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# The Relative Importance of Road Density and Physical Watershed Features in Determining Coastal Marsh Water Quality in Georgian Bay

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**Abstract** We used a GIS-based approach to examine the influence of road density and physical watershed features (watershed size, wetland cover, and bedrock type) on water quality in coastal marshes of Georgian Bay, Ontario. We created a GIS that included landscape information and water-quality data from a 9-year synoptic survey of 105 coastal marshes covering 28 quaternary watersheds. Multiple regressions and partial correlations were used to discern confounding effects of human-induced (road density) versus natural physical watershed determinants of water quality. Road density was the dominant factor influencing many water quality variables, showing positive correlations with specific conductivity (COND), total suspended solids (TSS), and inorganic suspended solids (ISS) and a negative correlation with overall Water Quality Index scores. Road density also showed positive correlations with total nitrate nitrogen (TNN) and total phosphorus (TP). By comparison, larger watershed area was the main factor leading to elevated TP concentrations. The proportion of the watershed occupied by wetlands explained the largest amount of variation in TNN concentrations (negative correlation) and was also negatively correlated with COND and positively correlated with TSS and ISS when we controlled for road density. Bedrock type did not have a significant effect in any of the models. Our findings suggest that road density is currently the overriding factor governing water quality of coastal marshes in Georgian Bay during the summer low-flow period. We recommend that natural variation in physical watershed characteristics be considered when developing water quality standards and management practices for freshwater coastal areas.

**Keywords** Wetland · Watershed · Water quality · Anthropogenic stress · Geographic information system · Road density

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Within the Laurentian Great Lakes basin, coastal wetland water quality has been closely linked to the magnitude of anthropogenic stress (i.e., the degree of urban and agricultural development) on these systems (e.g., Crosbie and Chow-Fraser 1999; Chow-Fraser 2006; Trebitz and others 2007; Morrice and others 2008). It has been demonstrated that suspended solid, nutrient, and ionic concentrations in wetlands are strongly related to a gradient of agricultural and/or urban intensity in the watershed (Chow-Fraser 2006; Trebitz and others 2007). Further, Morrice and others (2008) have found that human population is the strongest independent predictor of total phosphorus levels, and a significant factor affecting several other water quality variables. These studies have occurred across basin-wide scales within the United States (Trebitz and others 2007; Morrice and others 2008) or both the United States and Canada (Chow-Fraser 2006) and have therefore captured conditions across broad gradients of human disturbance.

Currently, little information is available on water-quality impacts in relatively remote areas of the Great Lakes with low levels of human development; in such areas, hydrological and geological factors can be expected to play a significant role in determining near-shore water quality and have the potential to confound effects of human activities. Eastern and northern Georgian Bay, Lake Huron, has had limited human development in comparison with the lower

Great Lakes (Lake Erie and Lake Ontario) because shallow soils and underlying granitic bedrock in all but the most southern portion of the eastern coast render it poor for agriculture (Weiler 1988). Human activity in the area occurs mainly in the form of recreational and cottage development. The lack of more land- and resource-intensive development projects has allowed coastal wetlands in the Bay to remain among Ontario's most pristine (Chow-Fraser 2006; Cvetkovic 2008).

Determining an appropriate measure of anthropogenic stress for sparsely populated regions such as Georgian Bay can be a challenging task. Common metrics for quantifying stress within the Great Lakes basin include measures of agricultural intensity, atmospheric deposition, human population density, road density, landcover, and density of point sources within watersheds (Houlahan and Findlay 2004; Host and others 2005; Danz and others 2007; Trebitz and others 2007; Morrice and others 2008). Many of these metrics are inappropriate for use in eastern and northern Georgian Bay due to the minimal amounts of industry and agriculture. Road density is an attractive metric because road data are widely available in Canada and relatively simple to interpret. The creation of road networks provides access to previously remote areas; consequently, an increase in road density is often associated with an increase in housing density and population density and a decrease in forested area on lakeshores in North America (Hawbaker and others 2005; Wolter and others 2006; Danz and others 2007). Further, Danz and others (2007) have shown that road density and other measures of human population are strongly correlated with an overall stress index for the Great Lakes basin that incorporated multiple types of anthropogenic impacts on coastal wetland water quality.

Accounting for natural variation in water chemistry among regions that results from variation in physical watershed features is important in conservation of aquatic ecosystems since water quality standards and regulations are best established when there is knowledge of the natural (reference) state (Keough and others 1999). Research on small inland lakes and rivers elsewhere in North America has provided some insight into the capacity for physical processes in watersheds to affect water quality dynamics (e.g., Johnston and others 1990; D'Arcy and Carignan 1997; Dillon and Molot 1997; Devito and others 2000; Eimers and others 2008). The size of watersheds can play a crucial role in determining the chemistry of tributary outflow. Larger watersheds can produce higher nutrient concentrations in receiving waters because they generate larger volumes of runoff with greater opportunity for physical and chemical alteration of water as it flows over and through the land toward the outlet (Johnston and others 1990). Landscape features such as bedrock and upstream wetlands may also play a role. Runoff in temperate Precambrian Shield regions is topographically controlled (D'Arcy and Carignan 1997), with wetlands positioned in low-lying areas (Devito and Hill 1997). As a result, much of the generated runoff passes through wetlands as it moves toward the outlet. These inundated upstream areas often have a high efficiency for denitrification (Whigham and others 1988; Seitzinger 1994) and collection of organic matter and sediments that can store phosphorus and sulfur (Richardson and others 1997; Mandernack and others 2000). Therefore, areal extent of wetlands within a watershed may be related to nutrient and ionic concentrations in runoff (Johnston and others 1990; Dillon and Molot 1997).

While previous work in Georgian Bay has shown significant variation in coastal marsh water quality in relation to varying levels of human disturbance throughout the Bay (Cvetkovic 2008), there is also considerable natural variation in drainage basin characteristics, and the relative importance of each factor in determining coastal water quality has not yet been ascertained. The goal of this paper is to separate the effects that road density, watershed size, wetland cover, and bedrock type have on water quality. Based on the well-documented potential for human development to impact coastal freshwater ecosystems by increasing nutrient and sediment loadings (Chow-Fraser 2006; Trebitz and others 2007; Morrice and others 2008), we expect road density to explain a significant amount of variation in water chemistry variables such as phosphorus, conductivity, and suspended sediments, even at the low end of the human disturbance gradient. The influence of physical watershed characteristics such as size of the watershed, extent of wetlands, and bedrock type is likely to play a secondary role for most water quality parameters, although wetland cover may show a stronger relationship to processes related to nitrogen uptake and transformation.

## Methods

### Study Area

Georgian Bay is a naturally oligotrophic waterbody on the eastern arm of Lake Huron (Fig. 1). The eastern and northern shores of the Bay as far south as the Severn River are underlain by Precambrian Shield rock, covered by approximately 30 cm of sandy, infertile soil (Weiler 1988) (Fig. 2); 21 of the 28 watersheds in our study are located within this landscape. The remainder of the watersheds examined in this study were located on the Cambrian and Ordovician rocks of the southern portion of the eastern shoreline, the tip of the Bruce Peninsula and Manitoulin Island; there, rocks are covered by about 30 cm of more fertile clay, silt, and sandy loam soils. The Shield rock is associated with ground water of low alkalinity, while

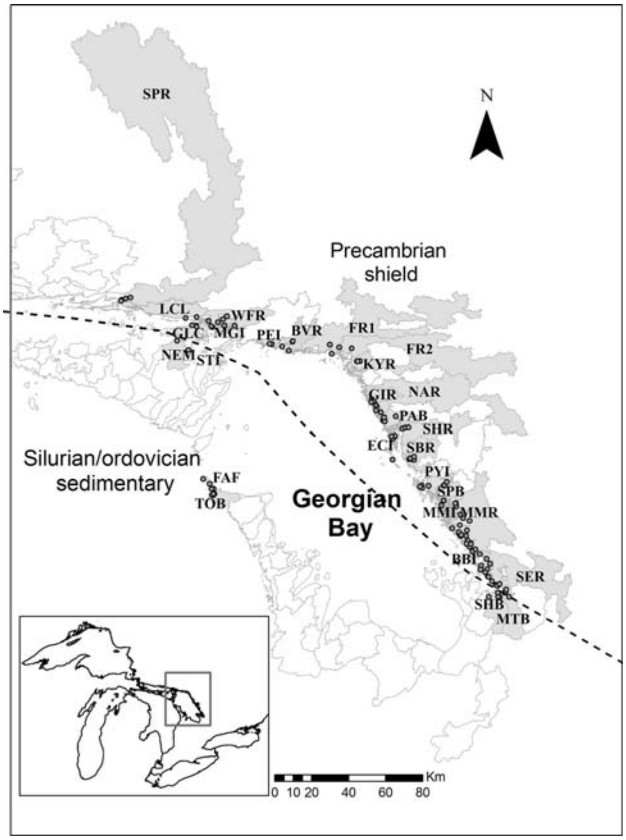


**Fig. 1** Map of water quality testing sites at outflows of quaternary watersheds in Georgian Bay. The dashed line separates watersheds underlain by Precambrian Shield bedrock and sedimentary bedrock

**Fig. 2** Photo of a typical Precambrian Shield landscape that characterizes much of eastern and northern Georgian Bay. Thin soils and vegetation overlie hard, metamorphic rock

Ordovician rock is associated with hard, highly mineralized groundwater (Weiler 1988).

Historically, natural geological and hydrological factors in the drainage basin and water exchange with Lake Huron and the North Channel have been dominant controls over water chemistry in the Bay. Atmospheric inputs are a significant source of nitrogen, phosphorus, and chloride (Weiler 1988). There are extensive areas of wetland extending back from the shore, which have been largely influenced by beaver (*Castor canadensis*) activity. While the age of the beaver ponds is difficult to estimate without a complete inventory of historical aerial photos, these systems are known to be both temporally and spatially dynamic, and can strongly influence hydrology and runoff chemistry (Johnston and Naimen 1990; Collen and Gibson 2001). In addition to beaver ponds, several other types of wetlands exist in the watersheds, including conifer swamps, bogs, and fens.

Georgian Bay has maintained a low human population density relative to other parts of the Great Lakes, but the presence of humans in the area has introduced a new watershed dynamic. Beginning in the late 1800s, resorts and cottages were being developed on the shores, along with septic beds and marinas (Sly and Munawar 1988). Until the early part of the 1900s, deforestation occurred along much of the coast to feed a thriving lumber industry. As a result, much of the forest that now surrounds the shores of Georgian Bay is second-growth.

Today, timber extraction occurs at a slower pace in areas where soils are most productive (Sly and Munawar 1988). Within our study area, the southeast portion of the Bay (south of Severn River) has been most heavily populated with seasonal cottagers and year-round residents (>20 persons/km$^2$ in the Township of Severn). There, agriculture creates additional impacts, particularly in the Sturgeon Bay-Hog Bay and Matchedash Bay watersheds (Sherman 2002). With the exception of Manitoulin Island, other portions of the bay examined in this study have had minimal agricultural impacts. In the late 1980s, eastern hardwood forests still covered about 95% the land area north of the Severn River, but only about 63% south of it (Weiler 1988). The southeast corner of the bay also shows more signs of water quality degradation than any other section of the east or north coasts (Cvetkovic 2008).

## Watershed Data

We obtained the most recent version of road network and wetland shapefiles from the National Topographic Database (NTDB; Natural Resources Canada) and imported them into ArcMap 9.0 (ESRI, Redlands, CA, USA). These data were produced at the 1:50,000 scale from scanned topographic maps that originated in the late 1980s and were updated as data became available. The files have been recently (post 2000) planimetrically enhanced with Landsat 7 orthoimages. Road networks were obtained in two separate files. The first contained non-limited-access (non-limited-use) roads; in the Georgian Bay area, these included a sparse network of highways and primary (numbered) roads and a more widespread network of secondary local or rural roads and streets. The second file contained limited-access (limited-use) roads, which in the study area consisted of car tracks and dry-weather use roads; these roads generally extended out a short distance from non-limited-access roads. The wetland and road network shapefiles were overlaid onto quaternary watersheds draining into Georgian Bay that were obtained from the Ontario Ministry of Natural Resources (OMNR). Names were assigned to the watersheds according to major rivers, bays, islands, or other dominant features (modified from Cvetkovic 2008). NTDB files were clipped for each watershed. Wetland area was expressed as the proportion of the quaternary watershed that it occupied (PROPWET), while road density was calculated as length per unit area for the entire watershed.

We used only the 28 quaternary watersheds for which water quality data were available (Fig. 1). This included all but four watersheds located along a continuous stretch of the east and north coasts of Georgian Bay and eastern North Channel, as well as two watersheds in the Tobermory area on the northwest coast of the Bay at the tip of the Bruce Peninsula (Fig. 1). Two of the watersheds from the eastern coastline of Georgian Bay for which no water quality data were collected are located near the town of Parry Sound. This area was not sampled during surveys because it contains relatively few coastal wetlands compared to the rest of the eastern Georgian Bay coast. The other two watersheds for which no data were available are located in the more northern portion of the Bay, where accessibility poses an additional constraint on sampling efforts. In watersheds for which water quality was assessed, the number of sampling stations at marshes along the stretch of watershed outflow ranged from 1 to 22.

### Water Quality Sampling

Water quality data for this study were obtained over a 9-year period of synoptic sampling along eastern and northern Georgian Bay. During this time, 105 coastal marshes (126 site-years) were sampled. Most of these data have been used in previously published studies (e.g., Chow-Fraser 2006; Cvetkovic 2008). Although sampling each year took place between late May and early September, wetlands were sampled primarily in June, July, and August. This sampling period excluded any influence of snowmelt, which is known to elevate concentrations of nutrients and dissolved minerals in coastal zones each spring (Weiler 1988). The time of day at which sampling occurred varied among sites but fell between 0900 and 2000 h.

Field sampling and analytical procedures for determining specific conductivity (COND), pH, temperature (TEMP), turbidity (TURB), total phosphorus (TP), soluble reactive phosphorus (SRP), total nitrogen (TN), total ammonium nitrogen (TAN), total nitrate nitrogen (TNN), total suspended solids (TSS), and inorganic suspended solids (ISS) followed procedures outlined in detail by Chow-Fraser (2006); for TN analysis, however, we used Hach Test 'N Tube reagents and protocols beginning in 2005. The samples were processed on the day of collection for TAN and TNN. At that time, water was filtered for suspended solids (TSS and ISS) and chlorophyll (CHL) *a*, and filtered water was frozen for SRP analysis. Raw water samples were frozen and brought to the lab for TN and TP analyses. All frozen samples were processed in the lab within roughly 4 months of collection.

### Use of the Water Quality Index

We have chosen to use the Water Quality Index (WQI) developed by Chow-Fraser (2006) to provide an overall measure of the condition of Georgian Bay coastal marshes with which to compare across watersheds. Unlike other water quality indexes available in Canada, many of which were designed for use on inland lakes and rivers (e.g., Rocchini and Swain 1995; CCME 2001), the WQI used here is specific to assessing Great Lakes coastal marshes and was developed using water quality data from 110 marshes spanning all five Great Lakes, including 18 in Georgian Bay.

The index was derived using principal components analysis of 12 water quality variables (TSS, TURB, ISS, TP, SRP, TAN, TNN, TN, COND, TEMP, pH, CHL) that were log10-transformed (Chow-Fraser 2006). The WQI score for each site was formed by the weighted sum of all 12 principal component site scores. A predictive equation that can be used to generate WQI scores from a given dataset containing the required water quality parameters was then created using stepwise multiple regression to give

$$\begin{aligned}\mathrm{WQI} = 10.0239684 &- (0.3154965 * \log\mathrm{TURB})\\ &- (0.3656606 * \log\mathrm{TSS}) - (0.3554498 * \log\mathrm{ISS})\\ &- (0.3760789 * \log\mathrm{TP}) - (0.1876029 * \log\mathrm{SRP})\\ &- (0.0732574 * \log\mathrm{TAN}) - (0.2016657 * \log\mathrm{TNN})\\ &- (0.2276255 * \log\mathrm{TN}) - (0.5711395 * \log\mathrm{COND})\\ &- (1.1659027 * \log\mathrm{TEMP}) - (4.3562126 * \log\mathrm{pH})\\ &- (0.2287166 * \log\mathrm{CHL})\end{aligned}$$

The WQI has been found to be significantly negatively correlated with the proportion of altered land in the associated watershed, assessed as the sum of urban and agricultural landuse types, and can be used to indicate the degree of anthropogenic impairment of coastal marsh water quality (Chow-Fraser 2006).

### Data Analysis

Because we were interested in a landscape approach to water quality modeling, we chose to focus on water quality variables that can be linked directly to watershed processes, including nutrients (TP, SRP, TN, TAN, TNN), COND, TSS, and ISS. These variables, along with TURB, TEMP, pH, and CHL *a*, were combined to give a WQI score for each wetland. Means of water quality variables were computed for each watershed. Variables that were noticeably right-skewed were log or square root (sqrt) transformed prior to further analysis to approach normality.

Statistical analyses were performed in SAS JMP 4.0. We used a significance level of 0.05 for all statistical tests. ANOVAs were used to look for potential effects of bedrock type on landscape and water quality variables. Two classes were used: (1) sedimentary rock (Silurian/Ordovician) and (2) Precambrian Shield rock (Southern/Grenville provinces). A Pearson's correlation matrix was used to explore relationships among water chemistry and landscape variables, and to examine the degree of colinearity among the landscape variables. The density of limited-access roads (RDLA) was initially included; however, because it showed only weak relationships to some water quality variables, we opted to focus on non-limited-access road density (RDNL) rather than limited-access or total road density in further analyses. We also believe the density of non-limited-access roads to be a superior indicator of the degree of human activity in the watershed, since these roads are easily accessed year-round and provide the primary access routes into most areas, whereas most limited-access roads extend only short distances out from non-limited-access roads.

We used a forward stepwise multiple regression procedure to derive explanatory equations and to determine the relative power of each watershed variable for explaining variation in water quality parameters. Bedrock type was included as a dummy variable along with continuous watershed variables. Only watershed variables with significant effects were retained in the models. When forming models, we also tested for significance of interactive effects of two variables. Partial correlation plots were used to illustrate relationships between individual watershed variables and water quality variables when more than one watershed variable had a significant effect in the multiple regression models (Draper and Smith 1981; Findlay and Houlahan 1997; Houlahan and Findlay 2004). Partial correlation statistically controls both the independent and the dependent variables for the effects of another variable, thereby removing spurious correlations and unmasking relationships hidden by the effects of other variables.

## Results

### Watershed Landscape Characteristics

Landscape variables were tabulated at the level of quaternary watersheds for eastern and northern Georgian Bay (Table 1). Watersheds ranged in size from 1372 to 555,653 ha, with a mean area of 46,377 ha. The largest was Spanish River watershed, located at the east end of the North Channel; it was more than four times the size of the next largest watershed. Several of the watersheds consisted of a group of islands or island and mainland contributors rather than a single landmass. Many island watersheds were only accessible by boat and contained no roads. The highest density of non-limited-access roads was 16.13 m ha$^{-1}$ in the Matchedash Bay watershed in the most southerly portion of eastern Georgian Bay. The adjacent Sturgeon Bay-Hog Bay watershed also had an exceptionally high density, at 14.00 m ha$^{-1}$. Great La Cloche Island had the highest density of limited-access roads, at 6.71 m ha$^{-1}$, despite having a relatively low density of non-limited-access roads. Wetlands constituted between 0.16% (in the Beausoleil-Bone watershed) and 23.29% (in the Shebeshekong River watershed) of the land area, with an average coverage of 7.01%.

There was minimal colinearity observed among landscape variables (Table 2). The density of non-limited-access roads was not significantly correlated with proportion wetland or log-transformed watershed area, though it was noted that the watersheds with very high road densities had low values for proportion of wetlands in the watersheds. There was also no significant relationship between watershed size and proportion wetland. With respect to bedrock class, we found that the density of non-limited-access roads was higher ($r^2 = 0.284$, $P = 0.0035$) and the proportion wetland lower ($r^2 = 0.151$, $P = 0.0407$) in sedimentary watersheds than in watersheds located on the Precambrian Shield.

### Water Quality

Water quality characteristics varied considerably among the 28 watersheds (Table 3). Mean WQI score was lowest in the Matchedash Bay watershed (–0.059), indicating moderately degraded water quality conditions. This watershed also had the highest levels of TP (43.06 μg L$^{-1}$), TNN (0.427 mg L$^{-1}$), COND (331 μS cm$^{-1}$), and TSS (13.29 mg L$^{-1}$). Mean WQI score for all other watersheds was >0.000, indicating that these watersheds are among the least human-disturbed in the context of the entire Great Lakes Basin (Chow-Fraser 2006). Mean WQI score was highest in the Shawanaga River watershed (2.150); this watershed also had the lowest COND measurement

**Table 1** List of the 28 quaternary OMNR watersheds, along with their assigned names and codes, and a summary of landscape characteristics for each, including watershed area, proportion of the watershed occupied by wetland (PROPWET), nonlimited access road density (RDNL), limited access road density (RDLA), and bedrock type (classified as Precambrian shield [PS] or sedimentary [SED])

| OMNR code | Assigned name | Assigned code | Area (ha) | PROPWET | RDNL (m $ha^{-1}$) | RDLA (m $ha^{-1}$) | Bedrock type |
|---|---|---|---|---|---|---|---|
| 2CE-01 | La Cloche | LCL | 27,269 | 0.0321 | 1.973 | 1.918 | PS |
| 2CE-02 | Spanish River | SPR | 555,653 | 0.0412 | 1.367 | 1.166 | PS |
| 2CF-02 | Philip Edward Island | PEI | 4,909 | 0.0868 | – | – | PS |
| 2CF-18 | McGregor Islands | MGI | 2,179 | 0.0699 | – | – | PS |
| 2CG-06 | Great La Cloche Island | GLC | 9,643 | 0.0132 | 1.402 | 6.711 | SED |
| 2CG-07 | Northeast Manitoulin | NEM | 13,847 | 0.0520 | 7.418 | 3.618 | SED |
| 2CG-33 | Strawberry Island | STI | 1,625 | 0.0486 | – | – | SED |
| 2CH-01 | Beaverstone River | BVR | 12,957 | 0.1081 | 0.061 | 0.033 | PS |
| 2CH-04 | Whitefish River | WFR | 26,527 | 0.0408 | 1.582 | 0.864 | PS |
| 2DD-01 | French River 1 | FR1 | 126,103 | 0.0588 | 2.170 | 1.315 | PS |
| 2DD-03 | French River 2 | FR2 | 105,176 | 0.0533 | 0.937 | 1.746 | PS |
| 2EA-01 | Key River | KYR | 19,669 | 0.1071 | 0.335 | 1.387 | PS |
| 2EA-04 | Giroux River | GIR | 10,949 | 0.2246 | 1.226 | 0.143 | PS |
| 2EA-05 | Pointe au Baril | PAB | 11,554 | 0.1263 | 2.555 | 0.732 | PS |
| 2EA-06 | Shebeshekong River | SBR | 19,720 | 0.2329 | 5.167 | 0.809 | PS |
| 2EA-07 | Parry Island | PYI | 7,666 | 0.1129 | 4.476 | 1.161 | PS |
| 2EA-08 | Spider Bay | SPB | 8,816 | 0.0523 | 0.600 | 0.123 | PS |
| 2EA-10 | Naiscoot River | NAR | 92,622 | 0.0791 | 2.388 | 1.275 | PS |
| 2EA-13 | Shawanaga River | SHR | 30,979 | 0.1035 | 2.333 | 1.019 | PS |
| 2EA-24 | East Coast Islands | ECI | 11,151 | 0.0332 | – | – | PS |
| 2 EB-01 | Moon-Musquash Islands | MMI | 4,392 | 0.0102 | – | – | PS |
| 2 EB-02 | Moon River-Musquash River | MMR | 71,731 | 0.0660 | 3.075 | 1.490 | PS |
| 2EC-17 | Severn River | SER | 70,445 | 0.0902 | 6.221 | 3.713 | PS |
| 2EC-18 | Beausoleil-Bone Islands | BBI | 1,683 | 0.0016 | 0.027 | 3.112 | PS |
| 2ED-04 | Sturgeon Bay-Hog Bay | SHB | 18,887 | 0.0438 | 14.002 | 1.098 | SED |
| 2ED-05 | Matchedash Bay | MTB | 21,727 | 0.0258 | 16.134 | 1.340 | SED |
| 2FA-05 | Tobermory | TOB | 9,307 | 0.0295 | 8.042 | 3.707 | SED |
| 2FA-13 | Fathom Five | FAF | 1,372 | 0.0183 | – | – | SED |

(26 $\mu$S $cm^{-1}$). The Fathom Five, Giroux River, Great La Cloche Island, and Moon River-Musquash River watersheds also contained marshes with exceptionally good water quality.

Several of the water quality variables were significantly correlated with each other (Table 2). WQI score showed highly significant negative correlations with log TP, log COND, log TN, sqrt TNN, log TAN, log TSS, and sqrt ISS. We found log TP to be positively correlated with log SRP, log TAN, log TSS, and sqrt ISS, while sqrt TNN was positively correlated with log COND. There was also a strong positive relationship between sqrt ISS and log TSS.

## Relating Water Quality to Watershed Variables

COND was the only water quality variable for which bedrock type was a significant ANOVA predictor; log COND levels were higher in watersheds with sedimentary bedrock than in watersheds underlain by Precambrian Shield rock ($r^2 = 0.196$, $P = 0.0182$). The initial bivariate analyses involving continuous landscape variables (Table 2) revealed a highly significant negative relationship between the density of non-limited-access roads and WQI scores and a significant positive relationship between road density and all other water quality variables except log TN, log TAN, and log SRP. Watershed area (log transformed) showed a positive relationship with log TP but was not significantly related to other water quality variables. Sqrt TNN and log COND both had a significant negative relationship to proportion wetland. All of these relationships remained significant when partial correlations were used to control for the confounding effects of other watershed variables identified in the stepwise regression procedure (Table 4, Fig. 3). Correlation of proportion wetland with both log TSS and sqrt ISS also became significant once we controlled for road density (Fig. 3).

**Table 2** Correlation matrix for water quality and watershed variables

| | PROPWET | log area | RDLA | RDNL | WQI | log TP | log SRP | log TN | sqrt TNN | log TAN | log COND | log TSS | sqrt ISS |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| PROPWET | 1.00 | | | | | | | | | | | | |
| log area | 0.11 | 1.00 | | | | | | | | | | | |
| RDLA | –0.28 | 0.16 | 1.00 | | | | | | | | | | |
| RDNL | –0.04 | 0.20 | 0.26 | 1.00 | | | | | | | | | |
| WQI | 0.02 | –0.34 | –0.11 | **–0.76** | 1.00 | | | | | | | | |
| log TP | 0.01 | **0.57** | 0.21 | **0.49** | **–0.60** | 1.00 | | | | | | | |
| log SRP | –0.32 | 0.32 | 0.21 | 0.08 | –0.16 | **0.45** | 1.00 | | | | | | |
| log TN | –0.23 | –0.01 | –0.08 | 0.22 | **–0.48** | 0.07 | 0.14 | 1.00 | | | | | |
| sqrt TNN | **–0.45** | 0.06 | 0.12 | **0.38** | **–0.39** | 0.06 | 0.14 | 0.06 | 1.00 | | | | |
| log TAN | –0.18 | 0.17 | 0.31 | 0.34 | **–0.57** | **0.44** | 0.37 | 0.33 | 0.32 | 1.00 | | | |
| log COND | **–0.42** | –0.05 | 0.13 | **0.56** | **–0.66** | 0.11 | 0.17 | 0.29 | **0.59** | 0.37 | 1.00 | | |
| log TSS | 0.32 | 0.25 | –0.01 | **0.51** | **–0.67** | **0.46** | –0.30 | 0.01 | 0.11 | 0.22 | 0.30 | 1.00 | |
| sqrt ISS | 0.37 | 0.09 | 0.02 | **0.45** | **–0.60** | **0.46** | –0.30 | 0.12 | –0.07 | 0.29 | 0.19 | **0.81** | 1.00 |

*Note*: $P < 0.05$ when $r \geq 0.38$ (in boldface)

**Table 3** Summary of water quality data from the 28 quaternary watersheds

| Code | *N* | WQI score | TP ($\mu g\ L^{-1}$) | SRP ($\mu g\ L^{-1}$) | TN ($mg\ L^{-1}$) | TNN ($mg\ L^{-1}$) | TAN ($mg\ L^{-1}$) | COND ($\mu S\ cm^{-1}$) | TSS ($mg\ L^{-1}$) | ISS ($mg\ L^{-1}$) |
|---|---|---|---|---|---|---|---|---|---|---|
| LCL | 2 | 1.522 | 21.36 | 10.76 | 0.950 | 0.205 | 0.015 | 83 | 0.92 | 0.49 |
| SPR | 4 | 0.621 | 27.80 | 5.80 | 1.334 | 0.317 | 0.056 | 123 | 4.78 | 3.10 |
| PEI | 3 | 1.420 | 11.15 | 2.45 | 1.500 | 0.038 | 0.012 | 123 | 4.26 | 2.32 |
| MGI | 2 | 1.555 | 12.54 | 5.44 | 0.520 | 0.420 | 0.030 | 102 | 0.92 | 0.01 |
| GLC | 3 | 1.812 | 16.03 | 6.11 | 0.387 | 0.137 | 0.017 | 100 | 1.13 | 0.69 |
| NEM | 1 | 0.848 | 18.81 | 2.37 | 0.834 | 0.327 | 0.037 | 101 | 7.78 | 3.55 |
| STI | 1 | 1.778 | 9.22 | 2.37 | 2.400 | 0.060 | 0.001 | 67 | 1.21 | 0.54 |
| BVR | 3 | 1.254 | 12.91 | 3.87 | 5.467 | 0.040 | 0.011 | 64 | 2.32 | 1.62 |
| WFR | 6 | 1.538 | 17.58 | 9.68 | 0.882 | 0.307 | 0.039 | 104 | 1.00 | 0.20 |
| FR1 | 2 | 1.577 | 28.61 | 6.30 | 0.450 | 0.075 | 0.001 | 74 | 1.77 | 0.05 |
| FR2 | 2 | 1.377 | 29.30 | 7.27 | 0.400 | 0.020 | 0.030 | 82 | 1.85 | 0.79 |
| KYR | 3 | 1.477 | 30.94 | 7.33 | 0.181 | 0.118 | 0.004 | 54 | 6.25 | 4.39 |
| GIR | 4 | 1.857 | 9.91 | 3.70 | 0.100 | 0.031 | 0.003 | 51 | 2.43 | 0.97 |
| PAB | 5 | 1.351 | 22.84 | 4.60 | 0.444 | 0.088 | 0.018 | 101 | 4.18 | 3.40 |
| SBR | 3 | 0.939 | 23.83 | 1.59 | 0.334 | 0.090 | 0.011 | 94 | 11.88 | 7.81 |
| PYI | 4 | 1.195 | 26.70 | 5.70 | 2.375 | 0.084 | 0.036 | 53 | 3.21 | 3.78 |
| SPB | 3 | 1.658 | 9.43 | 3.36 | 0.333 | 0.233 | 0.001 | 133 | 3.04 | 0.62 |
| NAR | 2 | 1.361 | 20.34 | 6.91 | 0.520 | 0.320 | 0.010 | 81 | 3.80 | 0.01 |
| SHR | 3 | 2.150 | 23.56 | 2.55 | 0.162 | 0.043 | 0.004 | 26 | 2.20 | 1.76 |
| ECI | 12 | 1.531 | 14.26 | 4.48 | 0.367 | 0.169 | 0.014 | 121 | 3.47 | 2.50 |
| MMI | 5 | 1.627 | 19.45 | 3.40 | 0.377 | 0.243 | 0.015 | 87 | 4.23 | 1.01 |
| MMR | 22 | 1.805 | 11.93 | 2.65 | 0.431 | 0.184 | 0.006 | 65 | 2.84 | 0.71 |
| SER | 13 | 0.991 | 22.37 | 5.88 | 0.893 | 0.270 | 0.022 | 160 | 5.78 | 2.48 |
| BBI | 3 | 1.620 | 16.87 | 6.76 | 0.480 | 0.063 | 0.028 | 105 | 2.46 | 0.71 |
| SHB | 3 | 0.651 | 26.29 | 11.07 | 1.460 | 0.210 | 0.027 | 216 | 3.51 | 1.71 |
| MTB | 3 | –0.059 | 43.06 | 4.62 | 1.377 | 0.427 | 0.037 | 331 | 13.29 | 6.74 |
| TOB | 5 | 1.277 | 12.95 | 3.00 | 0.350 | 0.312 | 0.012 | 129 | 2.56 | 1.11 |
| FAF | 4 | 1.937 | 9.34 | 2.69 | 0.350 | 0.337 | 0.003 | 138 | 1.34 | 0.70 |

*Note*: See text for explanation of abbreviations of water-quality variables

**Table 4** Watershed parameters retained in stepwise multiple regression models to predict water quality variables

| WQ variable | Watershed variable | $r^2$ | $P$ | Regression coefficient (±SE) |
|---|---|---|---|---|
| log TP | log area | 0.33 | 0.0027 | +0.1462 (±0.0438) |
| | RDNL | 0.15 | 0.0136 | +0.0401 (±0.0151) |
| sqrt TNN | PROPWET | 0.20 | 0.0132 | –1.2188 (±0.4569) |
| | RDNL | 0.13 | 0.0358 | +0.0138 (±0.0062) |
| log COND | RDNL | 0.31 | 0.0010 | +0.0639 (±0.0171) |
| | PROPWET | 0.16 | 0.0122 | –3.4028 (±1.2589) |
| log TSS | RDNL | 0.26 | 0.0028 | +0.0912 (±0.0275) |
| | PROPWET | 0.11 | 0.0432 | +4.3085 (±2.0225) |
| sqrt ISS | RDNL | 0.21 | 0.0072 | +0.0783 (±0.0267) |
| | PROPWET | 0.15 | 0.0219 | +4.8075 (±1.9660) |
| WQI | RDNL | 0.58 | <0.0001 | –0.0864 (±0.0144) |

*Note*: Watershed variables are listed in the order they were entered into the model

Explanatory Models

We present explanatory models built from forward stepwise multiple regressions for all water quality variables that showed a significant relationship with at least one watershed variable (Tables 4, 5). The density of non-limited-access roads was the only watershed variable to have a significant effect on WQI, alone explaining 58.2% of the variation (Fig. 4). Road density also contributed significantly to all other models, explaining the largest percentage of variation in log COND (31.4%), log TSS (26.1%), and sqrt ISS (20.6%) of all watershed variables. Watershed area (log transformed) contributed strongly to the model for log TP, accounting for 32.6% of the variation, with road density explaining an additional 14.8%. Proportion wetland contributed significantly to models for log COND, sqrt TNN, log TSS, and sqrt ISS, accounting for 15.5%, 20.3%, 11.3%, and 15.3% of the variation, respectively. Bedrock class did not have a significant effect in any of the models. Accounting for interactive effects between two watershed variables never resulted in a significant improvement to the models and interaction terms were therefore omitted.

## Discussion

Road density showed the strongest relationship to water-quality conditions in Georgian Bay coastal wetlands. It accounted for a large amount of variation in several water-quality parameters, including COND, TSS, ISS, and overall WQI score, and was a secondary factor influencing TP and TNN levels. Increased nutrients and suspended sediments have been observed across increasing anthropogenic stress gradients at the scale of the entire Great Lakes basin, including one stress gradient that was partially based on and highly correlated with road density (Chow-Fraser 2006; Danz and others 2007). The positive relationships between road density and phosphorus and nitrates have also been reported for wetlands in southeastern Ontario (Houlahan and Findlay 2004). Phosphorus is naturally limiting in coastal zones of Georgian Bay (Weiler 1988), and the anthropogenic contribution to loadings of P and other nutrients can therefore potentially have large impacts on the productivity of these systems. The degree to which road density changed over the 9-year period of our study and the extent to which this may have influenced our results are unknown. However, despite this data limitation, our findings strongly support the idea that expansion of road networks has the ability to impair coastal wetland water quality through increasing nutrient and sediment loadings from the watershed. Furthermore, this effect often overpowers the influence of natural variation in physical watershed characteristics.

The relationship with the largest percentage of variation explained (both with and without controlling for confounding variables) was that between road density and WQI score. This reflects the strong ability of this index to detect anthropogenic degradation of coastal marsh water quality even within the lower end of the disturbance gradient, and suggests that it is an appropriate measure for assessing human-induced degradation of coastal marshes. Based on the regression model, an increase in road density of just 11.6 m ha$^{-1}$ would be expected to decrease WQI scores by 1.00 unit, representing a considerable decline in water quality. This water-quality impairment in turn has the potential to alter trophic dynamics and cause a shift toward more degradation-tolerant flora and fauna (Lougheed and others 2001; McNair and Chow-Fraser 2003; Seilheimer and Chow-Fraser 2006; Danz and others 2007).

Watershed area explained the greatest percentage of variation in TP levels in coastal wetlands within Georgian Bay. Johnston and others (1990) also found that watershed

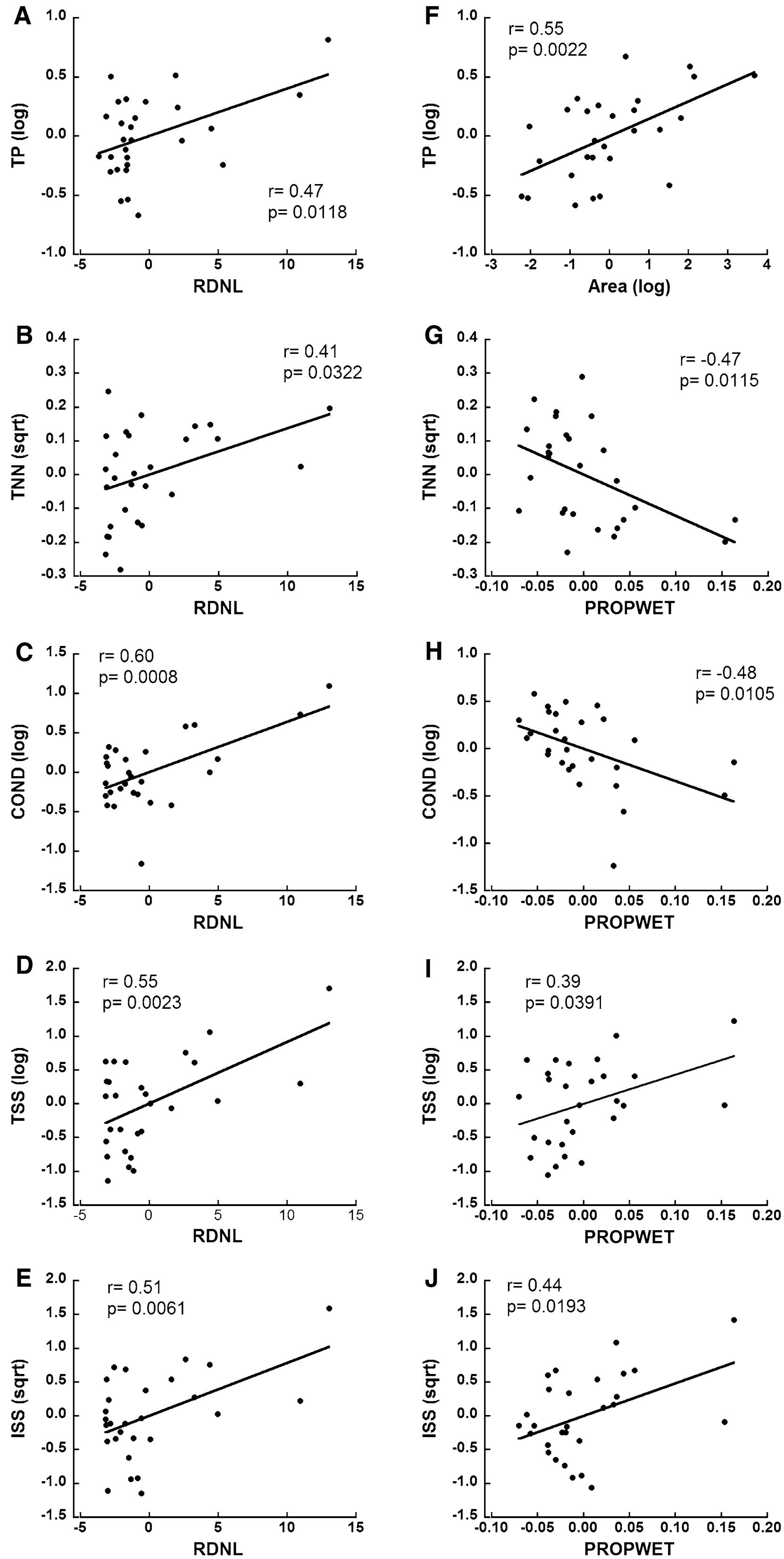


**Fig. 3** Partial plots of watershed variables by water quality variables, controlling for other variables in their respective multiple regression models: **a** log area, **b–e** PROPWET, and **f–j** RDNL

**Table 5** Overall equations modeling relationships between water quality variables and watershed variables, generated using stepwise multiple regressions

| WQ variable | $r^2$ | $r^2_{adj}$ | $p$ | Equation |
|---|---|---|---|---|
| log TP | 0.47 | 0.43 | 0.0003 | +1.3604 + 0.1462(log area) + 0.0401(RDNL) |
| sqrt TNN | 0.33 | 0.28 | 0.0062 | +0.4455 – 1.2188(PROPWET) + 0.0138(RDNL) |
| log COND | 0.47 | 0.43 | 0.0004 | +4.5978 + 0.0639(RDNL) – 3.4028(PROPWET) |
| log TSS | 0.37 | 0.32 | 0.0028 | +0.4883 + 0.0912(RDNL) + 4.3085(PROPWET) |
| sqrt ISS | 0.36 | 0.31 | 0.0038 | +0.6408 + 0.0783(RDNL) + 4.8075(PROPWET) |
| WQI | 0.58 | 0.57 | $<0.0001$ | +1.6388 – 0.0864(RDNL) |

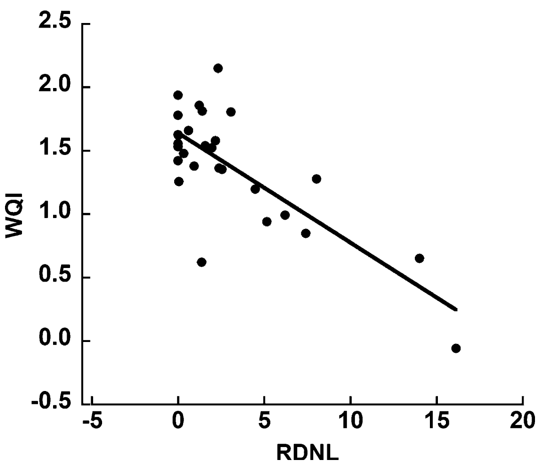


**Fig. 4** Regression of WQI on road density (non-limited-access roads). Road density alone explained 58.2% of variation in WQI scores among quaternary watersheds

area influenced concentrations of P in relatively undisturbed watersheds of Minnesota. In these areas, thin soils and impermeable bedrock have the tendency to direct runoff through organically rich surface layers with high levels of biologically available soil nutrients (Schiff and others 1998). Larger watersheds provide a greater land surface over which hydrologic flushing of nutrients can occur, leading to higher TP concentrations in outflow. While not evident in our study, watershed area can also be positively related to concentrations of other nutrients, including several forms of nitrogen (Johnston and others 1990).

Bedrock type can be an important determinant of water chemistry. Weathering of carbonate minerals occurs at a much faster rate in sedimentary bedrock than in the metamorphic bedrock found in the Precambrian Shield and is a major source of dissolved ions such as calcium and bicarbonate (Gorham and others 1983; Weiler 1988; Keough and others 1999); therefore, we expected to see naturally higher specific conductivity levels in marshes at the base of sedimentary watersheds. Likewise, weathering of sedimentary bedrock is known to promote increased nutrient enrichment of lakewaters (Conroy and Keller 1976). Our results did not identify bedrock type as a significant factor governing COND or other water quality variables once the effects of other watershed features were accounted for. Specific conductivity is also known to be a reliable indicator of anthropogenic impact on freshwater systems (Lott and others 1994; Chow-Fraser 2006), and our data suggest that it is variation in road density, rather than bedrock type, that has the dominant influence on specific conductivity values in Georgian Bay.

Understanding the relationship between wetland cover in the watershed and coastal marsh water quality is important not only for the purpose of predicting natural variation in water quality, but also for understanding the implications of wetland loss that often occurs as a result of human development (Wolter and others 2006). Like Johnston and others (1990), we found wetland cover to be a significant factor determining COND levels. Wetlands have the ability to filter dissolved ions and nutrients in surface runoff (Hemond and Benoit 1988; Johnston and others 1990) and can therefore help reduce ionic concentrations. As expected, we also found that greater wetland cover is related to lower levels of TNN in marshes at the watershed outflow. This is consistent with a large body of literature that outlines the importance of wetlands in the nitrogen cycle. Wetlands produce anoxic conditions which promote denitrification and permanent removal of nitrates from runoff (e.g., Whigham and others 1988; Seitzinger 1994). While other studies have confirmed that Precambrian Shield wetlands are important in the retention of nitrates (Devito and others 1989; Devito and Dillon 1993), other factors could have influenced the relationship observed in this study. Road density was the sole measure of human development examined, and it is possible that within Georgian Bay an inverse relationship exists between wetland cover and other measures of development such as the proportion of human-altered land; if this were the case, an increase in TNN loadings as a result of human activities could result in a negative relationship between wetland cover and TNN.

Previous studies have found conflicting results regarding the relationship between wetland cover and TP export; some researchers have shown a negative relationship

(Johnston and others 1990; Houlahan and Findlay 2004), while others have found a positive relationship between the two variables (Dillon and Molot 1997; Devito and others 2000). We did not find wetland cover to be a significant factor affecting TP in Georgian Bay coastal marshes. This may be due to overriding factors not accounted for in our study, including the hydrological setting of wetlands within the landscape; wetlands connected directly to each other and to the lake tend to export P and other nutrients more readily than those that are separated by a large upland area (D'Arcy and Carignan 1997; Devito and others 2000). Additionally, wetland type can play a large role in determining whether they behave as sources or sinks of P (Devito and others 1989); our inability to distinguish among wetland types in the database means that we are uncertain as to whether they differed among watersheds or different bedrock types. Our finding of higher TSS and ISS levels in marshes with a larger proportion of the watershed covered by wetlands is somewhat consistent with the findings of Johnston and others (1990); they found that while wetlands tend to retain ISS, wetland extent was positively related to suspended solids export when wetlands drained directly into streams. This suggests that where wetlands are hydrologically well connected to the lake, as is often observed in beaver-influenced nearshore areas of Georgian Bay, the drainage of wetlands can lead to higher concentrations of suspended solids in coastal zones.

The seasonality of our sampling likely influenced the observed associations between wetland cover and water quality dynamics. Johnston and others (1990) found that wetlands were more effective at removing suspended solids, phosphorus, and ammonia during high-flow periods and nitrates during low-flow periods. Our sampling was conducted only during late spring and summer months and, therefore, represents water quality conditions during low flow. Since export of sediment, nutrients, and dissolved ions tend to vary considerably with season due to changes in biotic uptake and the amount of precipitation and runoff (Devito and others 1989; Johnston and others 1990; Schiff and others 1998; Eimers and others 2008), sampling during peak spring runoff may lead to very different conclusions regarding the importance of wetlands in influencing water quality variation within Georgian Bay.

Our study was based on the premise that water quality conditions in coastal wetlands are influenced by characteristics of associated watersheds. We have shown this to be true for several water quality variables within Georgian Bay. Furthermore, we have shown that within this sparsely populated region, road density has a stronger influence on water quality than do physical watershed features such as size, wetland cover, and bedrock type. Due to the high potential for nutrient and sediment loading during spring snowmelt, future work should focus on quantifying watershed impacts during both low and high flow to capture variation in nutrient pulses. The insight provided by this study, however, gives a solid basis for further investigations into watershed-water quality interactions in Georgian Bay coastal marshes and other freshwater coastal systems. These investigations should lead to establishment of more appropriate site-specific water quality standards and permit proper assessment of anthropogenic contributions to corresponding changes in water quality.

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