)	Volume (m ³)	<i>E. coli</i> (cfu/100mL)
Jan 14	25.75	186	1, 980, 000
Feb 20	38.0	2617	18,400
March 3	12	158	
March 9	22	430	
April 8	15.62	325	7100
April 13	12	350	
May 17	14	806	
June 4	12	217	4800
July 22	3.5	68	
August 28	13.5	268	
August 29	2.0	15	
August 31	4.5	44	7200
December 19	26.5	270	

Clinton STP:

The Clinton STP discharges to the Bayfield River 23 km from the lake, only slightly further upstream than the Vanastra STP.

This plant's effluent consistently met the established limit for *E. coli* in 2001-2003. Under its Certificate of Approval, *E. coli* levels in the effluent must meet a limit of 200 cfu/100 mL (based on the monthly geometric mean) during the period of May 1 to October 1. The municipality is disinfecting the effluent on a year round basis and consistently meets a more stringent objective of 150 cfu/100 mL for *E. coli* in the final effluent.

There has been routine reporting of bypasses and overflows. Bypasses are not normally disinfected. Of the four reported bypasses from 2001 to 2003, three were from the holding lagoon and were related to excess flows to the facility during the February- March period of each year. Bypass duration averaged 56 hours and 4625 m³ volume. In 2004, Clinton was ordered to submit an action plan to reduce stormwater infiltration into the sanitary sewer system.

Table 19: Design features of the Clinton Sewage Treatment Plant, annual effluent flows and *E.coli* levels in final effluent for the years 1998 to 2003.

Clinton STP final effluent quality Rated capacity: 3100 m3/day Connected population: 3500 Treatment: extended aeration with tertiary filtration and UV disinfection				Receiver: Bayfield River Distance to Lake Huron: 23 km		
	E. coli cfu/100mL geometric mean, biweekly samples					
	1998	1999	2000	2001	2002	2003
Jan	6300	2700	4300	1	0.5	0
Feb	1805	2000	22,200	1	16.5	6
Mar	456	8900	5050	1.5	1	20.5
Apr	3670	7155	105	1.5	0	26.6
May	3	7	3	1	0.3	3.5
June	4	7	86	1	0	0
July	3	0	1	1	0	0
Aug	1	0	1	1	0	0
Sept	0	0	1	1	0	1
Oct	0	0	1	3.6	0	3.6
Nov	10300	2263	1	1	0	4
Dec	35300	6566	2	1	0	3.5
Effluent daily flow						
Average flow (m3/day)				2567.00	1930.00	2133.00
Maximum flow (m3/day)				6202.00	3421.00	5701.00

Bayfield STP:

The community of Bayfield has recently switched from private septic systems to a communal STP which was approved June 14, 2000. The treatment system consists of a twin-celled sewage lagoon system with intermittent sand filtration, designed to treat an annual total sewage volume of 391,186 m³ with effluent discharge during the nonfreezing season to the Bayfield River about 6 km from the lake, with a daily capacity of 1072 m³/day. The plant must meet a monthly effluent limit of 200 *E.coli* cfu/100 mL based on weekly samples.

A filter bypass occurred March 5 to 17, 2004 due to a high flow of raw sewage being received following a snow melt and rain event. Two samples of the bypass showed *E. coli* readings of

14,100 and 5,100 cfu/100 mL. The municipality was ordered to conduct a study to identify and abate the source of the extraneous sewage flow.

Seaforth STP:

The Seaforth STP discharges via a drain to the Bayfield River 40 km from Lake Huron, and is the most upstream of the four STPs discharging to the Bayfield River in Huron County.

Table 20: Design features of the Seaforth Sewage Treatment Plant, annual effluent flows and *E.coli* levels in final effluent for the years 1999 to 2002.

Seaforth STP final effluent quality Rated capacity: 1990 (m3/day) Connected population: 2300 Treatment: secondary plus Sutton process polishing lagoons			Receiver: Crozier drain to Bayfield River Distance to lake Huron: 40 km	
	E. coli cfu/100mL			
	1999	2000	2001	2002
Jan	18	19	993	<1
Feb	7	136	1993	25
Mar	1	69	486	<1
Apr	3	6	2	<1
May	3	4	<1	<1
June		9	<1	<1
July		6	0	<1
Aug		8	0	<1
Sept		5	0	<1
Oct		12	4	<1
Nov		42	0	<1
Dec		352	1	<1
	Av. daily flow m3			
annual	1405	1811	1750	2096

Since 1994 effluent has been disinfected year round by UV and must meet a monthly effluent limit of 200 *E. coli* cfu/100mL based on weekly samples.

Information on bypasses was found for selected years. There were no bypasses in 1994 and 1999 and one event in 2001. However, in 2000 there were six events of a total duration of 76 hours. Five events were in May and the last event occurred in June.

5.2 Other communal sewage works

Several other locations were identified where communal sewage treatment systems, other than STPs, are being used for sewage and waste water disposal. Communal septic systems are used for sewage disposal at several campgrounds and trailer parks (Table 21).

Table 21: Features of communal septic systems and other non-STP modes of communal sewage disposal in Huron County portions of the drainage areas from Bayfield to Amberley Beach (list is not inclusive).

Facility	Approximate Location	Type	Units Served
Riverside Park	Nine Mile River at Highway 21	< 10 m ³ /day	100 sites
Happy Hollow Campground	Nine Mile River at Port Albert	tile beds	100 sites
Lake Huron resort	8 km north of Goderich	spray irrigation	160 sites
Shelter Valley Campground	8 km east of Goderich on Maitland River		160 sites
Princess Huron trailer park	3 km south of Goderich, lakefront	tile beds	85 sites
Kitchigami Campground	10 km south of Goderich	tile bed	80 sites

At Atwood Cheese located in Perth County, effluent is spray-irrigated in the summer (approximate volume 11,000 to 15,000 m³ per year). This method does not require bacteriological analysis. Overall, annual reports show that this site has no impact on water quality (nutrients and organic material) in the Turnball Drain. There likely additional sites where communal sewage works are in operation which have yet to be identified.

The degree to which STP discharges, spills and bypasses contribute to adverse beach water quality cannot readily be determined from existing information, however, the potential significance of these point sources to beach water quality should not be overlooked. Recent risk-based approaches to managing recreational water quality attach particular significance to sources where fecal material are of human origin (WHO 1999, 2003). Goderich is the largest urban centre in Huron County and is arguably the largest human fecal point source over the study area. The close physical proximity of the STP outfall and storm sewer outfalls to the public beaches suggests that loading from the town's sewer systems should be an ongoing focus of evaluation in conjunction with assessment of water quality at the Goderich Beaches.

The information presented here indicates that quality and quantity of bypass and overflow events are frequently poorly defined. Given the erratic and short duration of high loads and levels of fecal pollutants associated with these events any evaluation of these sources as drivers of adverse water quality at beaches will likely require good characterization and reporting of the events.

6.0 Agriculturally-based Activities as Sources of Microbial Pollutants of Fecal Origin to the Huron County Shoreline of Lake Huron

6.1 Livestock Commodities in the Huron County Study Area

Livestock on farms in the study area are hosts to many microorganisms, including *E. coli*, which is monitored as a fecal pollution indicator. Research from other regions suggests that the quantity, rates and timing of the shedding of these bacteria in manure can be expected to vary between livestock commodities and even between farms with the same type of animals. It is important to understand which livestock groups are present, in what numbers and managed under what practices in order to assess their potential impacts as sources of fecal pollution at the study area beaches. To date, no detailed study has determined the extent to which area agricultural operations may be linked to the recurrent recreational water quality impairments at these Lake Huron beaches.

Huron County is one of Ontario's leading livestock producing regions. Swine, cattle and poultry are most prominent in this diverse livestock commodity base, followed by sheep and horses which are also a significant farm livestock. Livestock numbers on farms proximal to the Lake Huron beaches and within the watersheds associated with these beaches can be determined. Federal agricultural census data, although collected for individual farms, is generally aggregated and reported at the county scale. These data, when provided at the "quaternary watershed scale", also aggregate more land area than occurs within the individual catchments associated with the beaches of study. To date a characterization of livestock over individual catchments on a beach by beach basis (as in 2.4.10) has not been possible. Livestock agricultural census data

for Huron County are provided in Table 22. But note that the drainage area of some of the tributaries discharging to Lake Huron over Huron County extends beyond Huron County. Over the most recent decades, some livestock farms in Ontario have experienced substantial changes while others have not changed very much at all. The newest swine, poultry and dairy farms have generally expanded while many of the small beef farms have operated in much the same way over this time period in Huron County. It could be argued that the proportion of livestock manure in the county generated by these smaller farms has decreased (even though the total amount on these farms may have remained quite static) while the fraction produced on larger, confined animal farming operations has increased over this time period.

Table 22: Livestock Agricultural Census Data for Huron County (Statistics Canada, 2001)

	Number of Livestock and Poultry and Corresponding Nutrient Units					
	1981		1991		2001	
	#	NU	#	NU	#	NU
Bulls: 1 year & older	1,556	1,556	1,100	1,100	1,076	1,076
Dairy Cows: calved at least once	24,671	41,118	19,221	32,035	17,746	29,577
Dairy Heifers: 1 year & over	10,755	3,585	9,187	3,062	9,054	3,018
Beef Cows: calved at least once	16,005	16,005	15,523	15,523	18,320	18,320
Beef Heifers: breeding 1 year & over	3,514	1,171	3,309	1,103	4,506	1,502
Beef Heifers: slaughter 1 year & over	38,281	12,760	22,290	7,430	23,207	7,736
Steers: 1 year & over	64,553	21,518	30,968	10,323	38,809	12,936
Calves: under 1 year	31,162	-	31,050	-	34,817	-
Total Cattle	190,497	97,714	132,648	70,576	147,535	74,165
Sows & Boars: 6 mo & over	41,451	12,448	40,680	12,216	71,046	21,335
Pigs: under 20 kg	108,887	5,444	115,838	5,792	-	-
Pigs: under 20 to 60 kg	105,738	5,287	115,294	5,765	258,847	12,942
Pigs: over 60 kg	87,238	14,540	99,876	16,646	300,423	50,071
Total Pigs	343,314	37,719	371,688	40,419	630,316	84,348
Sheep: 1 yr & over	6,293	787	6,750	844	11,186	1,398
Lambs	4,917	-	5,123	-	9,345	-
Total Sheep & Lambs	11,210	787	11,873	844	20,531	1,398
Horses: on farm	2,347	2,347	1,888	1,888	2,679	2,679
Goats	2,117	265	1,503	188	4,014	502
Hens: over 20 wks intended for laying	777,613	5,184	746,597	4,977	1,013,476	6,757
Pullets: over 20 wks intended for laying	501,777	1,004	484,928	970	669,281	1,339
Other Chickens	1,632,485	5,503	1,976,924	6,664	3,519,474	11,863
Total Chickens	2,911,875	11,690	3,208,449	12,611	5,202,231	19,958
Turkeys	91,520	1,028	170,709	1,918	231,365	2,600
	Total NU	151,549	Total NU	128,443	Total NU	185,650
	Compared to 1981			-15%		23%
	Compared to 1991					45%

Source: Statistics Canada, Census of Agriculture

6.2 Manure Form and Methods of Environmental Release

Animal excreta, including that from livestock, are important parts of natural and agricultural ecosystem functioning and biogeochemical cycling. The microbial communities present in this

material, and the microbes that act on the material after its environmental release, are subject to a number of “environmental stresses”. Livestock manure management practices can greatly influence the persistence (survival and fate) of these microorganisms because they each influence the stress factors to different degrees. These practices, therefore, can determine how and whether or not manure *E. coli* (and other microorganisms) from any given farm can eventually cause or contribute to local stream or beach water quality impairments. A systematic examination of some of these livestock manure management practices is given below.

6.2.1 Manure Forms on Lake Huron Shoreline Region Farms

Solid Manure:

Several decades ago, manures from all livestock commodities in Ontario were handled in “solid form”. Management of this manure form involves its storage in piles and the subsequent mechanical spreading of it onto fields. Solid manure is the form currently found on poultry, beef, sheep, horse and some dairy farms within the study area.

Manure piles, or “solid manure storages”, have a number of management factors that greatly influence the persistence of their particular microbial communities. Once shed from their livestock hosts, these microbial communities experience moisture and temperature related stresses in the manure piles. Some piles are formed with daily increments added by either loader buckets or elevators to form conical or berm-like piles. Manure from each of these daily additions is typically exposed directly to air, light and sometimes precipitation (rain or snow) on the “pile face” until covered by future new additions. Exposure time to these environmental stresses and their impact on microbial populations would be expected to vary greatly. Re-inoculation (addition) of the entire manure pile with new microbes would not be expected to occur in this handling system because no thorough mixing of the entire pile is involved. This solid manure management system is used for dairy and horse manure management in the study area.

Some solid manure management systems involve storage pile formation by way of a series of short-term “clean-out” events. Typically, under this manure management system, the animals occupy the same barn or yard location as their bedding and wastes. They remain in this

situation until relocated to different housing (as with beef cattle or sheep in a yard before they are released to pasture) or removed from the farm (as in an “all-in-all-out” poultry operation). Layers of bedding and wastes, which accumulated over time, are then mechanically scraped with a loader and added to a storage pile. Manure age at any location in these piles is highly variable and consequently so is the exposure to thermal and moisture stresses that impact microbial population communities. This handling system is used for poultry, beef cattle and sheep manure management in the study area.

Liquid Manure Systems:

Huron County was one of the first locations for the introduction of “liquid manure” systems for handling swine and dairy manure in Ontario. This manure handling technology uses water to facilitate the movement of manure, in slurry form, away from the animal housing areas and into storage “tanks”. The appeal of this manure handling system comes from improved animal hygiene and reduced farm labour. Management of this manure form involves its storage in tanks or lagoons and subsequent pumping into tankers for surface spreading or injection on land. Liquid manure systems have become the most prominent method of manure management for the larger, confined animal swine and dairy farms in the study area.

Liquid manure storages have either continuous or “very regular” additions of new manure. The regular additions management practice is utilized when an “under barn” temporary storage occurs before subsequent pumping to the external storage. This practice is common in farms where animals stand on slatted floors through which their waste falls to pits below. Manure microbes remain in a liquid environment for the entire route from the animal to the storage and then through the spreading process in liquid manure systems. The continuous addition of “fresh” (recently excreted) manure to the storage tank from the livestock encourages re-inoculation of the entire storage with newly shed microbes. More research is required to understand microbial behaviour and to develop knowledge about population die-off rates in these storages under Ontario (and Huron County) climatic conditions.

6.2.2 Manure from Confined Animal Feeding Operations

The climate of the Lake Huron shoreline and connected upland watershed regions has challenged farmers to develop livestock management practices that involve some level of animal confinement – especially during the winter season. Some commodities have moved to confined animal feeding operations year round – mostly for swine and poultry, others have a mix of “free-range” or pasture time – dairy cattle, beef cattle, sheep and horses. Systems with “complete” or year-round confinement result in all manure being collected and managed in storages until it is evacuated for land application. Exposures of the manures in these storages to air and sunlight or any treatment technologies can be controlled. This provides both challenges and opportunities for managing the potential environmental impacts of the microbial community shed from these animals. The other systems, such as grazing on pastureland, involve animal confinement for only part of the year and, as such, only part of the manure produced is “captured” and controlled in storages. More research is required to better understand the impacts of storage time and conditions and any “in-storage” treatments on microbial loads in livestock manure prior to land application.

6.2.3 Manure on Barnyards and Feedlots

Barnyards and feedlots on farm properties are areas where animals drop feces directly onto the ground or yard floor and where there is often direct environmental exposure of the manure. Accumulations of manure are handled very differently on different farms. For example, some barnyards on beef farms may have manure accumulating throughout an entire winter and early spring period before any clean out is done. Other farmers may scrape up accumulated manure from yard or feedlot areas into piles on a regular basis. The manure dropped in these areas is exposed to a range of sunlight, thermal and moisture conditions over the time it lays in the barnyard. Microbial communities in the manure are subjected to these environmental stresses and their numbers should be impacted accordingly, however, no studies have been conducted on barnyard and feedlot manure microbial communities within the study area to confirm this assumption.

Migration of manure microbes off-site from barnyards and feedlot locations is of concern for water quality protection in Ontario. Runoff waters, especially from sloping sites, are potential conveyors for microbial transport and resulting impairments of surface waters. Farmers have been encouraged to manage stormwater and snowmelt water runoff from these locations, and the use of berm structures and vegetated filter strips are among the recommended runoff management practices.

Historically, barns and their yards and feedlots were built in close proximity to streams throughout Ontario – Huron County farms in the study area are no exception. The travel distance for runoff waters by way of overland flow or in channels that originate on or near these locations is therefore quite short. Runoff that has been in direct contact with manure on these sites can deliver significant microbial loads to the streams.

6.2.4 Manure on Pasture Land

Farms for several livestock commodities make use of part of their land base for grazing animals on pastures for part of the year. Beef cattle, dairy cattle, sheep and horses are commonly grazed on Huron County pastureland. Pastured animals drop feces directly onto the ground where it is exposed directly to the environmental conditions at that time. Sunlight exposure and the thermal and moisture conditions change constantly from the time this manure is dropped. A suite of indigenous organisms actively decomposes the manure in the field and these manure microbial communities experience environmental stressors and their numbers should be impacted accordingly.

Migration of manure microbes off-site from pastures has potential for surface water quality impairment. Runoff waters that have been in direct contact with manure, especially on sloping fields, are potential conveyors for microbial transport. However, the density of animal droppings on the pasture land base is very low compared to that of barnyards and feedlots. Areas where animals both regularly congregate and defecate would have a higher likelihood of yielding higher microbial loads in local runoff waters than pasture lands.

Livestock Access to Water Sources:

Streams with fertile “bottomland” areas were considered prime assets for livestock farms when Ontario farms were first established. Pasture areas with direct stream access sites, where animals could get water, were common features of the rural landscape. Awareness of negative impacts of this livestock management practice on stream water quality and aquatic habitats has changed this aspect of livestock farming and federal fisheries law prohibits direct livestock access to surface water bodies.

Subsection 36(3) of the Federal Fisheries Act (1985) is pertinent to addressing microbial impairments of streams from livestock. It states that “no person shall deposit or permit the deposit of a deleterious substance of any type in water frequented by fish or in any place under any conditions where the deleterious substance or any other deleterious substance that results from the deposit of the deleterious substance may enter any such water.” Environment Canada has enforcement responsibilities for this pollution prevention provision in the Fisheries Act. Manure is considered to be a “deleterious substance” and farmers are not to “permit” their livestock to “deposit” it in streams. The latter part of this subsection also covers the spreading or spillage of manure on land near watercourses.

Some stream reaches within the study area were observed to have fenced riparian zones to prevent livestock from the proximal pastureland from directly accessing the water. However, not all pastures along stream reaches are fenced to prevent direct livestock water access and at present, no inventory of “livestock access” situations has been completed for the catchments in this Lake Huron shoreline study area.

6.2.5 Manure Applied to Cropland

Beneficial use of manure for crop production has long been part of agricultural practices in Huron County. Manure is applied to land at specific rates (mass or volume per area) to achieve both plant nutritional and soil quality benefits. Different crops require different nutrient inputs to achieve maximal yields. Manure is typically applied in the study area after crop harvest in late summer or into the fall or prior to crop planting in the early spring. Crops are rotated from field

to field within a farm property as part of a land stewardship strategy that maintains and enhances soil quality and controls plant disease. A result of this crop rotation practice is that only some of the fields on each farm receive manure in any given year. Manure microbial loadings from these land applications, therefore, vary both in time and space.

Surface Application of Solid Manure to Cropland:

As was mentioned previously, solid manure from several livestock commodities has been historically and continues to be produced and applied to land in the study area. The technology for field delivery involves loading the manure from the pile into open-topped manure spreaders that are pulled and powered by tractors as they traverse the fields. Solid manure spreaders generally operate with a conveyor bed that moves the manure to the rear of the machine where rotating blades fling the manure onto the field. The operation of the spreader and the speed of the tractor both control the rate and, to some extent, the uniformity of the manure application (tonnes/hectare or tons/acre) across the field. The environmental stresses of sunlight exposure, soil surface thermal conditions and soil moisture status all can contribute to reductions in manure microbial populations in this land applied solid manure. Overall, one can assume that microbial loads per aerial unit across the fields in the study area from solid manure applications are highly variable.

Surface Application of Liquid Manure to Cropland:

Liquid manure handling systems are used extensively on the swine and larger dairy livestock operations within the study area. As discussed earlier, these systems differ greatly from solid manure systems in terms of expected microbial persistence potential in the manure itself. The “continuous feed” technology of liquid manure storages means that most recent additions have the opportunity to re-inoculate the older manure in the entire storage each day. The spreading of “today’s manure tomorrow” – complete with its recently excreted microbial load - is possible with this choice of manure handling system on study area farms.

The technology for field delivery of liquid manure has had several variations since it was first introduced. The manure in the storage tank is mechanically agitated and mixed in preparation for evacuation by pumping prior to field delivery. This process should produce a more

homogeneous distribution of the manure microbial community (of a range of ages) in the tank just prior to manure removal for land application. This is in marked contrast with the microbial load in the poorly mixed solid manure situation described previously. To date, no studies have been performed on liquid manure storages in the study area to examine their microbial populations.

Field applications of liquid manure in the study area have involved delivery from the storage to the field via pipelines or tanker wagons. High trajectory irrigation “guns” (sprinklers) were connected to these pipelines or to pipes/hoses from pumps that evacuated the tankers at the field. These “guns” were moved throughout the field to apply manure at specific rates on an aerial unit basis. Problems of spreading of manure on non-target (including directly into streams) areas and odour complaints have contributed to the discontinued use of this application method in the province. Tanker wagons have replaced the “guns” and pipelines as the dominant liquid manure delivery and application technology in the study area. These tanker spreaders are pulled and powered by tractors as they traverse the fields. They can either “dribble” manure from a “spray bar or boom” at the rear of the machine onto the field or “irrigate” manure a short distance from a rotating sprinkler onto the field. Again, the operation of the spreader and the speed of the tractor both control the rate and, to some extent, the uniformity of manure application (litres/hectare or gallons/acre) across the field.

Arguably, manure mixing during agitation, spreader loading and the actual spreading processes should result in a much more even distribution of both beneficial nutrients and the microbes on fields, compared to solid manure application methods. For any given field in the study area, however, one may assume that microbial loads per aerial unit from liquid manure applications may be greater and less variable than for solid manure applications. Cropland fields receiving liquid manure could be considered as much more area-concentrated sources for potential microbial water quality impairments than are the fields where solid manure is applied. To date, however, no detailed study has been conducted within the study area to critically test this assumption.

Liquid Manure Injection on Cropland:

The most recent technology trend in liquid manure application on cropland within the study area involves the direct injection of manure into soils from tanker wagons as they traverse the fields. The manure slurry is delivered to the soil in shallow furrows that are first opened and then

closed by mechanical shoes and wheels on the base of each injector arm. The spacing between arms on the injector boom and the flow rate from the injector nozzles combine with the tractor speed to determine the application rate. This technology has won favour among farmers for its ease of use, elimination of a subsequent incorporation step, compatibility with minimal tillage systems and reduction in odour from manure spreading.

The nutrient and microbial distribution from injected liquid manure at any point on the field has some noteworthy differences from that for surface broadcast methods. The “banding” of injected manure concentrates it in discrete rows rather than in a somewhat uniform surface coverage. Injected manure microbes may have left a covered manure storage tank (recommended for odour mitigation), been pumped into an enclosed manure tanker wagon and delivered to the soil beneath its surface through injector hoses – all with minimal opportunity for sunlight exposure or thermal or moisture status changes. As such, the persistence of manure microbial populations delivered to cropland through liquid manure injection methods may be considered to be greater than for those from surface applied methods. Cropland fields receiving injected liquid manure could be considered as more persistent sources for potential loading of microbial fecal pollutants to surface waters than fields where liquid manure is applied directly on the surface. To date, no studies have been conducted within the study area to test these hypotheses.

Nutrient Management Planning and Manure Application in the Study Area:

Nutrient management planning on farms in the study area and the rest of Huron County is in its first decade. Since 1997, a growing number of farms in the region have developed formal plans for managing the manure generated on their farms and the application of it and other nutrient-bearing materials to their fields. Ontario’s Nutrient Management Act (2002) (NM Act) will result in all farms within the study area eventually addressing their manure application with more formal plans and strategies. Current regulations under this Act do not specifically address manure microbial populations and their potential impacts on surface water quality. They do however, provide for limitations on the timing and rate of manure applications and setback distances for placement of the manure with respect to watercourses. Farms in the study area that actively follow their manure application strategies according to their nutrient management plans may reduce their potential to impact local streams with manure microbes. Some limited stream water quality monitoring data in the study area show lower *E. coli* values in a catchment dominated by a cluster of fields managed under these plans than for two adjacent watersheds of

similar size but with very few or no nutrient management plans for their fields. Further validation and verification of the NM Act regulations and the various plans produced by study area farmers to mitigate stream water microbial loadings and hence contributions to the beaches are needed.

6.3 Manure Microbial Persistence and Transport in the Study Area Landscape

Previous sections have described manure storage and land application technologies and their various implications for microbial population differences from land applications of manures from the various livestock commodities in the study area. The persistence and transport of these microbes after environmental release could also depend to some extent on the soils and cropping systems in this region and the inherent environmental tolerance of the bacteria. The study area landscape can be evaluated for relative soil loss – and inferred applied manure loss – under the current agricultural management practices.

6.3.1 Soils and Cropping Systems in the Study Area

Soils in the study area owe their origins to the glacial history of this region of Ontario. The eastern upland areas of the watersheds are hilly moraines with soil textures that range from silt loam to sandy loam. A feature known as the “Algonquin Bluff” marks the western margin of this region. It was the shoreline of a higher glacial lake level in the Huron Basin. To the west of this feature, the land is more evenly sloping towards the present Lake Huron shoreline. Soils of this region are generally shallow sands over a clayey (St. Joseph’s) glacial till unit.

Pastureland and hay fields may be somewhat more prevalent than cropland in the hilly areas in these watersheds. Dominant crops grown in the study area are corn, soybeans and wheat. They are commonly grown in rotation and in that order. Manure is applied generally to wheat land after harvest and prior to corn planting. Tillage operations in preparation for these croplands, vary between farms. A common scenario would have the land tilled after the wheat in preparation for planting the corn crop. Minimal-till or no-till planting of the soybeans after corn and wheat after the soybeans would follow in the next two cropping seasons. The more erodible soils would experience the greatest soil losses in the “corn year” part of this crop

rotation system. As this part of the cropping system also receives the manure application, this is also the time window for greatest risk of microbial impact by runoff waters from these fields. In addition to farming management practices, field slope and the continuity of delivery channel systems (rills and gullies) within fields are critical factors in the movement of runoff water and entrained sediment, manure and microbes off these fields and into proximal streams.

6.3.2 Bacterial Persistence in Study Area Agricultural Soils

Factors that potentially influence the field loadings of manure microbes on cropland were discussed in previous sections. Once delivered to the field soil, manure microbes are subjected to a number of environmental stresses – some a direct consequence of soil properties.

Soil temperatures affect bacterial persistence in agricultural soils. Soils in the study area experience a wide range of surface temperatures from below freezing to greater than 50°C during any given year. Higher temperatures reduce and even eliminate microbial populations near the surface. Alternatively, microbes that are moved deeper into the soil do not experience these extreme temperatures and may persist much longer.

Soil moisture varies between soil types in the study area and between cropping years. Coarse-textured soils like sands drain much more rapidly than finer-textured loam and clay soils. Manure microbe populations can be reduced dramatically by desiccation, since persistence of microbial communities in soils is critically dependent on the maintenance of adequate soil moisture levels.

No detailed field studies have been conducted on the persistence of manure microbial populations in agricultural fields within the study area. However, results from work in other regions with similar climates suggests that manure *E. coli* can persist in agricultural field soils well beyond the cropping season in which it was applied.

6.4 Microbial Transport from Agricultural Land in the Study Area

The potential for manure microbes to move from agricultural land to watercourses is central to the issue of the extent of any agricultural connections to beach water quality impairments in the study area. The natural processes of runoff, soil erosion and sediment transport from the land to streams contribute to the replenishment of the beaches at Lake Huron's shorelines. The magnitude of these contributions to this dynamic beach environment is unknown. However, tillage of land for crops in the study area provides more bare soil areas for erosion by water. These same processes would permit manure microbes to enter streams and make their way to the shores of Lake Huron.

6.4.1 Runoff Waters and Microbial Transport

Runoff from farm fields is generated any time water ponds on soil surfaces at rates faster than it can infiltrate the soil. It then moves laterally over the ground surface according to the slope of the terrain. This situation can happen when the soil reaches field capacity and is saturated in the near surface region as may occur during spring snow melt or at times of long duration, low intensity rains. It can also occur when rainfall intensity exceeds the infiltration rate during storm events. Once water has ponded on the soil surface, it can move laterally as "sheet flow" until it encounters small channels (rills) that are tributaries of larger ones (gullies) that feed permanent stream networks. Free-living microorganisms washed into flowing water from land applied manure have the opportunity to move with these runoff waters from farm fields.

6.4.2 Soil Erosion and Sediment Yield

Manure microbes, particularly bacteria like *E. coli*, can attach themselves to soil particles which can then be eroded and carried by runoff waters from the agricultural fields. The land management and cropping system practices described in previous sections of this report are key factors in evaluating soil erosion potential in this region. Soils with no vegetation on the surface for long periods of time and those planted with row crops like corn are at greatest risk of

erosion by water. Critical landscape factors for soil erosion by water include slope and soil texture. Areas with steeper slopes and loam to sandy textures are prone to rilling and gulleying as runoff waters incise these fields. The greater the amount of runoff generated on steeper sloping areas that have been plowed and manured in preparation for a corn crop in the study area, the more sediment yield there will be from these fields and the more potential there is for microbial loading to streams.

Predicting Runoff, Soil Erosion and Microbial Transport in the Study Area with the WEPP Model:

The Water Erosion Prediction Project (WEPP) model provides a “within-field-scale”, process-based approach to using landscape information, weather data and agricultural land and crop management system information to generate runoff water volumes, flowpaths and associated sediment yields from all farm fields in an entire catchment. This model has been applied to several small catchments within the study area. Preliminary results from this model have identified and mapped portions of study area fields expected to deliver the greatest runoff volumes and sediment yields to streams. Assuming that manure microbes will move off fields with either runoff waters or their entrained sediment loads, WEPP model results can also reveal field areas that potentially contribute most to microbial water quality impairments.

The WEPP model approach offers a way to inform a field-monitoring program of potential “hot spots” for stream water loading from erosion events. Once calibrated and validated for study area field conditions, WEPP model results can also be used to identify farm field regions where management practices could be changed to most dramatically reduce off-field water, sediment and microbe movement to streams. A field-based research program is required to evaluate the effectiveness of using WEPP for microbial water quality impairment mitigation within the study area.

6.4.3 Tile Drainage and Microbial Transport

Throughout most southern Ontario agricultural regions, tile drain systems have been installed in fields to better manage water in the soil profile – especially in the near-surface, crop-rooting zone. The cropland within the Lake Huron shoreline study area is no exception. Urban areas,

woodlots, areas of permanent pasture and riparian zones are generally not tiled within the study area.

Two systems of tile drains have been used in Huron County. These are random and systematic. Regular “tile run” spacing within the field characterizes the latter. Tiles are generally installed at depths of between 60 and 120 cm below the soil surface. These tiles intercept percolating soil water and control water table rise in the lower rooting zone. They permit land to be worked, planted and traversed for harvesting at times when soil moisture conditions would normally limit such activities. Drainage waters are delivered through these tiles to agricultural drains which, in turn, drain to streams.

Minimal tillage or “no-till” systems encourage the establishment of structural voids called “macropores” in the soil profile. Macropores are formed naturally by soil organisms (plant roots, earthworms, etc.) and soil physical and chemical processes (wetting and drying, etc.). Networks of these larger pores can rapidly conduct water from the surface to the tile drains in soils with textures that would otherwise drain very slowly (silt and clay soils). Tillage of the soil, particularly plowing, disrupts macropore continuity and their function as conduits for air and water movement to depth in the profile. The combination of pervasive macropore networks and tile drainage in fields has been observed, in Ontario, to aid water in getting to streams rapidly after precipitation events, augmenting the runoff portion of stream discharge. This water movement pathway can also potentially make significant microbial loading contributions to streams from manured fields in the study area. Injection of liquid manure directly into minimally tilled soils with established macropores and tile drain systems is of particular concern. Further research is required to evaluate the significance of this situation in terms of microbial water quality impairments in the study area.

Locally studies were conducted in the early 1990’s by the ABCA to examine movement of bacteria within tile drains after various modes of manure application and to examine operational factors affecting the movement of bacteria (Foran 1992; Foran and Taylor 1993).

Farmers are urged to inspect tile drain water quality prior to and during manure applications to their fields. Should manure be observed in the tile water, the drain outlets are to be blocked and clean-up procedures are to be carried out. No data are available regarding manure in tile water occurrences observed by farmers or the effectiveness of their corrective actions to deal with problems that may have occurred within the study area.

6.5 Other Agricultural Sources of Fecal Bacteria in the Study Area

Bacteria can enter the environment from a number of “non-manure”, agricultural sources in the study area. These sources include milking centre wastes and washwaters from dairy farms and dead animals (deadstock) from all livestock operations.

6.5.1 Milking Centre Wastes and Washwaters

Dairy farms in Huron County are important milk producers in Ontario. Milking centre wastes and washwaters, if not managed properly, are potential sources of bacterial loading to land and watercourses on these farms. Recommended practices for these liquid wastes would have all first time rinse waters collected and fed to animals rather than discharged. Alternatively, these rinse waters could be stored for field application under appropriate conditions. In addition, less than 1-gallon (4.5L) of milk per day should ever enter the milking centre washwater. Manure, excess feed and other solids should be removed from the parlour or milkhous floor prior to wash-down. Practices other than these, elevate the microbial load in milking centre washwaters.

Historical practices for milking centre washwater management included direct discharge into a channel/trench outside the milkhous or to pipes connected to field tile drains. Practices have evolved to better manage these wastes. Sediment tanks with upwards of 8 days of storage capacity are recommended. Treatment trenches with upwards of 5 feet of length for every 1.5 gallons (6.75L) of washwater produced daily (minimum length of 320 feet) are now constructed on dairy farms. Trenches considered to be “best” in terms of their design have greater than 6 feet (2m) of soil depth beneath their bottoms over bedrock or to the water table. These trenches would also be greater than 500 feet away from the nearest surface water source.

Milking centre wastes and washwaters remain potential microbial loading sources for local streams if appropriate management practices are not followed to mitigate their impacts. No data are available for the study area with which to adequately assess this potential microbial source.

6.5.2 Deadstock

All livestock operations face mortalities as part of their normal farm practices. These animal carcasses are a public health concern and can impact the environment (e.g. be microbial sources) so farmers must plan appropriately for their disposal. The number of animals that die annually before being shipped to markets from farms in the study area is unknown.

Deadstock management in Ontario has been accomplished by removal to off-site facilities (rendering plants, landfills, etc.), by burial on the farm or by on-site composting. Recent beef industry events, the “BSE crisis” in particular, have substantially changed the dead animal removal opportunities for Ontario farmers. No data are available for the on-farm composting and the deadstock burial management options presently used by study area farmers.

The on-farm deadstock management options for study area farmers are not currently monitored for their respective microbial load impacts to the environment. Farmers are encouraged to follow appropriate animal burial and composting practices that aim to minimize environmental impacts. Improper practices could result in animal carcasses yielding microbial loads to streams or aquifers and then to the Lake Huron shoreline in the study area. There is potential for this source to contribute to the *E. coli* (and other microbial) load to surface waters but no studies have been conducted to assess this pathway.

Deadstock composting can be accomplished with a number of techniques on Ontario farms. They range from “in-vessel” confined systems to field “windrow” methods. Common to all composting systems is the use of decomposer and detritivorous organisms that are managed under specific temperature and moisture regimes to reduce the organic waste (carcasses in this case) to beneficial humic soil amendments. Temperatures maintained at and above 55°C during this process are required to kill off the pathogens in the compost. Unfortunately, no data are available for deadstock composting in the study area.

6.6 Implications for the Transport of Bacteria from Farms to Lake Huron Beaches

This section of the report has discussed Huron County farms and farming practices in the context of potential sources contributing loads of microbial fecal pollutants to surface waters. Past studies on the shores of Lake Huron (see 2.2 and 2.3) provide evidence that agricultural activities have in the past contributed to the fecal pollution of surface waters of Huron County, and provide a basis to predict that agricultural activities likely continue to contribute to the wide-spread fecal pollution of tributary waters observed today. However, the contribution of agricultural sources to the composite load of fecal pollutants delivered to the shores of Lake Huron has never been defensibly established nor has the significance of the tributary loads of fecal pollutants (irrespective of source) on water quality at recreational beaches been clearly identified.

Over the past several decades, several studies have concluded that agricultural sources contribute to the bacterial loads measured for the beach waters in the Great Lakes area. The August 2004 report from the Natural Resources Defense Council (NRDC) entitled “Testing the Waters 2004: A Guide to Water Quality at Vacation Beaches” suggests that uncontrolled runoff waters from farms “may contain high concentrations of pathogenic animal waste” that can foul beaches (Dorfman 2004). This report included a “Focus on the Great Lakes” section with results of beach water monitoring from 2002 and 2003. It was quite vague in its accounts of actual agricultural sources of beach water quality impairments in the Great Lakes basin. Watershed and beach water studies in the Lake Huron basin – both within and out of the study area – have made much more specific statements about agricultural sources and potential contributions to impairments.

Agriculture has been included as a potential source of beach water quality impairments in most reports on this topic that cover the study area. A recent report by the Lake Huron Centre for Coastal Conservation (see 2.10) (Lake Huron Centre for Coastal Conservation 2004) summarizes the findings of beach and nearshore environment-related studies on water quality in the southeastern Lake Huron area since the 1980’s. It includes comments about potential agricultural sources that span the last several decades from the “Clean Up Rural Beaches (CURB)” reports of the mid-1980’s to the most recent series of Environment Canada reports by Blackie and Tuininga (2003a,b,c) on chronic and acute water quality impairments from livestock and manure mismanagement. This Lake Huron Centre for Coastal Conservation report, as with many of the others that it reviews, correctly identifies Huron County shoreline tributary rivers

and creeks (Maitland, Pine, etc.) as draining lands with among the highest concentrations of livestock in Canada according to census data from Statistics Canada (2001). However, there are major limitations on the degree to which information based on land-use statistics, non-specific monitoring data and results of small-scale mechanistic studies can be combined to make inferences on the realized agricultural impacts on beach water quality in the study area.

Blackie and Tuininga (2003a,b,c), in their suite of recent Environment Canada reports, provided a very useful characterization of some of the water quality impairment threats from agricultural activities in Ontario. These threats are aligned with the federal Fisheries Act issues (described previously) that involve habitat destruction and/or deleterious material entry into streams. These authors have described “chronic” threats as those that produce persistent or regular discharges to watercourses. These threats included runoff from manure storages, exercise areas, barnyards or feed storage, unrestricted livestock access to watercourses and dairy farm milking centre wastes and washwater discharges. Collectively and individually, they have cumulative microbial loading potential for these watercourses over time. The individual factors associated with each of these “chronic” threats were described in some detail previously in this report. Blackie and Tuininga (2003a) also devised an “acute” category of threats for those that are much less regular or frequent or even rare events but which can have significant water quality impacts. These events would include spills of manure, other land-applied materials, silage, milk, etc. One of the most important microbial loading aspects of such “acute” impacts is that these “raw sources” (manure, milk, etc.) have “short-circuit” or “bypass” travel opportunities, by virtue of the spill event itself, to enter watercourses rapidly.

Watersheds within and near the study area (Maitland, East Huron and Bayfield) were among those in the province with the highest number of reported manure spills between 1998 and 2001 (Blackie and Tuininga 2003a). Periodic delivery of bacterial loads from on-farm spill events to watercourses and then to the shores of Lake Huron may re-inoculate the nearshore water column and sediments with a new *E. coli* population. However, the extent to which any of these spills may have impaired Lake Huron beach water quality in the study area has not been established.

The Blackie and Tuininga (2003a) report noted that the “chronic” problem of direct livestock access to watercourses occurs along most of the major tributaries along the southeastern Lake Huron shores. As with many related issues it is difficult to quantify the magnitude of the

problem or make conclusive linkages to beach water quality. Consequently, the evaluation of the significance of the problem is a matter of debate which is destined to be inconclusive.

The magnitude of these contributions to the microbial fecal pollution delivered to the shore of Lake Huron will not be static in space and time. It is important to recognize the complexity, uncertainty and variability associated with the potential differential microbial load contributions from agricultural activities through the seasonal cycles on the land areas that are drained by these tributaries. There is a need for a better understanding of how the various microbial sources contribute to the load delivered to the lake and how these contributions impact upon the water quality of Lake Huron beaches. Significant effort should also be expended to further our understanding of the agricultural management practices that are associated with the generation of the key sources of loading to the environment and how they can be enhanced to mitigate the problems that are identified.

7.0 The Potential Impact of Septic Systems as a Source of Bacteria and Fecal Pollution to the Beach Shoreline of SE Lake Huron

Septic systems are widely used for domestic waste water treatment in the predominately rural areas of the lands draining to SE Lake Huron. Several previous studies, notably the CURB Plan reports for the MVCA and ABCA (see 2.3) have documented the distribution and features of private septic systems over the areas draining the shoreline of Huron County and have raised concern that septic system failures, or the operation of poorly designed or poorly maintained septic systems may be contributing to the fecal pollution of the tributaries and shoreline waters of Lake Huron. The reports also note the potential of many septic systems located along the shoreline to impact directly on water quality at the shoreline of Lake Huron. The discussion which follows focuses on the function and potential environmental impacts of septic systems as they occur on the shores of a large lake, such as Lake Huron. Much of the information is applicable to the numerous septic systems which occur away from the lake and potentially impact upon water quality of tributaries to the lake.

7.1 Septic Systems

Septic systems are the most common means of disposal of individual domestic waste water in rural areas. A typical septic system servicing a single-family dwelling accepts all liquid wastes from the household, including human, kitchen, laundry and general cleaning waste water. The waste water is composed primarily of water, but also includes organic matter, ammonia, phosphorous, non-organic compounds, and microorganisms. The average domestic loading of waste water to a septic system is approximately 160 litres per person per day (Wilhelm et al. 1994).

The basic design, construction and operation of septic systems have not changed significantly in over 100 years (Cotteral and Norris 1969; Wilhelm et al. 1994). Septic systems are composed of three parts: (1) the holding tank or septic tank, (2) the tile-drain field, and (3) the permeable sand and gravel drainage bed. The holding tank, the first stage into which the domestic waste water flows, acts as both a settling chamber to remove solids from the waste water and a digester to treat and thereby reduce the organic matter in the solids and waste water. The tile-drain field consists of a network of horizontally spaced and perforated pipes, or tile drains. Waste water from the holding tank is released from the septic system via these drains which are designed to introduce the waste water over a large area of the uppermost portion of the drainage beds by gravity flow. The sand and gravel bed below the tile bed is placed in the vadose zone well above the water table. This drainage bed should be highly permeable to facilitate both the leaching of the waste water and the evapotranspiration of water out of the drainage bed. It is also designed to facilitate the treatment of the waste water before it reaches the water table.

There are distinct regions or geochemical zones associated with septic systems, and these are related primarily to their reducing and oxidizing (redox) environments in which the waste water is treated (Wilhelm et al. 1994; Robertson and Cherry 1995; Ptacek 1998). The four main regions or redox zones are: (1) the holding tank - anaerobic, (2) the biological zone of microbes (biomat or biofilm) immediately below the tile drains - anaerobic, (3) the sand and gravel bed above the water table - aerobic, and (4) the saturated zone below the water table - anaerobic. Due to a lack of oxygen in the holding tank, the waste water undergoes anaerobic reactions in which microbes use organic carbon, organic nitrogen, H^+ , SO_4^{2-} and produce CO_2 , CH_4 , H_2 , H_2S and NH_4^+ . This zone is characterized as being low in dissolved O_2 and high in organic matter. The second geochemical zone consists of a 2-5 cm thick biological mat or biofilm surrounding

the tile drains composed of suspended particles, organic matter and microbes that are strained by the small pore spaces within the sand and gravel, as well as other microbes that grow within these pores. Waste water flow through the biomat, undergoes the same anaerobic reactions as in the holding tank. This biologically active biofilm may take years to form. The third geochemical zone is encountered as waste water leaches through the sand and gravel bed in the vadose zone, where it undergoes aerobic reactions due to the high concentration of O_2 infiltrating from the surface. Here microbes use O_2 to oxidize CH_4 , H_2S and NH_4^+ to CO_2 , SO_4^{2-} , NO_3^- and H^+ . NO_3^- is mobile in water and readily moves with infiltrating water towards the water table. However, PO_4^{3-} is rapidly removed from waste water through adsorption to aquifer material or by precipitation as minerals deposit within the vadose zone (Ptacek 1998; Zanini 1988). If the depth of the water table is shallow ($\sim <1$ m) and infiltration through the unsaturated zone is rapid, there may be insufficient time for the oxidation of NH_4^+ and it will enter the groundwater flow regime unmodified. The fourth geochemical region is encountered when the waste water reaches the water table and the contaminants enter the groundwater system. Because of the limited availability of O_2 within the saturated zone, microbes will use NO_3^- and produce N_2 during the oxidation of organic carbon in natural aquifer materials. This will result in anaerobic groundwater where the contaminants are found (Roberston and Cherry 1992; Wilhelm et al 1994; Roberston and Cherry 1995). If the levels of NO_3^- persist and organic carbon is consumed, the groundwater will remain aerobic and NO_3^- will not be affected. Because of the relatively high levels of organic carbon required for the NO_3^- to N_2 conversion and the limited amount of organic carbon typically in beach sands, groundwater contaminant plumes from septic systems typically contain NO_3^- . As in the vadose zone, PO_4^{3-} will be rapidly attenuated onto the aquifer material (Roberston and Cherry 1995; Ptacek 1998; Zanini 1988).

Fecal bacteria are commonly found in septic system waste water, and these bacteria are released to the subsurface as septic system effluent leaches through the tile drains (Magdoff et al. 1974; Matthess and Pekdeger 1981; Duda and Cromartie 1982; Alhajjar et al. 1988; Reneau et al. 1989; Paul et al. 1995; Weiskel et al. 1996; Alfreider et al. 1997; Shadford et al. 1997). The most common group of bacteria in both septic system holding tanks and tile drain effluent include total coliforms, fecal coliforms and fecal streptococci (Table 1). Most studies investigating bacteria and septic systems have focused on total coliforms, fecal coliforms and fecal streptococci. There are only a few studies that provide information about *E. coli* or *E. coli* NAR released from septic systems (Matthess and Pekdeger 1981; Shadford et al. 1997; Arnade 1999). Total coliforms are capable of reproducing in septic system holding tanks, but fecal coliforms are much less likely to reproduce.

Table 23: Range of numbers of bacteria in raw sewage and septic tank effluent.

	raw sewage (counts/100mL)	septic effluent (counts/100mL)	source
total coliform	-	$2.0 \times 10^5 - 1.2 \times 10^7$	1
“	$2.2 \times 10^5 - 1.6 \times 10^7$	$3.7 \times 10^5 - 1.5 \times 10^7$	2
“	?	?	3
“	-	$3.5 \times 10^3 - 6.4 \times 10^8$	4
fecal coliform	-	$3.0 \times 10^4 - 6.5 \times 10^5$	1
“	$1.5 \times 10^3 - 6.6 \times 10^5$	$1.0 \times 10^4 - 2.6 \times 10^6$	2
“	?	?	3
“	-	$2.0 \times 10^3 - 5.5 \times 10^8$	4
“	-	4.2×10^5	5
fecal streptococci	-	$6.0 \times 10^2 - 1.4 \times 10^5$	1
“	$1.5 \times 10^3 - 5.6 \times 10^5$	$2.2 \times 10^3 - 5.1 \times 10^5$	2
“	?	?	3
“	-	$4.5 \times 10^2 - 8.3 \times 10^5$	4
E. coli		$\sim 10^6$	1

1: Magdoff et al. (1974) 2: Viraraghavan (1976) 3: Matthes and Pekdeger (1981) 4: Alhajjar et al. (1988) 5: Reneau et al. (1989)

In a properly functioning septic system, most of the solids and nutrients are removed through the anaerobic reactions within the holding tank and the biofilm and the aerobic reactions within the vadose zone. Any waste water that reaches the water table should theoretically be depleted of organic carbon but should also have high concentrations of phosphorous and NO_3^- . Studies show that total coliforms, fecal coliforms and fecal streptococci are not removed in the septic system, in fact concentrations may actually increase in septic system effluent. However, most of the bacteria that are released through the tile beds are filtered (adsorbed to aquifer soil particles) by the underlying sand bed within a metre of the tile drains.

How well a septic system functions to remove contaminants depends on: (1) soil conditions, (2) depth to the water table, (3) loading conditions, (4) retention time in the holding tank, (5) availability of O_2 , and (6) age of the septic system (Cottrel and Norris 1969; Viraraghavan 1976). Because sludge accumulates in a holding tank of a household septic system at a rate of 40 litres per person per year, the holding tank must be regularly emptied to prevent solids

moving in the tile beds which would block the dispersal of waste water. After several years of operation, the biomat typically becomes sufficiently developed that it causes a significant reduction in the hydraulic conductivity of the biomat region. This prevents the leaching of water through the biomat to the vadose zone, ponding of waste water above the tile bed, and in worse case scenarios, the ponded waste water may rise to the ground surface. Thus, in these systems, it is the hydraulic conductivity of the 2-5 cm thick biomat and not the hydraulic conductivity of the underlying sand and gravel bed that controls the leaching of the waste water. The sand and gravel bed requires both a sufficiently large thickness (>1 m) between the tile bed and the water table to allow for the oxidation of the waste water and the filtration of bacteria. Small distances between the ground surface and the tile bed allows for the movement of O_2 from the atmosphere to the waste water. The sand bed must have a sufficiently high hydraulic conductivity to allow for the movement of waste water from the tile bed, but not too high such that it would enable the passage of bacteria to the water table. Studies have shown that it is possible for complete oxidation of NH_4^+ to occur within 1 m of the tile drains and with a few hours of exposure to O_2 . But in typical silica-based beach sands the limited amount of carbonate material in these sands limits the buffering capacity for H^+ produced during the oxidation of NH_4^+ , and hence the pH of waste waters in the vadose zone may fall to as low as 5.5, inhibiting most microbial activities.

There are several factors that control the life-span of a septic system, but the most important is the infiltration capacity of the sand and gravel bed (Cotteral and Norris 1969; Viraraghavan 1976). A reduction in the infiltration capacity of the sand and gravel bed will reduce the leaching of waste water from the tile bed, and hence cause the septic system to fail. This can result from: (1) development of a thick biomat, (2), compaction of the sand and gravel bed, (3) invasion of the tile drains by roots and (4) reduction of pore space by solids. Mitigation strategies for these problems include regular pumping of the solids from the septic system holding tank (every 3-5 years) to reduce the discharge of solids through the tile drains and discontinuous use of the septic system minimizing the anaerobic conditions that favour the development of a thick biomat. This increases the life expectancy of a septic system. However, the life of the septic system can be lengthened considerably if the sand and gravel bed is subjected to extended and continuous dry periods (a few months). This introduces oxygen hence stopping the development of the biomat. A typical life expectancy for a septic system that services a typical year-round family residence is about 12 years.

7.2 Persistence and Transport of Bacteria in Sand and Gravel

It is well known that pathogenic bacteria and other microorganisms in groundwater, can and have caused numerous cases of illness from drinking contaminated groundwater (MacIer and Merkle 2000). However, much of our knowledge about the persistence and transport of bacteria in subsurface water has come from research investigating the use of microbes to remediate groundwater sites contaminated by organic chemicals, either by stimulating the growth of natural bacteria or introducing bio-engineered bacteria, (Schäfer et al. 1998; Murphey and Ginn 2000; Mailloux et al. 2003). In addition, although there have been many studies, over the past several decades, on bacteria in septic systems, only during the past few years have studies begun to characterize the transport and survivability of bacteria as they move from the septic system and through the subsurface waters. These predominantly laboratory (column) studies on bacterial transport through porous media are difficult to extrapolate to field conditions (Kinoshita et al. 1993; Bolster et al. 2000; Mailloux et al. 2003). Nevertheless, all of these studies, including laboratory experiments, have provided a basic understanding of the types of bacteria present at septic systems, how far they can travel through the subsurface, and what affects their migration. The transport of bacteria within the saturated groundwater flow regime are affected by numerous factors, including: (1) sediment size (silt vs. sand) and interstitial pore size, (2) preferential pathways and aquifer heterogeneities, (3) bacterial adsorption efficiencies, (4) clay content and organic carbon content, (5) surface coating mineralogy of sand grains (e.g., Fe), (6) groundwater velocities, (7) groundwater temperatures, (8) the presence of natural bacteria in the subsurface, and (9) the redox environment.

Bacteria can only travel through aquifer material if the pore space between soil particles is sufficiently large to permit their movement. Bacteria, with a diameter of 0.2 - 5 μm , will not travel through the pore space within fine sand, silt or clay, but can pass medium to coarse sand and gravel. Thus, bacteria can travel through the sand and gravel deposits that comprise the coastal beaches of the Great Lakes (Boehm et al. 2004), furthermore, bacterial transport is substantially increased if transported with groundwater flow through preferential pathways such as fractures in rocks or clay.

Bacteria have a strong tendency to adsorb to the soil particles, rather than remain in suspension in the groundwater. Studies consistently show that the density of bacteria adsorbed to soil is 100 to 1000 times greater than bacterial densities in adjacent groundwater (Kolbel-Boelke et al. 1988; Alfreider et al. 1997; Hazen et al. 1991). This ratio of bacteria attached to sediment to

bacteria in groundwater is essentially the same ratio as reported for sediment to lake-water or river water (Duda and Cromartie 1982; Thorn and Ventallo 1988; Doyle et al. 1992). The attachment process is a very rapid one and is selective. Bacteria have a preference for clay and organic carbon particles compared to sand grains typically composed of quartz, or feldspar. As such, sediment composition can increase bacterial populations (Desmarais et al. 2002), for example, a 1% to 2% clay and/or organic carbon content can dramatically increase the occurrence of adsorbed bacteria. Furthermore, bacteria are more likely to adsorb to the iron coatings of sand grains due to the residual charge, rather than the actual sand grains.

Studies indicate that bacteria tend to attach and detach from the aquifer material (Harvey and George 1987; Kinoshita et al. 1993). While the attachment process can be very rapid, the detachment process tends to be very slow (Magdoff et al. 1974; Matthess et al. 1988; Kinoshita et al. 1993). Because of these different rates of movement, the attachment process always dominates over the detachment process and most bacteria are found bound to the aquifer material. Other studies indicate that there are actually two populations of the same bacteria within the subsurface; a “sticky” population that rapidly attaches to and remains on, the aquifer material and a non-sticky population that has a reduced adsorption tendency (Bolster et al. 2000). The sticky population dominates the sediment grains close to the source and do not travel great distances while bacterial populations further away from the source tend to be dominated by the non-sticky microorganisms.

On average, groundwater velocity will not significantly affect the transport of bacteria, because most will be sediment bound. Groundwater velocities will, however, influence the free-floating bacteria and the process of bacterial desorption. As a result, distances traveled by bacteria, especially the sticky bacteria, via groundwater flow through sand are typically small, in the order of metres to a few 10's of metres (Matthess et al. 1988; Shadford et al. 1997; Bolster et al. 2000). But it has been reported that the non-sticky bacteria can travel 10's to 100's of meters (Duda and Cromartie 1982; Bolster et al. 2000). Because of the predisposition of bacteria to adsorb onto the aquifer material, bacteria travel rates and distances, relative to other septic system contaminants such as nitrate, phosphorous and chloride, are as much as 500 times less (Matthess and Pekdeger 1981).

Water table fluctuations also affect bacterial populations. Bacteria accumulate and grow at the water table (air-water interface) and in saturated pores just above the water table (Schäfer et al. 1998). Although high levels of *E.coli* are often measured in sand at the water table beneath

beaches and at wells, *E.coli* does not appear to survive below the water table (Desmarais et al 2003; Whitman and Nevers 2003). Thus, as the water table beneath a beach fluctuates due to infiltration and changes in lake levels, the bacterial populations will also change accordingly.

Temperature also has a significant impact on the survivability (mortality) of bacteria in groundwater flow regimes (Matthess and Pekdeger 1981). Studies show that as temperature increases from 4°C to 35°C, bacteria mortality and densities in groundwater decreases exponentially (Howell et al. 1996). Typically deep groundwater temperatures in watersheds adjacent to Lake Huron are about 10°C to 12°C. The temperature of shallow groundwater (i.e., within a few metres of the ground surface) below beaches is quite variable throughout the year due to the impact of air temperatures and the temperature of the infiltrating water. However, it can be expected that the temperature of groundwater near the water table beneath beaches will be warmer and cooler than that of the adjacent lake in the winter and summer, respectively.

Most studies indicate that bacteria from septic systems reach the water table below septic systems, yet, some studies report that no bacteria occur in the groundwater below septic systems (Magdoff et al. 1974; Alhajjar 1988). This discrepancy between studies is likely due to the presence of a vadose zone between the septic tile drains and the water table. Although bacteria can leach from the tile drains, they can only leach to the water table in the presence of leaching waste-water; bacteria will not migrate through unsaturated soil. In addition to all the factors affecting the transport of bacteria within the saturated zone, transport in the vadose zone is also controlled by the moisture content (saturation) of the vadose zone and the thickness of the vadose zone between the tile drains and the water table. Studies have reported that only 1 m distance between the tile beds and the water table is sufficient to attenuate bacteria and prevent bacterial leaching into the groundwater regime.

Thus, the transport of bacteria from septic systems occurs and must be considered as transport within two distinct zones:

- (1) transport through the vadose zone from the tile drain to the water table, and
- (2) transport from the water through the saturated groundwater flow regime.

The two main factors controlling whether or not bacteria will reach the water table are the soil moisture content with downward movement of the soil water, and the thickness of the vadose zone. The main factor controlling whether or not bacteria that reach the water table will actually migrate to the lake is the distance between the lake and the septic system.

7.3 Groundwater Flow at Beaches

Beaches composed of sand or sand and gravel, generally have very consistent hydrogeological properties throughout the Great Lakes, due to the nature of their deposition and source materials (Crowe and Shikaze 2004). Sand and gravel deposits have a high hydraulic conductivity of about 10^{-1} to 10^{-2} cm/s and a high porosity of 0.35 to 0.45 (Crowe et al. 2004). Within the vadose zone, sands and gravels have a very low residual moisture content of ~3% and the uppermost 8 to 12 cm at the ground surface is typically dry (moisture content = 0).

The direction of groundwater flow and water table fluctuations beneath beaches are very similar throughout the Great Lakes. The groundwater flow regime and water table fluctuations beneath beaches are affected by three main factors: (1) infiltration at the beach and in the adjacent area, (2) drainage to the lake, and (3) fluctuations in lake levels. Infiltration of precipitation at the beach and the area adjacent to the beach maintains a water table that is higher in elevation than the adjacent lake. Hence, with the exception of storm events as described below, the direction of groundwater flow through the beach sands is always towards the lake, and infiltration of precipitation is the primary source of groundwater beneath a beach. In large bodies of water (such as Lake Huron), groundwater discharge tends to be focused within a few metres of the shoreline and the discharge rate decreases exponentially with distance offshore (McBride and Pfannkuch 1975, Pfannkuch and Winter 1984, Cherkauer and Zager 1989). Discharge rates along the shore are quite variable, from 0 to 8 L/min (Pfannkuch and Winter 1984; Boehm et al. 2004)

The high hydraulic conductivity and porosity of beach sand and gravel enable rapid infiltration; the downward movement of infiltration through up to 2 m of sand to the water table can occur within 2-3 days (Mills 2004). Infiltration can rapidly raise the water table, with the magnitude of the rise being a function of the intensity of the storm (e.g. more precipitation = greater rise,) and the depth of the water table (e.g. shallow water table = greater rise) (Crowe and Shikaze 2004). After an infiltration event, removal of entrapped air and re-adjustment of hydraulic heads cause rapid short-term declines in the elevation of the water table. The water table can also decline due to drainage towards the lake and evapotranspiration. Evapotranspiration will only influence infiltration and the water table in the vegetated areas adjacent to the beach, but not on the beach itself where there is little to no vegetation. The water table will respond rapidly to other

external stressors and sources of water, such as infiltration from septic systems, infiltration from water applied to lawns, lake-level fluctuations, and changes in vegetation.

The greatest impact, and hence the greatest seasonal rise of the water table (0.5 – 1.0 m), occurs during the spring due to the infiltration of the snow melt and spring rains (Doss 1993; Crowe et al. 2004). Infiltration of precipitation during the summer and fall tend to cause short term rises of the water table by a few centimeters to a few 10's of centimeters. On average, however, the water table declines throughout the year due to drainage towards the adjacent lake.

Although lake water does not flow into the beach sands, the lake does exert considerable influence on the groundwater flow regime beneath the beach through fluctuations in lake levels. Lake levels in the Great Lakes are affected by short term (seiches, storm-driven waves), seasonal (low levels during the fall and winter, high levels during the spring and summer), and long-term (continental climatic patterns, droughts) factors.

Storm driven waves do not change the overall lake levels at a beach by more than a few 10's of centimeters, however, because individual waves can be much higher, and depending on the width and the slope of a beach, the run-up of waves can be quite large (>10 m). The run-up of waves creates a transient land-water interface that moves across the beach at a rate equal to the time required for wave run-up (Baird et al. 1998; Horn 2002). The run-up of waves during a storm will cause the infiltration of lake water into the beach sands and some backwash of groundwater flowing towards the shore. These two factors can raise the water table below the beach, but the rise of the water table due to the infiltration of lake water during wave run-up occurs only below the swash zone. In fact, infiltration can raise the water table to ground surface resulting in the development of seepage faces (Horn 2002). Away from the swash zone, the rise of the water table is primarily due to the backwash of groundwater flowing towards the shore and due to the transmission of pressure forces. These can raise the water table several centimeters up to a distance of 10's of meters from the edge of a beach. Any lake water that does infiltrate into the beach will remain there for a few hours to days before the groundwater flow regime re-adjusts to the unstressed lake levels.

Lake levels of the Great Lakes also fluctuate seasonally, with the highest levels during the spring and early summer, and the lowest levels during the winter. Lake levels are highest during the spring and early summer due to runoff of snowmelt and spring rains. Lake levels are

lowest during the winter due to increased evaporation from the lake surface and decreased runoff from tributaries. Seasonal fluctuations of the surface elevation of Lake Huron are approximately 0.3 m. These seasonal fluctuations occur at the same time as long-term fluctuations caused by years of lower than or higher than average precipitation and temperature. During the past 100 years, the surface of Lake Huron has fluctuated by about 2 m, with the lowest level occurring during 1964 (175.58 m amsl) and the highest occurring during 1986 (177.50 m amsl).

There are two main impacts of seasonal or long-term higher lake levels on groundwater-lake interactions. First, as the surface of the lake rises or falls due to seasonal lake-level changes, the position of the groundwater-lake interface (shoreline) moves perpendicular to the shoreline. Because groundwater discharge is focused near the shoreline, the location of this discharge zone will shift with the shoreline interface during the year (Crowe and Shikaze 2004). Second, the elevation of the water table in the near shore environment will rise and fall seasonally as the elevation of the water level in the lake surface rises and falls. The rise of the water table during the spring is not caused by water from the lake moving into the adjacent aquifer, but is caused by the sudden and large amount of infiltration of snow-melt and precipitation. The higher elevation of the lake surface produces a higher elevation of the point of groundwater discharge. As the water levels in the adjacent lake fall during the fall and winter, the draining groundwater discharges at a lower point in response to a lower water table.

7.4 Implication for Septic-System-Derived Bacteria to Migrate to the Shoreline at Lake Huron

There have been no studies that have tracked the migration of bacteria from septic systems adjacent to the beaches of Lake Huron to the shoreline to 1) determine how far bacteria can migrate from these septic systems, or 2) determine if bacteria from these septic systems are reaching the water. Thus very little is actually known about bacterial movement from septic systems to lake water (Lake Huron Centre for Coastal Conservation 2004). Based on the current state of knowledge about the fate and migration of septic system derived bacteria in the subsurface, our knowledge of groundwater flow beneath beaches, and our cursory knowledge of the area, we can make some inferences as to the potential for bacteria to travel to the shoreline.

The hydrogeological properties of the sands and gravels that comprise the beaches and dunes along the shores of Lake Huron are very conducive to the leaching of contaminants from the surface to the water table. These are in turn transported with groundwater through the saturated groundwater flow regime. Overall, net annual groundwater flow within beach environments is towards the shoreline.

Many studies have documented the transport of septic system derived contaminants, including bacteria, discharging into adjacent creeks, river, lakes, wetlands, drains and marine waters (Duda and Cromartie 1982; Chen 1988; LaPoint 1990; Robertson and Cherry 1992; Paul et al. 1995; Weiskel et al. 1996; Paul et al. 1997; Ptacek 1998; Paul et al. 2000). It is likely that some contaminants (chloride, nutrients) from septic systems located along the shore of the Lake Huron beaches are discharging into the lake. It is also possible that fecal bacteria in septic system effluent are reaching the shoreline of these beaches given that the pore size in the beach sands and gravels is sufficiently large to allow for bacterial movement. There are also local conditions along the beaches of Lake Huron which increase the probability of bacteria reaching the shoreline. However, past research also indicates that there are many factors that can reduce and perhaps eliminate the possibility that bacteria from septic systems will ever reach the shoreline.

Before bacteria from septic systems can migrate towards the shoreline, the bacteria must be able to migrate downward through the vadose zone to the water table. The hydrogeological properties and the very low organic carbon and clay content of the sands in which the septic systems along the shores of Lake Huron are typically constructed, are very conducive to the leaching of bacterial contaminants from the surface to the water table. While the water table will likely be 2-3 m below most tile drains along the beaches of Lake Huron, small volumes of water released into the tile drains will move rapidly downward and possibly introduce bacteria to the water table.

Once in the saturated zone, bacteria will migrate in the direction of groundwater flow towards the shoreline. If bacteria are able to migrate to the shoreline, bacterial discharge will be concentrated within a few metres of the shoreline, corresponding to the greatest flux of groundwater. However, given that the residences, and associated septic systems, along the beaches are generally over 40 m from the shoreline, it is unlikely that, under the current natural conditions, the dominant sticky bacteria will migrate these long distances. On the other hand, the free floating or non-sticky bacteria, if present, could travel that far.

There are several factors along the beaches of Lake Huron that could influence bacterial migration to the shoreline, and in most cases enhance the probability that septic-system derived bacteria are present along the shoreline.

1. Although the distance from the tile drains to the shoreline is too large for direct migration of bacteria to the shoreline, there is a much shorter migration pathway to the beaches via the creeks and drains that run past residences and their septic systems and discharge at the shoreline. Any septic system within 15 - 20 m of a creek or drain will probably discharge bacteria into these creeks or drains.

Long-term lake level fluctuations move the shoreline perpendicular to the beach, thus reducing both the travel distance between the shoreline and the septic system, and moving the zone of groundwater (and contaminant) discharge across the beach.

Currently the level of Lake Huron is quite low, but it will likely rise in the near future, reducing the travel distance for bacteria between septic systems and the shoreline.

Conversely, the current low lake levels and receded shorelines, create a wide beach, and the residual areas of groundwater discharge which may retain bacterial populations could be exposed during shoreline erosion.

2. The depth to the water table under natural conditions is probably 2-3 m below the tile drains. But the loading of water from the septic system will cause a small mounding of the water table and hence slightly reduce this depth to the water table. In the past, most of the residences along the Lake Huron beaches were seasonal cottages that contributed low volumes of water to the subsurface flow (occupied for 2-4 months each summer, only one bathroom, etc.). However, in recent years many of the residences have been converted into year-round homes that contribute considerably more water to the subsurface flow (occupied 12 months each year, 2-3 bathrooms with bath tubs, dish washers, etc.). In addition, many of these residences have replaced the natural sand dunes and vegetation with lawns, especially over the septic systems, and their watering also contributes additional loading to the water table. As a result the water table beneath these residences is higher, and thus the thickness of the vadose zone between the tile bed and the water table is less. Also, the additional loading has created a groundwater mound below the tile bed, and hence increases groundwater velocities towards the lake.
3. It is well documented that higher levels of fecal indicator bacteria are measured along the shoreline during periods of high winds and waves. Studies have indicated that the bacterial populations that live in the sand along the shore are introduced into the lake

water during these high wind and wave events due to erosion of the sand along the shoreline. These shoreline events also raise the water table adjacent to the shoreline, causing an infiltration into the location of the bacteria (bacteria inhabit the groundwater-air interface). Raising the water table immediately adjacent to the shoreline during a storm, could raise bacteria to the ground surface where they could be exposed to erosion and introduced into the lake water.

4. Most of the shoreline along Lake Huron supports a low density of residences (e.g., 1 row of residences parallel to the lake) and therefore a low septic system density of about 1 septic system every 20 - 40 m of shoreline. If septic system effluent is reaching the shoreline from these residences, the length of the actual shoreline impacted by contaminated groundwater discharge would be quite narrow (10-15 m) and loading rates would be relatively small. However, there are several locations along the shoreline that have a high density of residences (multiple rows of residences and small communities) that are not serviced by municipal waste-water treatment facilities, and hence septic systems are in high density in these areas. The cumulative impact from a large area of septic systems would result in a substantially higher loading rate and wider lengths of impacted shoreline.
5. The groundwater environment along the shoreline provides a stable environment for bacteria because it is rich in fine particles for attachment, rich in nutrients, isolated from predators, and has a fairly stable thermal environment. In fact studies undertaken within a few metres of the shoreline have shown that the concentration of bacteria, including *E. coli*, both adsorbed onto the sand beneath the beach and in the groundwater below are much higher than in the adjacent lake water. Other studies have shown that after the bacterial population within the subsurface zone is disturbed, they return to pre-disturbed levels within two weeks. Although it is not clear if these bacteria are being transported to the shore with septic system effluent, it is clear that the shallow (i.e., at the water table) groundwater environment is a potentially important source for bacteria in the nearshore water.

8.0 Shoreline Storm Water Drainage as a Source of Fecal Pollution to the Shoreline of Lake Huron

8.1 Stormwater Runoff in an Urbanized Area

Water collected from surface runoff in urbanized areas is classified as urban runoff. Most urban runoff occurs as a result of storm events and snow melt. Rain or snowmelt washes over impermeable surfaces (e.g. roads, driveways, parking lots) and picks up surface contaminants. It then makes its way through constructed storm drains which typically discharge untreated runoff directly to surface waters, unlike sewer drains which channel their flows to waste water treatment plants before discharging their effluent.

It is likely that urban runoff picks up fecal material from urban animals such as dogs, cats and birds; in fact, urban runoff is known to have appreciable levels of fecal bacteria including *E. coli*. Since storm drains respond directly to rainfall, there is little delay in the outcome of the event and the flushing effect means that much of the bacterial loading occurs shortly after the start of any event. Storm water runoff is a potential source of fecal pollution where storm water drains discharge on or near beach areas.

Studies of urban storm water have shown that stormwater can contribute to the bacterial loading of waterbodies that receive stormwater discharge (e.g. Marsalek et al. 1996; Maunder et al. 1995; Reeves et al. 2004). A recent study from California compared a variety of urban land-use features (i.e. residential areas, parks) in terms of their bacterial loadings to storm water (Reeves et al. 2004). Storm water from residential areas was found to have comparatively higher levels of fecal indicator bacteria.

The town of Goderich and the community of Bayfield have storm sewers which discharge directly to the shoreline of Lake Huron, and in some cases, these drains are in close proximity to recreational beaches (Figure 66). The town of Goderich has multiple discharge points, three of which drain directly into Lake Huron and others which discharge to the Maitland River or Goderich Harbour. Figure 66 illustrates the approximate positions of the storm water discharge points, one of which is directly adjacent to St. Christophers Beach. No information could be

found on a storm sewer in Bayfield which discharges to the shores of Lake Huron near the Bayfield Beaches.

Limited information was found on the microbial quality of storm sewers in Goderich. A recent study of the sewerage system included limited work on *E. coli* levels in storm drains discharging directly to the lake (B.M. Ross and Associates 2004c). The results from a limited number of samples taken in 2003 from storm sewers draining directly into Lake Huron suggested the load was potentially of the same order of magnitude as bypasses or release of treated sewage effluent.



Figure 66: Aerial photograph of Goderich showing the approximate location of storm sewer outlets. The location of storm sewers is based on maps in (B.M. Ross 2004c).

8.2 Shoreline Drainage in Low Density Residential Areas

The numerous shoreline residential and cottage areas and the lands directly behind the shoreline of Lake Huron in Huron County, utilize surface and sub-surface drains to remove surface runoff from the inhabited areas to varying degrees. Visual observations of these shoreline areas in 2003 indicated that apparent surface drains, terminating on, or at the edge of beaches are commonplace (Figure 67). In many cases, flows are artificially routed to the shoreline through tubing running from slopes and cliffs behind the edges of the beach. There appears to be little information on the origin of such flows, the land areas generating the flows, or the volume and quality of the flows. Agricultural lands are frequently adjacent to the strips of residential and cottage-occupied lands along the lakeshore and further complicate the interpretation of the drainage areas generating the water transported in the beach drains. Irrespective of the source and function of such shore drains, the proximity of the drains to the beaches suggests that their periodic flows should be considered as potential sources of microbial input to the beach and possibly the shoreline.



Figure 67. Example of a shoreline drain terminating on a beach.

9.0 Wildlife contributions to fecal pollution along the Lake Huron shoreline

Fecal pollution from wildlife sources is known to contaminate drinking and recreational waters across Canada. In some cases, this fecal pollution has led to outbreaks of waterborne diseases. For example, an outbreak of toxoplasmosis in Victoria in 1995 was thought to be associated with fecal contamination of drinking water from cats like cougars (Bowie et al. 1997). In other cases, fecal pollution from gulls has contributed to the impairment of recreational waters near Quebec City (Levesque et al. 1993). For these reasons there is a need to understand the importance of wildlife species as potential contributors of fecal pollution into Lake Huron beach areas. The following section summarizes information that could be found about wildlife population sizes and trends along shores of Lake Huron. It also summarizes information about the possible significance of fecal pollution and *E. coli* loading from wildlife sources in the area.

9.1 Wildlife: population sizes and trends

Forest fragmentation and development of agricultural lands have had a significant impact on the diversity and size of wildlife populations in southern Ontario. While much of the land in many Lake Huron watersheds has been converted into agricultural use, these areas can still provide habitat for bird and small mammal species. In fact, many efforts to ensure buffer strips along streams can provide habitats for birds and small mammals that may facilitate fecal inputs into streams. In addition, there is the possibility that wildlife such as gulls, Canada geese, and deer may leave droppings on agricultural lands contributing to fecal loads into nearby waters.

There is little information available on the size and trends of wildlife species in the Lake Huron coastline area. This is particularly the case for mammalian wildlife (Malhiot, pers. comm.; Wesloh, pers. comm.). Information from the Ontario Ministry of Natural Resources and Environment Canada's Canadian Wildlife Service is mostly related to hunting data on deer populations and from sporadic surveillance of colonial bird species, respectively. The following sections summarize the available information pertaining to a few wildlife species that could be contributing to fecal pollution loadings in the Lake Huron area.

Deer:

The annual harvest of white tailed deer in Huron County is shown below (Figure 68), courtesy of Mike Malhiot, Ontario Ministry of Natural Resources, Clinton. In 2003, there were about 1800 deer harvested. OMNR modeling of deer populations in Ontario would suggest that hunting probably now harvests about 25% of the deer population (Malhiot pers. comm.). This would suggest there could be somewhere around 7200 white tailed deer in Huron County. The harvest data also suggest an increasing white tailed deer population despite a recent increase in hunters and season length in Huron County.

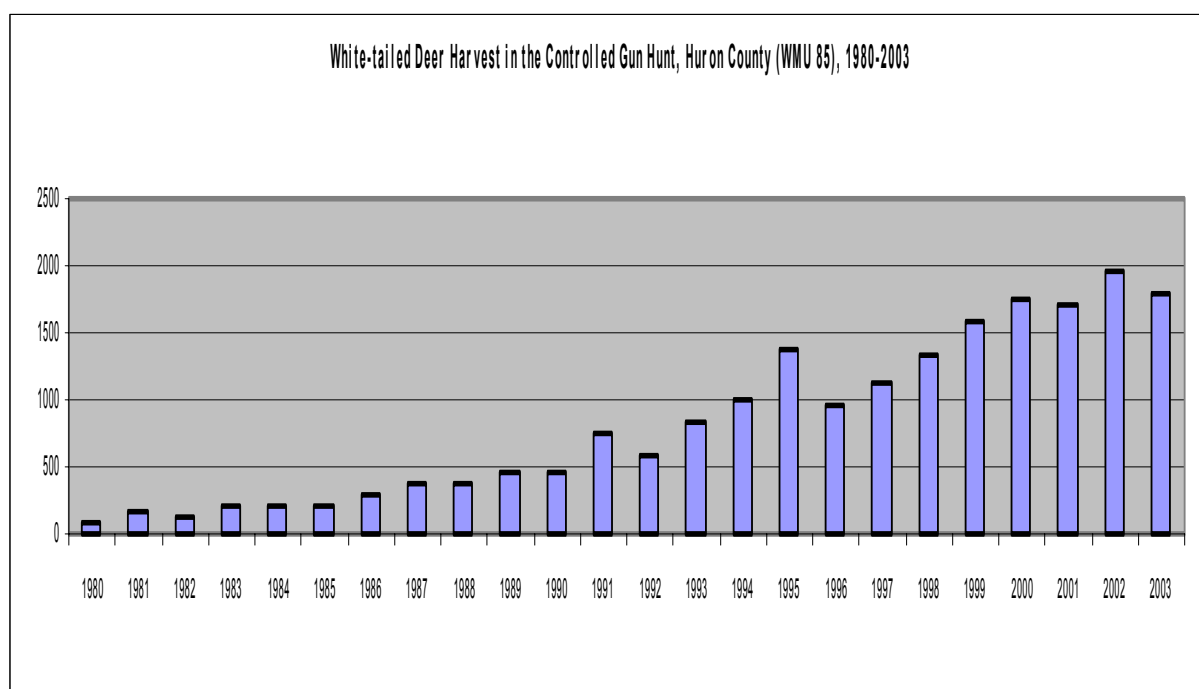


Figure 68. White-tailed deer harvest in Huron County (WMU 85) from 1980-2003

Bird populations:

Prince and Flegel (1995) identified 89 avian species in their review of the breeding avifauna of Lake Huron. They identified several species with high breeding densities in a number of areas along the lake's shoreline including: double-crested cormorant, great blue heron, mallard duck, common and red-breasted mergansers, osprey, killdeer, spotted sandpiper, ring-billed gull, Caspian and common terns, tree swallow, yellow warbler, and red-winged blackbird. Species such as the double-crested cormorant and ring-billed gulls were identified as having significant

population increases in the past decade. In particular, it appears that the ring-billed gulls have experienced the greatest population increase over the longest period of time of any Great Lakes species. The adaptability of this species has enabled it to exploit diverse food sources such as fish (e.g. alewives and smelt), garbage and mice in recently plowed fields.

Certain bird species are more commonly associated with shorelines and may occur frequently in beach areas. The National Water Research Institute (NWRI) studies of beaches in the Hamilton and Toronto areas have found large numbers of birds like gulls and Canada geese on beaches and surrounding grasslands. However, these beaches are also in urbanized areas, and it is uncertain whether such situations are likely to occur along the Lake Huron shoreline. The following bird species are among those more likely to contribute fecal contamination and *E. coli* to beaches along the Lake Huron shoreline.

Canada geese:

As a result of a management program that began in the late 1960s, estimates of the population of Canada geese (*Branta canadensis maxima*) in southern Ontario have grown from about several thousand birds in the late 1960s to more than 350,000 by August, 1998 (Dennis et al. 2001). An analysis of historical breeding pair data and more recent data from 2003 are presented below (Figure 69), courtesy of Jack Hughes, Canadian Wildlife Service, Environment Canada, Ottawa, Ontario. These data confirm the management program population estimates, showing a significant increase in Canada geese numbers in southern Ontario in recent years. However, survey data specific to the Lake Huron shoreline could not be located for Canada geese.

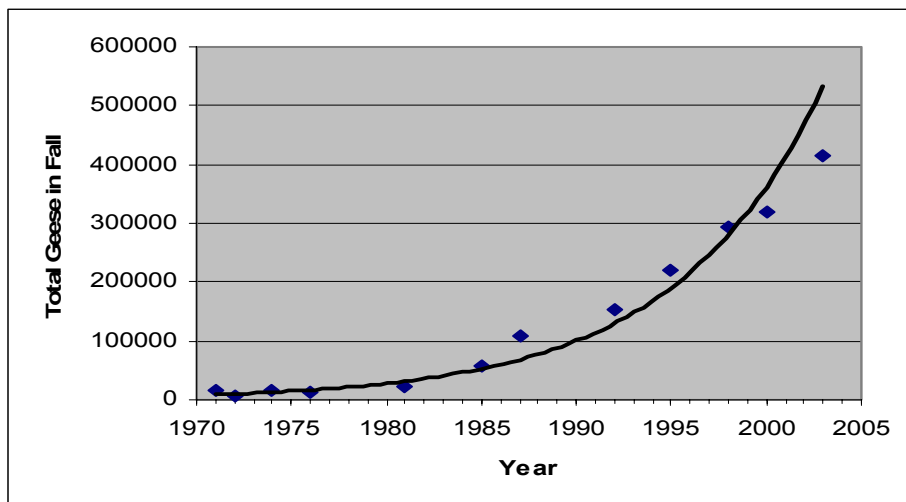


Figure 69. Estimated number of Canada Geese in southern Ontario

Gulls and other colonial waterbirds:

In 1980, the first complete census of breeding areas of selected colonial waterbirds was conducted in the Canadian waters of Lake Huron (Weseloh et al. 1986). At the time of this survey, all bird species were more numerous in Canadian waters of Lake Huron than in U.S. waters. The herring gull was the most widespread species found (33,800 nests on 376 colonies) although ring-billed gulls were the most numerous (138,000 nests on 81 colonies). In comparison to some historical data from the early 1960s, it seemed that some herring gull and ring-billed gull populations had grown about 33% and 350%, respectively. In addition, a six-fold increase in the number of Lake Huron cormorant nests was identified from 1973 to 1980.

While Weseloh et al. (1986) identified colonies of birds along the west shore of the Bruce Peninsula in their 1980 survey, there were no colonies along the shores of the main body of Lake Huron south of Southampton. Major nesting areas around the Lake Huron shores were generally found where there were many islands, which are absent south of Southampton. The closest bird colonies to the beach area of concern along the Lake Huron shoreline were located at Chantry Island (44° 29.6' Lat. and 81° 24.2' Long.) and Douglas Point (44° 19.1' Lat. and 81° 36.4' Long.). Chantry Island (closest to the beach area of concern) had Lake Huron's largest herring gull colony at 3,714 nests, in addition to 2,748 ring-billed gull nests, 97 black-crowned

night heron nests, and 24 great blue heron nests. On the other hand, the Douglas Point colony had 5,811 ring-billed gull nests and 285 herring gull nests.

Chantry Island and Douglas Point colonies were surveyed again in 1989 (Blokpoel and Tessier 1997) at which time Chantry Island had 2971 ring-billed gull nests and 2543 Herring gull nests while Douglas Point had 6,553 ring-billed gull nests and 152 herring gull nests. More recent information on bird colonies at Chantry Island from 1999 and 2000 was provided by Chip Weseloh, Canadian Wildlife Service, Environment Canada, Downsview, Ontario. Chantry Island had 7,440 ring-billed gull and 3,457 herring gull nests in 1999. The numbers of double-crested cormorant nests on Chantry Island has increased an order of magnitude over a dozen years, from 185 in 1988 to 1,429 in 2000. It is evident that there have been substantial increases in ring-billed gulls and cormorants on Chantry Island in recent years. This Island is the closest waterbird colony to the Lake Huron beaches area south of Kincardine, Ontario.

9.2 Wildlife: *E. coli* and fecal pollution

Wildlife species vary in the levels of *E. coli* found in their feces. Seyfried and Harris (1990) compared levels of *E. coli* in feces from humans, wildlife species and domestic animals in the Toronto area. The highest *E. coli* concentrations per gram of feces ($> 10^8$) were found in bird species such as pigeons, ducks, and gulls. High concentrations were also found in raccoons, chickens, dogs, and humans while lower concentrations were found in the feces of muskrats, cats, geese, and horses.

Gould and Fletcher (1978) calculated that the weight of daily fecal droppings from gulls, as a percentage of body weight, was about 10 times greater than that of humans. Based upon a comparison of their data to published data on other animal species, they concluded that the daily fecal coliform load from some gull species could exceed that from humans. While the daily amount of feces produced by gulls may be considerably less than humans or other animal species, its high concentration of fecal coliforms may actually result in a more significant fecal pollution load. In another study, Alderisio and DeLuca (1999) found that while Canada geese had larger fecal deposits than gulls (mean of 8.35 g versus 0.48 g), the gulls had a much higher concentration of fecal coliforms than geese (mean 3.68×10^8 FC per gram of feces versus 1.53×10^4 FC per gram of feces).

Behavioral aspects are also important in considering potential wildlife contributions of *E. coli* into beach environments. Few mammal species are likely to occur directly on beaches except, perhaps, companion animals like dogs. However, birds can be regular visitors of beaches. Species like gulls occur frequently at the waters edge leading to opportunities for direct fecal deposits into nearshore waters and the wet sand. NWRI studies have found significant loadings of fecal material from gulls and Canada geese onto the sand at some Lake Ontario beaches.

Colonial birds like gulls can occur in large concentrations in nesting areas, and they can range many kilometers when foraging for food. For example, gulls can forage as far as 40 kilometers from a colony (Cramp and Simmons 1983). Consequently, the gulls found on Chantry Island can be expected to forage for food in much of the area of concern along the Lake Huron shoreline. They can also venture inland while foraging, perhaps contributing to an additional fecal pollution load onto agricultural lands and into nearby streams.

It is unknown whether Canada geese are as numerous as gulls along the Lake Huron shoreline. However, if Canada geese occur in problematic numbers at some locations, the impacts of their fecal deposits are likely more noticeable at certain times of the year. Local acute nuisance problems can occur when adult Canada geese in Ontario are unable to fly for several weeks in June and July as they moult their flight feathers. Geese start moving into moulting areas around mid May, and by June 25, most geese are flightless (Dennis et al. 2001). These nuisance areas can be common particularly where there are parks with grass areas adjacent to rivers, lakes or ponds.

It is possible for birds such as gulls to contribute substantial *E. coli* loads to beach areas. Gould and Fletcher (1978) studied caged gulls to determine the frequency and characteristics of their fecal droppings. Individual gulls produced between 34 and 62 droppings in 24 hours. The total weight of fecal droppings over this period ranged from 11.2 g to 24.9 g per gull. While this study did not measure *E. coli* specifically, average fecal coliform count (per gram wet weight) ranging from $0.003 - 480.0 \times 10^7$ resulted in daily individual gull fecal coliform loadings from $3 - 50 \times 10^8$ cfu. The fecal droppings that were white with large green or brown (fecal) centers were generally associated with higher fecal bacteria counts than droppings that were mostly white.

Bird fecal impacts on water quality:

A number of studies have demonstrated that birds can be a significant source of fecal contamination leading to the impairment of water quality. Palmer (1983) found bridges crossing the Rideau River in the city of Ottawa, Ontario had between 36 and 155 roosting pigeons. In fact, the six bridges that were studied had a total of between 447 and 644 pigeons on any given survey day. These bridge birds were estimated to have a significant impact on fecal coliform levels downstream of these structures particularly during summer dry weather flows. Individual pigeons were estimated to have daily fecal coliform loads of 0.88 to 1.3×10^{10} organisms, and all the birds were estimated to contribute between 17 and 35% of the total dry weather river loadings. Palmer suggested the need to consider sampling on both upstream and downstream sides of bridges if any bird contamination from bridge sources was likely. It is possible that birds nesting under Lake Huron area bridges, or perhaps occurring in trees overhanging water courses, might be contributing *E. coli* directly to streams flowing into Lake Huron.

Fecal contamination from birds has also been demonstrated to cause impairment to watercourses used as drinking water sources. Benton et al. (1983) found that a serious deterioration in bacterial water quality, measured as *E. coli*, was correlated to the number of gulls roosting in a Scottish reservoir. Following the use of bioacoustic gull scaring (species-specific *Larus* gull distress calls), the numbers of both roosting gulls and *E. coli* diminished, and bacterial water quality returned to its previous characteristics.

Birds can also contribute fecal contamination leading to the impairment of recreational waters (Levesque et al. 1993; Levesque et al. 2000). Levesque et al. (1993) found ring-billed gulls contributed to the bacteriological degradation of recreational waters at a beach in the city of Quebec. After spreading food on the beach, the numbers of gulls increased rapidly, as did fecal coliform levels in the beach water. After only 2 days, and in the presence of 30 gulls, recreational water quality guidelines of 200 fecal coliforms per 100 ml had been exceeded. There was a significant correlation found between average number of gulls at the beach and fecal coliform concentrations in the water. Gulls are known to feed at refuse tips, on farm lands, and at sewage works and sewage outfalls (Gould and Fletcher 1978). Feeding at these locations could enable them to pick up *E. coli* from human or agricultural sources

9.3 Wildlife: waterborne pathogens

While sources of fecal pollution like sewage are often considered to present higher risks to human health from waterborne pathogens, wildlife sources should not be ignored. A variety of wildlife species are known to carry protozoan parasites such as *Giardia* and *Cryptosporidium*, Canada geese in particular, are known to disseminate *Cryptosporidium* oocysts and *Giardia* cysts in their fecal droppings (Graczyk et al. 1998), and to carry bacterial pathogens such as *Campylobacter jejuni* (Fallacara et al. 2001). In addition, a variety of rodents can serve as reservoirs for *Cryptosporidium*, and potentially contaminate bodies of water (Quy et al. 1999; Bajer et al. 1997). Rodents and white-tailed deer can also carry *E. coli* O157:H7 (Cizek et al. 1999). In fact, potential emergence of new waterborne pathogens of zoonotic origin is likely to be of considerable concern in the future (World Health Organization 2004).

10.0 Unintentional Releases of Organic Materials Contaminated with Microbial Pollutants of Fecal Origin

Spills of manure either directly or indirectly into watercourses stemming from the widespread handling of manure as a waste product and fertilizer is a periodic occurrence in agricultural areas. The handling and transport of septage, biosolids, and other types of non-agricultural wastes rich in organic material and in contact with human and animals sources of fecal material also present a risk of spillage and microbial pollution.

The potential significance of infrequent spills of manure on water quality at beaches is difficult to assess. A spill to a water course discharging in proximity to a recreational beach has the potential to deliver water with high concentrations of microorganisms to the shores of the lake as the spilled material is flushed from the watercourse. Unless the spills are recurrent, it would be expected that the impact would be localized to a specific area of the shoreline for a limited period of time.

A recent article (Stoneman 2004) has drawn attention to the frequency of manure spills relative to municipal sewage spills in Ontario. The author made the point that in many places over the 2002 and 2003 period, bypasses at sewage treatment plants were more frequent than manure

spills. Considering the municipalities of Huron County for which information was reported in the article, this observation also holds true. In 2003, the article noted three manure spills compared with 21 bypasses and municipal sewage spills.

Reported spills in Huron County:

The accidental release of pollutants into the natural environment are reported to the Spills Action Centre (SAC) at the Ministry of the Environment. Depending on the severity and nature of the spill, the SAC coordinates a response with area or district offices to organize containment and minimize damage. Spill reports to the SAC are stored in a database and organized by County, spills data for the years 1993 to 2002 were obtained from SAC.

The data for Huron County was manually parsed into a subset of spills that would be expected to carry some fecal pollutant load. Information on sewage bypasses for selected treatment facilities was presented in Section 5.1 and is not reiterated here. Table 24 below shows the frequency of spills according to year and type. Many of these spills directly impacted a watercourse.

Table 24: Occurrence and type of spill in Huron County between 1993 and 2002. Manure spills include both solid and liquid manure types. Data for bypasses at sewage works are not included (refer to Section 5.1)

Year	Spill type (number of occurrences)	Total number of spills
2002	manure (5)	5
2001	manure (2); silage (1)	3
2000	manure (4); septage (1)	5
1999	manure (9)	9
1998	manure (5)	5
1997	manure (5); silage (1)	6
1996	manure (5)	5
1995	manure (6); blood wastes (1)	7
1994	sewage (3); fertilizer from sewage lagoon (1)	4
1993	manure (3)	3

11.0 Discussion

The periodic elevation in the levels of the fecal indicator bacteria *E. coli* at recreational beaches of the SE shores of Lake Huron is a long-standing concern. The examination of the beach monitoring data collected by the Huron County Health Unit for the years 1993 to 2003 indicates that there are persistent sources of microbial pollutants of fecal origin along diverse areas of the Lake Huron shores of Huron County, and these sources periodically impact water quality at recreational beaches. The levels of occurrence of microbial pollutants in shoreline waters, as inferred from the indicator *E. coli*, appear to vary dramatically from year to year. However, there is no evidence of a systematic change over the past 11 years in the levels of the *E. coli* monitored at recreational beaches. Over the last 20 years and beyond, considerable effort has been devoted to trying to understand the sources of fecal pollution and to elucidate actions that might reduce the degree to which microbial pollutants are delivered to the lake from the lands draining to Lake Huron. Many of the observations and concerns on the shores of Lake Huron today are similar to those documented previously. The analysis conducted as part of this report provided no sense of either adverse or positive change in water quality conditions at beaches and shoreline of the lake since the early 1980s, however, the inability to detect changes may be due to inadequate information with which to track changes.

An aim of this report is to evaluate the causes of adverse water quality at recreational beaches based on the analysis of existing information. Potential sources of release of *E. coli* and fecal pollutants to the aquatic environment were identified over a gradient ranging from those which are well documented (e.g. sewage plants) to those where their significance as a source is less certain (e.g. septic systems) to those sources of unknown dimension (e.g. wildlife). A prevailing finding in the examination of the potential sources was that invariably, there was no, or little, information on how, or if, the pollutant loads once delivered to the lake would impact upon water quality at the shoreline (or beaches). The incomplete knowledge of the sources of *E. coli* reaching the lake combined with the sparse information on their dispersal at the shoreline, leads to the conclusion that the sources impacting shoreline water quality cannot be determined from existing information. Only the potential contributing sources can be identified, when there is a basis to expect a pollutant load to reach the shores of the lake from the source. Despite the lack of definitive answers as to what causes the periodically elevated levels of *E. coli*, there is a considerable base of information which can be used to better understand potential sources to

the lake and the factors which determine how loads of pollutants from these sources may affect water quality at the shores of the lake.

The dominant feature of the land area that drains to Lake Huron on the shores of Huron County is that lands have been modified, cultivated and utilized for agricultural activity. Previous studies, supported by recent monitoring of tributaries, indicate that agricultural activities contribute to the elevated levels of *E. coli* frequently observed in tributary waters. The high level of nitrate pollution observed in tributaries, and periodically in the nearshore of Lake Huron, clearly demonstrates the connectivity between agricultural activity, water quality in tributaries and the impact on the nearshore of the lake. Features of nitrogen loading to surface waters from agricultural activities are well documented (Neilsen et al. 1978). There is reasonable information from earlier studies to expect that agricultural activity on the SE shores of Lake Huron contributes significantly to the load of *E. coli* delivered to the nearshore of Lake Huron. However, there has been little advance over earlier work in understanding the degree to which agricultural activity in general, or specific agricultural practices in particular, contribute to the level of fecal indicator bacteria observed in the nearshore waters of Lake Huron.

It is certain that surface waters draining to the lakeshore, periodically, if not frequently, will have elevated levels of *E. coli* and will deliver loads of microbial pollutants of fecal origin to the lakeshore. A significant knowledge gap is that there is little basis to assess the scale (spatial or temporal) on which the delivery of *E. coli* via tributaries will affect *E. coli* levels along the shoreline. The size of the tributaries draining to the SE shores of Lake Huron is diverse, and while the relative volumes of water delivered among tributaries is poorly known, it is certain that it ranges from a minimal to a very substantial volume. A small volume of water delivered to the lake with a high level of contamination may affect conditions immediately adjacent to the point of discharge at the lakeshore area but will likely have a limited spatial effect away from the tributary mouth because of the rapid dilution of the tributary water. Conversely, substantial discharge volumes from a large river system, such as the Maitland River, can affect an appreciable length of shoreline as the river plume mixes with the lake, typically along the downwind shoreline.

A further difficulty in relating potential sources within a watershed to the effects at the shoreline is the uncertainty in travel potential of microbial pollutants within the watershed. An important task is the prioritization of the areas within the lands draining to Lake Huron that are most likely to contribute a load of pollutants to the lake. It is likely that the primary route for delivery of

microbial pollutants to the lakeshore from areas physically removed from the lakeshore is via tributary surface water flows. The distances from the lakeshore of upstream areas of tributaries is highly variable for the various drainage units on the SE shores of Lake Huron. In large drainage areas there are appreciable distances to travel to the lakeshore with large portions of the drainage areas far removed. The extent to which upstream areas contribute to the load of *E. coli* reaching the shoreline is not well understood. The survival rates of *E. coli* in tributaries and their travel time to the lake are key factors that will determine the proportion of the drainage areas contributing to the load reaching the lakeshore. At present, there is empirical evidence of transport distances of up to 16 km, however, the major drainage areas of SE Lake Huron reach substantially further away from the lake. Both the MVCA and ABCA have attempted to model and predict the delivery of upstream loads of fecal bacteria to Lake Huron. However, the factors that can affect the survival of *E. coli* and other microorganisms in tributary water and bed sediments are complex and do not appear to be adequately understood to allow strong predictions of survivability of *E. coli*. The usually moderate nutrient content, moderate organic levels and low clarity of tributaries on the SE shores of Lake Huron may provide conditions conducive to the survival of bacteria. There is good evidence to expect that the survival of *E. coli* within the tributary environment may be prolonged under some circumstances potentially allowing for considerable transport from upstream areas to the lake.

The Lake Huron shoreline of Huron County and SE Lake Huron, in general, is valued as property for private dwellings and there is a considerable number of permanent and seasonal homes located on the shoreline fringe. Most areas of the shoreline are not serviced by communal sewage works and rather rely on private septic systems for waste treatment. The development of the shoreline goes back many years in some areas and it is suspected that a proportion of the septic systems are dated. A trend towards conversion of seasonal to more long-term use of dwellings in recent years may be expected to increase demands on existing septic systems. There does not appear to be any direct evidence to indicate that failing septic systems are the source of the load of *E. coli* affecting an area of the shoreline. There has been, however, little field-based study of the effects of shoreline dwelling septic systems on water quality in the nearshore of SE Lake Huron. Previous information gathering exercises have documented the distribution and features of septic systems in Huron County (Huron County 1993) and the areas served by the MVCA (Fuller and Foran 1989) and ABCA (Hocking and Dean 1989) and these have identified concerns that the failure of domestic septic systems are contributing to the microbial pollution of Lake Huron. The work of the MVCA under the CURB program identified septic systems as the largest single source of fecal coliform load to

tributaries, and to Lake Huron via tributaries, based on a modeled analysis of fecal coliform sources in the MVCA watershed. The analysis assumed a 30% failure rate for the estimated 8,200 households with private septic systems in the watershed. The analysis did not separate the potential direct effects of shoreline septic systems on the lakeshore from the overall loading from septic systems throughout the drainage area and delivered via tributaries. However, it was noted that the potential for direct impact on the lake from the approximately 1,400 septic systems along the 48 km of the MVCA Lake Huron shoreline needed to be investigated. A similar modeling of loads of fecal coliforms to tributaries and Lake Huron was conducted by the ABCA as part of the CURB program (Hocking and Dean 1989) and this analysis also identified failure of septic systems as the single largest predicted source of loading to the aquatic environment. Of the 7,347 homes serviced by septic systems, 1038 were located on the shores of the Lake Huron. A higher failure rate of 60% was used in the estimation of load by the ABCA, the basis of which is not well established in the report. Overall, septic systems were identified as contributing 77% of the bacteria delivered to Lake Huron. The contribution of shoreline dwellings were calculated separately by the ABCA and were assessed as accounting for 37% of the fecal coliform load reaching Lake Huron all of which was attributed to septic systems (Hocking and Dean 1989). The relatively high relative contribution of septic systems to the loads of fecal bacteria in both modeling exercises is suspect and likely stems from a combination of factors. There is wide-scale use of private septic systems for waste disposal resulting in large numbers of septic systems over the landscape. The models used assumptions which yielded high loading estimates; failure rates were estimated at 30 to 60% and the environmental release rate from a failing septic system was estimated at 50% to 100% of the septic system load. The overall significance of leakage of microbial pollutants from septic systems on water quality in the nearshore of Lake Huron, either as delivered directly from shoreline dwellings, or indirectly via tributaries, remains unclear and warrants investigation.

The variability in patterns of occurrence of *E. coli* among the recreational beaches monitored by the HCHU examined in this report suggests that conditions at individual beaches are determined by different, possibly unique, sets of factors acting at the individual beaches. The physical conditions and anthropogenic features of beaches are, to some degree, unique among areas of shoreline and it is intuitive that water quality at even closely placed beaches may be affected differently by a suite of different factors. Proximity to tributary mouths, the extent of shoreline development and steepness of the beach gradients are some of the features which vary between shoreline areas and affect the delivery of pollutants to beaches. Over the 11 years of data examined, the frequency of *E. coli* PWQO exceedances have been distinctly

different among beaches. There have been a number of events when all beaches appeared to be responding in a similar manner to an environmental driver, but overall the temporal patterns in occurrence of elevated levels of *E. coli* varied widely among beaches. The temporal correlation in patterns of *E. coli* that would be expected if the shoreline was responding to a common set of environmental factors is lacking. The practical significance of the apparent uniqueness of different beach areas is that beaches, or reaches of shoreline, may need to be examined on a case by case basis when assessing sources of pollutants impacting the shoreline, and when evaluating abatement strategies to improve beach quality. The overall degree to which adverse conditions at a beach are the result of sources in the immediate area or from pollutants transported to the beach via drains or tributaries from further a field is a complex question which needs resolution.

In the lake environment, elevated levels of *E. coli* appear to be localized at the shoreline. Recent water quality monitoring in the nearshore of Lake Huron indicates that levels of *E. coli* in the open lake (depth >3m), away from the immediate shoreline, are low. When elevated levels of *E. coli* are observed in the lake it is primarily at the shoreline and in shallow water and in some instances in areas directly impacted by tributary plumes. Consequently, in-lake transport of *E. coli* is likely dominated by processes which play out at the shoreline notably onshore-longshore flow of tributary plumes and longshore transport of particulate material.

There is a limited understanding of the transport of *E. coli* in the nearshore of Lake Huron which is in part due to the uncertainty in the survival of *E. coli* upon release into the lake environment. The survival time of *E. coli* in the water column is expected to be limited, on the order of days. In the usually phosphorus poor, low organic, high clarity waters of Lake Huron conditions are not expected to be conducive to extend the survival of *E. coli*. However, there will be fluctuations in water quality that will likely impact on the die-off rates of *E. coli*. Variations in water transparency, nutrient concentrations and levels of organic material due to natural and anthropogenic factors are expected to and will likely affect survival times in the nearshore.

On the shores of Huron County, the spatial extent of the mixing gradients of most tributaries in the lake will be limited except possibly during high flow conditions due to the dilution with the vast volume of Lake Huron. The occurrence of elevated levels of *E. coli* attributable to tributary inputs to the lake is expected to be highly localized and sporadic in time. The beach areas likely to be impacted by mixing gradients from tributaries will likely be physically near the tributary

mouths, when tributary flow is high, water quality in the tributary is poor, and when wind conditions push the mixing gradient against the shores of the beach.

The association of *E. coli* with particulate material is thought to be a key mechanism by which survival and transport in the lake environment is enhanced. The enhanced survival is thought to be due to some combination of physical protection from solar radiation, proximity to nutrient and energy sources, and possibly a reduction in vulnerability to protozoan grazers. A proportion of the *E. coli* load delivered to the lakeshore by tributaries will be associated with particulate material much of which will settle out of the water column over the mixing gradient of the tributary with the lake. Evidence from past studies in Lake Huron and elsewhere indicate that *E. coli* may survive for an extended period of time in the lake associated with surficial sediment. Any factors which increase the survival time in the lake will increase the effective area and time period that a load of bacteria delivered to the lake may impact upon beach conditions. It is expected that particulate material deposited on the shallow lake bottom over the dilution gradients of the tributary plumes will not persist in these locations. The nearshore of SE Lake Huron is a high-energy and non-depositional environment. Water movement resulting from onshore winds periodically yield sufficient mixing energy at the lake bottom to resuspended bed sediments into the water column where it will be transported by lake currents. In this way, it is hypothesized that *E. coli* can be delivered to beach areas and result in adverse beach impacts attributable to sources separated in time and space from the impacted beach. Evidence for this mechanism of impact can be inferred from complementary pieces of information. There is good reason to suspect that tributary-mediated delivery of particulates contaminated with *E. coli* occurs on SE shore of Lake Huron. The potential for extended survival of *E. coli* associated with particulate material and in surficial lake sediments has been established and there is empirical evidence that levels of *E. coli* at beaches sometimes increase during periods of windy weather.

At present it is difficult to put the significance of particle associated *E. coli* persistence and redistribution in the lake into the broader context of factors affecting shoreline conditions. A better understanding of the ranges of *E. coli* survival times in the lake environment over the seasonal cycle is needed since the loading of water from the landscape to the lake varies significantly over the seasons. The highest flow in tributaries, and presumably the greatest loading of waterborne pollutants, occurs early to mid spring when flows are typically at seasonal highs. Lake temperatures are low at this time and recreational usage of the lake is expected to

be light. The long-term fate of the potentially high *E.coli* loads delivered to the lake at these times is poorly understood.

The load of particulate material delivered by the Maitland River will be substantially higher than other tributaries simply by virtue of its relative discharge volume. If redistribution of bacterial pollutants to the shoreline by transport of particulate material on the lake bed is a major factor in the delivery of *E. coli* to recreational beaches, it can be hypothesized that impacts would be most pronounced at the Goderich beaches and that far-field impacts of particulate material would be most notable at the beaches nearest to Goderich and particularly to the north (Points Farm and Sunset Beach) in the direction of the prevailing nearshore circulation. Examination of the HCHU beach monitoring data does not suggest any spatial correlation among the patterns of occurrence of *E. coli* among beaches moving away from Goderich, however, there may be too many other factors affecting conditions at the beaches to resolve a pattern. The significance of the loading of particulate material to the lake by the many smaller tributaries is unknown. While their relevance cannot be discounted it seems more likely that the effects from these smaller tributaries on beaches are more likely to be directly associated with the mixing zone of the discharge effluent than the redistribution of tributary-derived bed sediments.

Studies at the scale of individual beaches suggest that the dynamics of particulate material on the scale of individual beaches may also be important in understanding the occurrence of elevated *E. coli* and the sources of their loadings. There are a number of studies which indicate that beach sand and interstitial water within a beach area can contain elevated levels of *E. coli* which can conceivably contribute to elevated levels of bacteria in the lake water at the beach front. It is plausible that the often noted correlation between wind and elevated levels of *E. coli* detected at the shoreline may in part be due to physical beach disturbances resulting in the bacteria being washed from the beach to the lake shore. The sources which may account for the elevated bacteria in the beach material are diverse and possibly local in nature and likely include, bird droppings, anthropogenic recreational use, surface drains to the beach and poor quality shallow groundwater.

Also relevant at the beach scale, recent research has raised questions concerning a potential for enhanced survival, and possible growth of *E. coli* in association with benthic algae, notably the filamentous green alga *Cladophora*, on the shores of Lake Michigan (Whitman et al. 2003; Byappanahalli et al. 2003). The possible extent to which occurrence of *E.coli* in shoreline water quality is affected by the presence of benthic algae or other biological communities rich in

organic material, either growing on lake bottom or washed upon shoreline remains unclear. Any modifying effect of naturally occurring algal material along the shoreline on the persistence and distribution of *E. coli* loaded to the lake from fecal pollution sources will likely depend on the type and amount of material, and the physical-chemical conditions at the shoreline. A survey was conducted in 2003 by the MOE to evaluate the occurrence of nuisance benthic algae along the shores of southeastern Lake Huron (Howell 2004). Observations at sites along the shoreline from Point Clark to south of Bayfield indicated that limited amounts of algae and other plant material were detected washed upon the shoreline at some locations at the time of the survey in July. There is little basis to either suggest that algal and plant growth along the shoreline affects occurrence of *E. coli*, or reject the possibility.

There are diverse anthropogenic and natural sources, on any landscape, that will generate loads of microbial fecal pollutants to the environment and potentially impact conditions at recreational beaches and at the shoreline of a lake. The lands that drain to the shores of Lake Huron in Huron County support a number of different of land use practices that are known to generate loads of fecal pollutants to surface waters, a portion of which may ultimately reach the shores of Lake Huron. It can be assumed that livestock operations will generate a volume of manure, the handling of which will result in some release of fecal pollutants to the environment. The widespread use of tile drains in combination with the practice of spreading manure on fields as fertilizer will result in the transfer of fecal pollutants from fields to surface drains. The sewage treatment facilities will periodically receive inputs in excess of their capacities and will inevitably release poorly treated sewage to water courses or directly to the lake. Furthermore, a low level of ongoing loading of microbial fecal pollutants will be associated with discharges from sewage works. Any surface water reaching the lake which has drained over a developed area will likely be affected to some degree by microbial pollution. While the significance of leakage from septic systems associated with shoreline properties is unclear, the number of systems and the age of some, suggest that there will possibly be some areas where problems are encountered. The usage of the shoreline as a wildlife habitat, in the case of gulls and geese, or as beaches by humans may result in a degree of input of microbial pollutants to the lake water or beach. At present, we do not adequately understand which of the potential sources most affects the areas of shoreline where adverse water quality has been observed on the shoreline of Lake Huron. Is the occurrence of elevated levels of *E. coli* at a beach the result of the interplay between a unique set of source loadings to the beach environment and shaped by the unique physical character of the beach? Alternatively, do a limited number of key sources, delivered to beaches by relatively few modes of transportation account for most of the adverse events observed

among beaches? As part of the work by the MVCA and ABCA under the CURB program, modeling exercises were conducted in which the potential loads of fecal bacteria from a wide range of key suspect sources in the drainage areas of the conservation areas were estimated.

Ultimately, a model-based analysis of generation and delivery of microbial pollutants to Lake Huron based on up-to-date demographic and land-use information and directed by current state of knowledge on microbial transport and survival, and truthed by a vigorous field program may be needed to manage microbial pollution of fecal origin at the shores of Lake Huron. However, it is the opinion of the Committee that more basic information gaps need to be addressed before such a difficult undertaking is recommended. Previous work has not adequately defined the linkages between potential sources of fecal pollutants delivered to the shoreline of Lake Huron and the adverse water quality at recreational beaches. In the short-term the Committee recommends work to better identify the mechanisms by which beach water quality is impacted by potential sources of pollution. The objective of such work is to clarify the spatial and temporal scales on which potential sources of fecal pollutants impact upon water quality at recreational beaches. It is the opinion of the Committee that a better definition of how water quality impacts are occurring will help focus future, more synthetic, analyses of the diversity of potential sources over the landscape and in the short-term will provide a better basis to target areas and time of year for remedial actions.

12.0 Recommendations

It is through an improved understanding of the linkages between the loading of the pollutants to the lake and the effects on water quality at recreational beaches that the dominant, and/or the most environmentally manageable, of the drivers of adverse water quality may be appreciated. The relative extent of local, beach-scale sources compared to broader regional influences on microbial quality at the shoreline of Lake Huron remains an important question, the answer to which may potentially shape the geographic scale on which remedial measures are implemented. There are several facets of this issue which require further clarification and could potentially promote effective targeting and prioritization of remedial actions.

The recommendations which follow propose an inter-related suite of four studies designed to: i) evaluate mechanisms by which key potential pollution sources impact upon beach water quality,

ii) better resolve the scale of effect (spatial and temporal) of these potential sources, and, iii) evaluate suspected sources of bacterial pollutants on shoreline water quality. The geographic extent of the proposed studies vary in scale from a strip of beach, to the immediate shoreline impacted by a small tributary, to an expanse of shoreline impacted by a large river system. The purpose of the nested geographic scales among studies is to provide insight on the landscape origin of the sources that impact beach water quality. Emphasis is placed on evaluating mechanisms of delivery of pollutants to shoreline waters and their subsequent delivery to beaches, because of the need to understand pollutant transport processes which determine their point of origin.

Study 1. Investigation of beach-scale sources on the occurrence of E. coli at the shoreline

Activities occurring at or adjacent to a beach may indeed be responsible for the adverse levels of microbial pollutants observed at a beach. Currently, little information is available to assess the relative impact of local sources of inputs compared to larger-scale inputs from the entire drainage area or inputs through transport from more distal shoreline locations. The potential water quality impacts of failing septic systems adjacent to beaches and the impact of drains and small watercourses delivering surface runoff from adjacent lands are additional information gaps that need to be filled. Furthermore, the relative role of beach sand as a source of bacteria in the water has not been demonstrated yet many recent studies in Ontario, Michigan and California have found that populations of fecal bacteria, including *E. coli*, are present in the sands of beaches immediately adjacent to the lake (swash zone).

A study is recommended to fill these identified gaps, in particular, to examine the sources and loads of *E. coli* to the beach environment and the movement of these loads to the shoreline waters. The candidate study area should be a shoreline with residential developments, where beach monitoring indicates periodically elevated levels of *E. coli*, but, where there is a limited drainage area backing the shoreline. The over-arching objective of the proposed study should be to examine the significance of a variety of local small-scale sources of *E. coli* as a driver of adverse beach water quality. The study should attempt to establish linkages between discrete local sources of *E. coli* and beach water quality impacts by verifying candidate sources and further evaluating the mechanisms by which the load from these sources reach the lakeshore.

Candidate sources include: discharge from septic systems, surface drains, groundwater beach discharge, wildlife beach activity, beach recreational use and populations of *E. coli* in beach interstitial waters. Clearly delineating the overall significance of local sources relative to larger-scale loads on the incidence of adverse water quality will be challenging. However, the monitoring of physical processes in the nearshore of the lake and the application of microbial tracers and/or emerging microbial source tracking methodologies may provide insight on the degree to which local sources drive local water quality conditions.

Study 2. Investigation of the impact of tributary discharge from a small agriculturally-dominated watershed on E. coli occurrence at an adjacent beach

The beach water quality impact of the numerous small to moderate-size tributaries to the shores of Lake Huron is poorly understood. However, monitoring results indicate that the levels of *E. coli* in tributaries and drains are periodically elevated, suggesting their possible importance in influencing *E. coli*-based beach water quality. Given the abundance of watercourses reaching the coastline of southeastern Lake Huron, recreational beaches in Huron County are inevitably close to tributary mouths to varying degrees. Despite their potential for delivering significant loads of *E. coli* to the shoreline, much uncertainty remains with respect to the spatial or temporal extent of impact of small discharge volumes on shoreline water quality. The extent of impact on water quality will be highly variable over time and highly dependent on the volume and quality of the discharge, as well as depend on the mixing conditions at the tributary mouth at the time of discharge. The water quality impact will also depend on the watershed drained by these various small and moderate sized tributaries and any point sources of inputs to them. Land-use in these smaller tributary watersheds is dominated by agriculture with scattered communities of limited populations, and point sources of inputs, such as from sewage treatment plant effluents, are essentially absent, found primarily within the drainage areas of the larger tributaries within Huron County.

A study is recommended in which the hydrological, land-use and water quality characteristics of a representative small tributary are monitored. This monitoring should be complemented by a concurrent assessment of the impact mechanism of the tributary discharge on an adjacent beach and an adjacent stretch of the shoreline. The primary objective of the study would be to elucidate the relative role of smaller sized tributaries in driving adverse beach water quality at the shoreline of the lake. The study should examine how climatic and anthropogenic factors

influence the extent and timing of any impact that the tributary discharge has on *E. coli* levels along the shoreline.

The study should document the spatial and temporal features of the impacts on water quality at the shoreline adjacent to the tributary mouth, using hydrodynamic modeling supported by in-lake water quality monitoring and instrument-based monitoring of physical conditions in the nearshore. By linking the extent of adverse water quality at the shoreline to tributary water quality features, discharge volume and lake mixing conditions, it should be possible to begin scaling the tributary impacts on the shoreline to the environmental factors that are driving water quality, discharge and mixing conditions over time. Distinguishing tributary discharge adverse water quality effects from those of sources along the immediate shoreline and/or from loads delivered to the shoreline by longshore transport will be necessary to conclusively isolate tributary discharge impacts. This will be a technically challenging task, however, chemical-based methods can be used to track tributary waters in the lake and this approach used in combination with microbial source tracking methodologies may provide a basis to distinguish tributary *E. coli* loads from other source loads.

Previous studies have shown a pronounced temporal variability in tributary water quality including levels of *E. coli*, suggesting a need to understand the basis of the fluctuations in water quality to ultimately recognize the potential effects of tributary discharge on the shoreline. The proposed study should attempt to assess how water quality in the lower tributary varies over time as a function of climatic and hydrologic factors known to affect water quality. In addition to the transport-related or dilution-related features correlated with weather and hydrology, it is expected that tributary levels of *E. coli* will vary as a function of the timing of the loads delivered to the landscape from various sources. To the extent possible, the study should therefore collect information on the temporal features of fecal pollutant loadings over the drainage area. The purpose of this aspect of the study should be to provide information on how tributary water quality may be expected to vary over time as a determinant of potential effects on the shoreline water quality.

Both the volume and the quality of the discharge from a tributary are important factors determining the potential impacts upon the shoreline. Volume of discharge will influence the size of the mixing zone in the lake and will, along with pollutant concentrations (i.e. water quality), drive the overall load of contaminants reaching the lakeshore. The proposed study will

necessarily include both field and model-based examinations of the discharge characteristics of the tributary under study.

The findings of an intensive case study such as proposed here can provide a basis to frame predictions on the possible conditions at other areas of shoreline where watercourses discharge to the lake. Due to the wide variability in size of drainage areas and discharge volumes between watercourses, there is a need to understand how discharge characteristics vary among watercourses if the integrated impact of tributary inputs to the shoreline is to be assessed.

Study 3. Characterization of the Hydrology of Shoreline Tributaries of Huron County

There are a substantial number of watercourses which drain to the Lake Huron shores of Huron County, most of which are ungauged with the exception of the few larger tributaries (e.g. Maitland River, Bayfield River). However, the close proximity between the point of discharge of these smaller tributaries to the lake, and beach areas in general, suggests even minor watercourses may periodically affect beach water quality. The discharge patterns of the watercourses are likely highly variable over seasons and between years. Many of the smaller tributaries likely dry up or are reduced to minimal flows over the summer period except in response to wet weather events, while peak flows are expected during the spring melt and late in the spring and fall when precipitation events are more likely. In some cases, groundwater recharge may contribute to flows.

A study is recommended in which the hydrological features of a number of watercourses of representative sizes are evaluated to provide information on the discharge characteristics of watercourses to the shoreline of the lake. There is an uncertainty regarding the degree to which the delivery of pollutants by tributaries impacts upon water quality at recreational beaches over the bathing season when discharge rates may be low but possibly responsive to weather conditions. The volumes of water being delivered to the lake will greatly influence the size of the mixing zone. Volume is therefore a key factor in determining where and when the mixing zone extends to areas of the shoreline utilized for recreational activities, potentially having a direct impact on beach water quality.

A combination of hydrological modeling and field-based data collections should be used to provide information on expected patterns of discharge, and the variability in these patterns between watercourses across a gradient of sizes of drainage areas. The study should include an examination of the natural and anthropogenic factors contributing to short- and long-term temporal variability in discharge to the lake. The purpose of the study is to provide a basis to anticipate where, when and in what temporal sequence the watercourses to the lake may have an impact on water quality at the shores of the lake by virtue of the volume of water that they deliver.

Study 4. Examination of potential impacts of loading of particle-bound E. coli from a large river system to the lakeshore

Previous studies in Lake Huron have noted the role of particulate material in facilitating the transport and survival of *E. coli*. Bacteria may be found bound to the charged surfaces of particles composing bed sediments as well as to the suspended material in the water column. Within tributaries to the lake, *E. coli* associated with particulate material is transported downstream and is potentially discharged to the lake as particulate material. This association of *E. coli* with particulates is thought to prolong the survival of the bacteria in both the tributary and lake environments through protection from solar radiation, the supply of nutrients or possibly by providing a refuge from microbial grazers. Furthermore, the survival of particle-bound *E. coli*, once reaching the shores of the lake, is possibly enhanced as the particulate material settles out of the water column forming bed sediments. On the high energy shores of southeastern Lake Huron, deposits of surficial sediments are unstable and highly erosional. Consequently, bacteria deposited on the lake bottom at one time and in one place may subsequently be dispersed to other locations as the bed sediments continue to erode and deposit. This potentially expands the spatial and temporal scale of impact of the pollutant loads beyond that expected if the bacteria were suspended in the water column free of particulate material.

Despite the importance of bacteria-particle associations in *E. coli* dynamics around areas of the shoreline with appreciable tributary-based inputs of particulate material, there is little understanding of the spatial and temporal scales over which particle-bound loads of *E. coli* impact upon water quality on the shores of the lake. It is recommended that a study be conducted to investigate the transport and fate of loads of particle-associated *E. coli* delivered to

the shoreline from a large tributary to determine the potential to impact upon water quality at the shoreline.

The potential for adverse effects on water quality during the recreational season from the loading of particulate-bound *E. coli* to the lakeshore will depend on the time of year of bacterial delivery, the bacterial survival time in surficial sediments, the resuspension of particulate material and its movement in the nearshore of the lake. A focus of this study should be on evaluating the potential for tributary derived *E. coli* survival and transport along the shores of the lake. A key question to be addressed is whether adverse effects attributable to tributary loads of *E. coli* associated with particulate material are restricted to the mixing zone of the tributary (and area of greatest sediment deposition), or whether *E. coli* -laden particles can be dispersed to more distant areas of the shoreline with concentrations of *E. coli* elevated enough to impact water quality when surficial sediments are eroded.

The proposed study should also examine the time-frame over which loads of *E. coli* to the lakeshore may persist and contribute to *E. coli* measured at the shoreline during the summer period.

The proposed study will require detailed monitoring and physical modeling of the flux of particulate material from the mouth of a tributary over an extended area of shoreline in the lake. The spatial and temporal patterns of occurrence and persistence of *E. coli* in association with suspended particulates at the tributary mouth, in the nearshore of the lake and in the bed sediments of the lake will require evaluation. The difficulty level of the proposed study is high yet proportional to the potential value of information on role of particle-associated flux of *E. coli* in driving adverse levels of *E. coli* in shoreline waters. The present lack of understanding of particle-associated flux of *E. coli* is a significant impediment to understanding the linkages between pollution sources to adverse levels of *E. coli* at the shoreline of the lake.

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14.0 Acknowledgements

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Appendix A : Selected Results of Water Quality Monitoring in the lower reaches of the Bayfield, Maitland and Suageen Rivers and adjacent nearshore areas of Lake Huron in 2003 by the Environmental Monitoring and Reporting Branch, Ontario Ministry of the Environment.

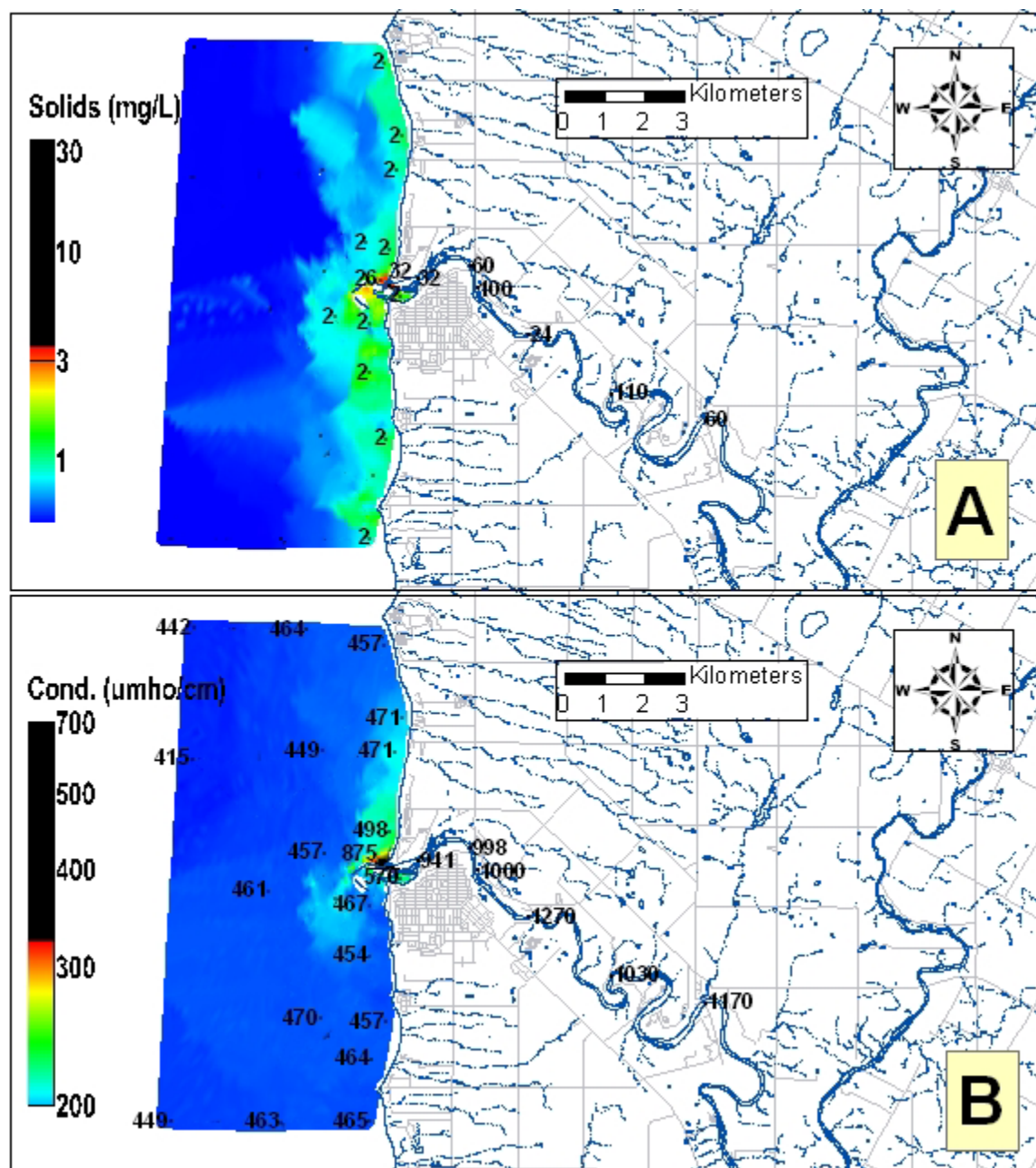


Figure A1: Surface water quality near the Maitland River, July 8, 2003; (A) Suspended solids (color image and scale bar) and *E. coli* (dots are counts/100 mL), and (B) Conductivity (color image and scale bar) and nitrates (dots are µg/L).

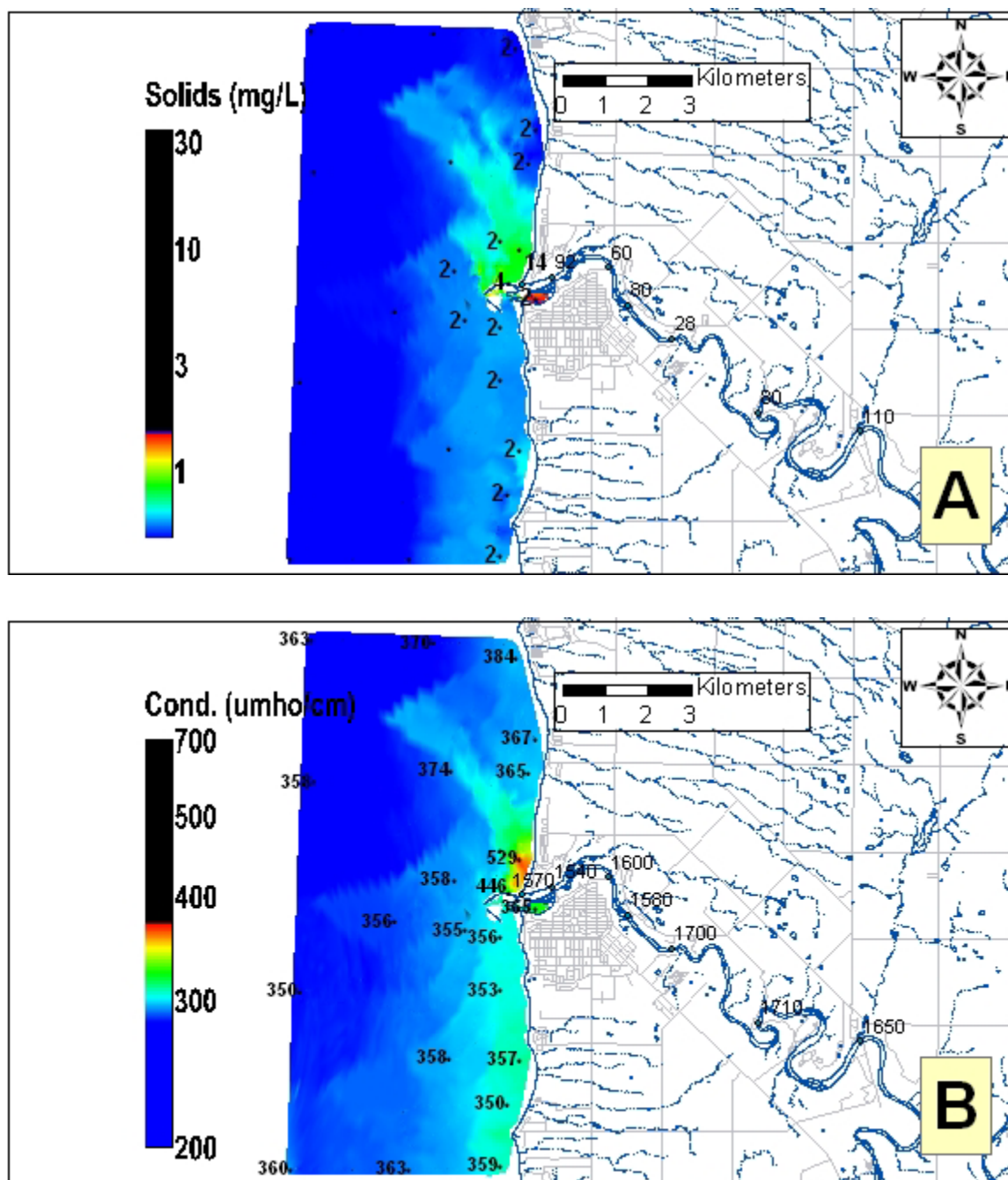


Figure A2: Surface water quality near the Maitland River, July 31, 2003; (A) Suspended solids (color image and scale bar) and *E. coli* (dots are counts/100 mL), and (B) Conductivity (color image and scale bar) and nitrates (dots are $\mu\text{g/L}$).

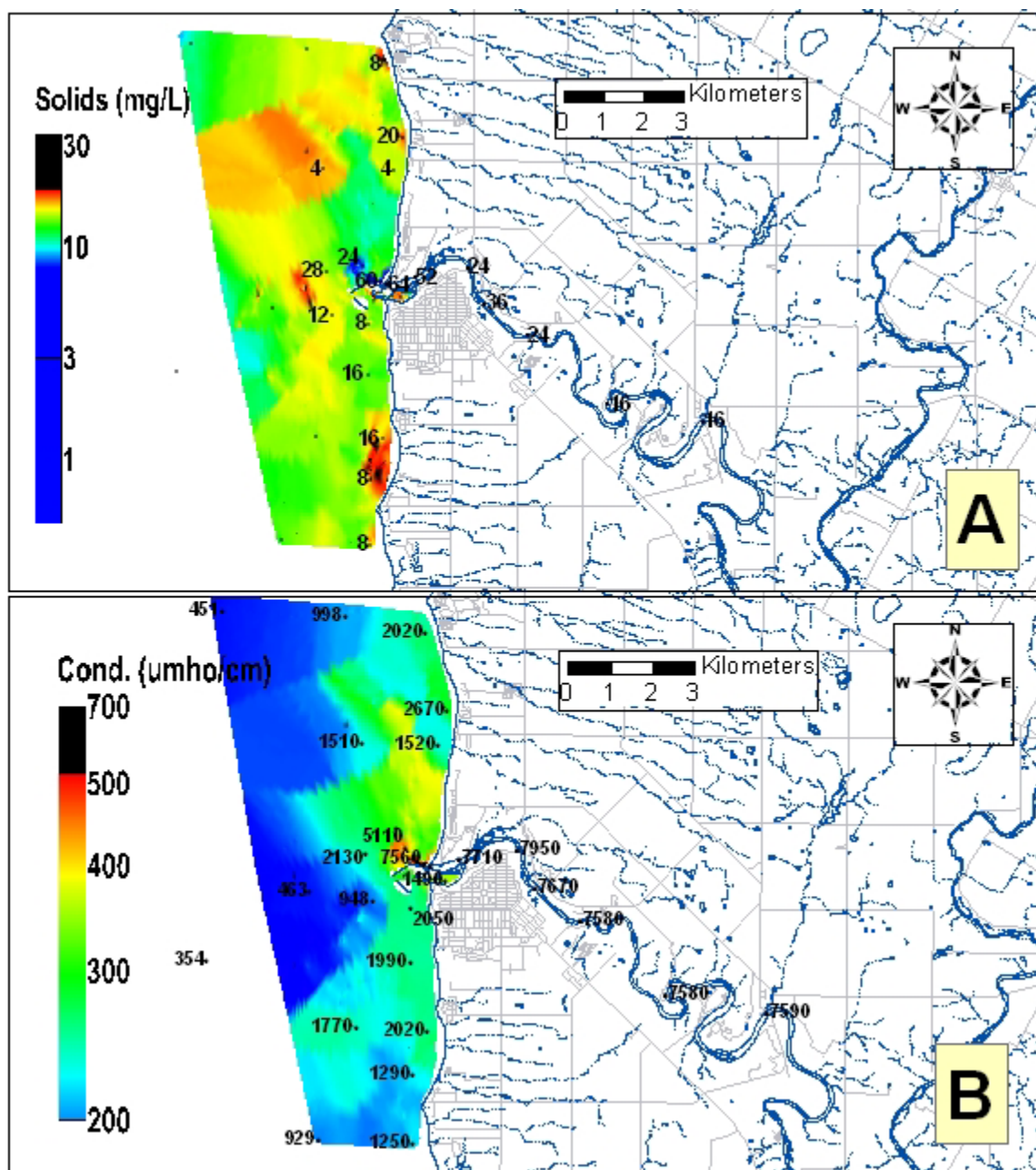


Figure A3: Surface water quality near the Maitland River, November 23, 2003; (A) Suspended solids (color image and scale bar) and *E. coli* (dots are counts/100 mL, missing values were below detection), and (B) Conductivity (color image and scale bar) and nitrates (dots are µg/L).

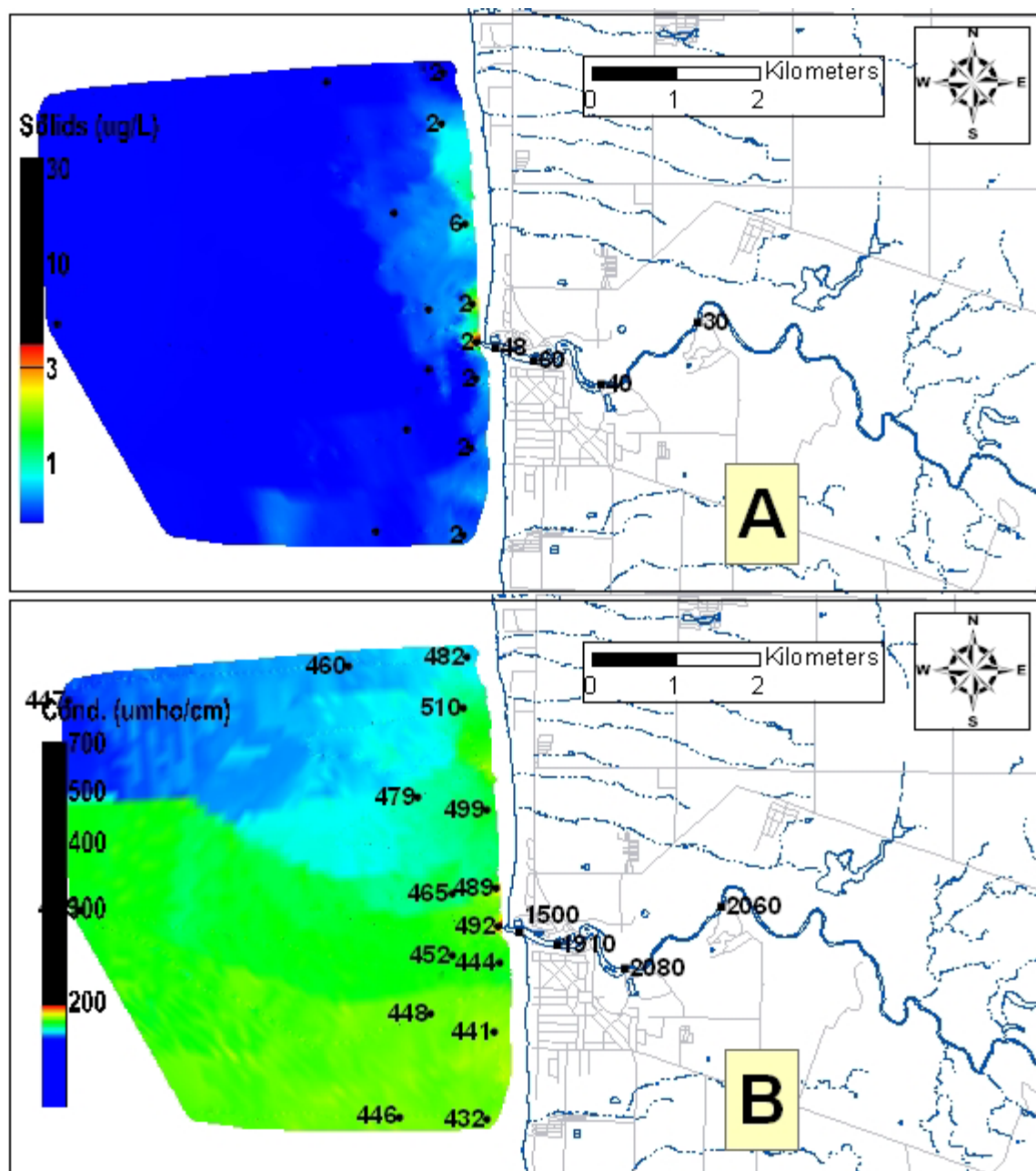


Figure A4. Surface water quality near the Bayfield River, July 7, 2003; (A) Suspended solids (color image and scale bar) and *E. coli* (dots are counts/100 mL), and (B) Conductivity (color image and scale bar) and nitrates (dots are ug/L). Results for two additional river sites beyond the east edge of the map are not shown.

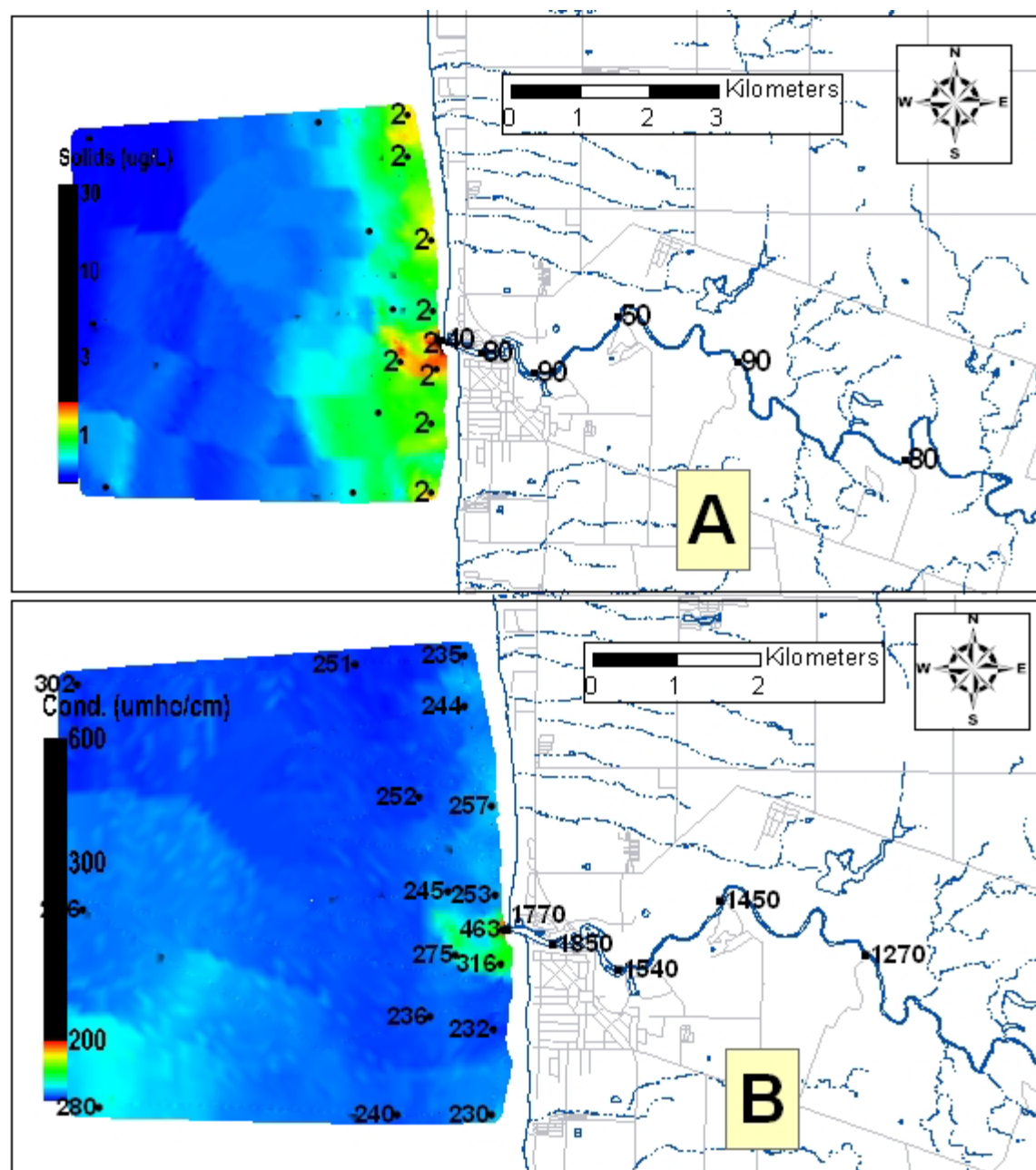


Figure A5. Surface water quality near the Bayfield River, September 18, 2003; (A) Suspended solids (color image and scale bar) and *E. coli* (dots are counts/100 mL, missing values were below detection), and (B) Conductivity (color image and scale bar) and nitrates (dots are $\mu\text{g/L}$).

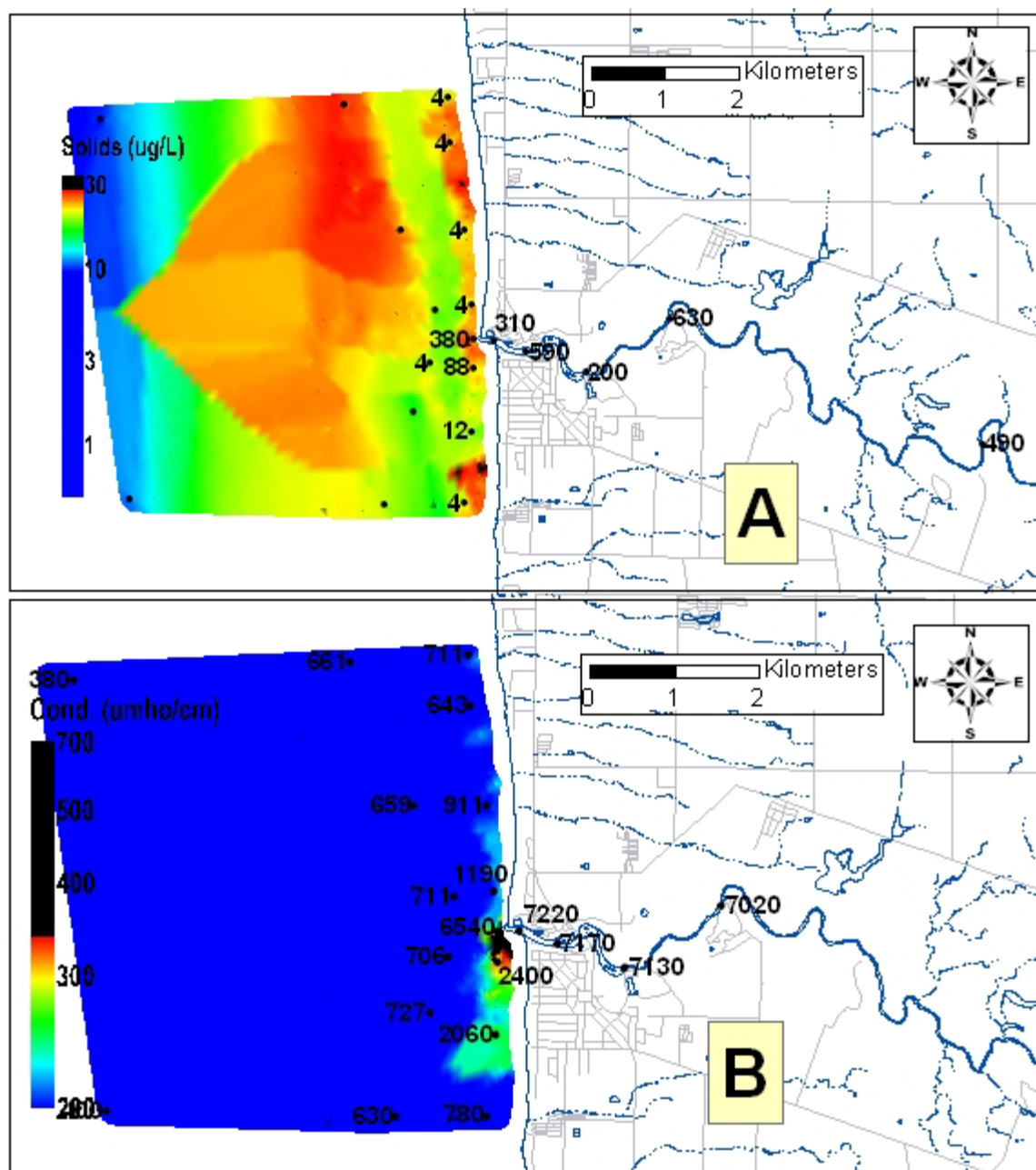


Figure A6. Surface water quality near the Bayfield River, November 19, 2003; (A) Suspended solids (color image and scale bar) and *E. coli* (dots are counts/100 mL, missing values were below detection), and (B) Conductivity (color image and scale bar) and nitrates (dots are $\mu\text{g/L}$). Results for one additional river sites beyond the east edge of the map is not shown.