Impacts of Golf Courses on Macroinvertebrate Community Structure in Precambrian Shield Streams

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ABSTRACT

The influence of golf course operation on benthic macroinvertebrate communities in Precambrian Shield streams was evaluated using rapid bioassessment and the reference condition approach. Streams were sampled for water chemistry and invertebrates in 1999 and 2000, six on operational golf courses, and seven in forested reference locations. Correspondence analysis (CA) was used to determine the major patterns in the macroinvertebrate taxa, and canonical correspondence analysis (CCA) was used to evaluate relationships with environmental variables. The reference streams were used to define the normal range of variation for a variety of summary indices to evaluate the golf course streams. In all cases, golf course streams were higher in nutrients and dissolved ions and more alkaline than the forested reference streams. There was considerable variability in the macroinvertebrate fauna from the golf course streams, which was related to differences in golf course land management practices and to the potential influence of highway runoff. Of the management practices evaluated, fertilizer application rates in particular were important, as was the presence of ponds upstream on the course. Invertebrate taxa with higher abundances in golf course streams included Turbellaria, Isopoda, Amphipoda, Zygoptera, and Trombidiformes. Taxa more common in the reference streams included Ephemeroptera, Megaloptera, Culicidae, and Plecoptera. There were marked differences in the overall benthic macroinvertebrate community in three of the six golf course streams studied relative to the forested reference streams, suggesting that golf course land management on the Precambrian Shield can be associated with significant differences in macroinvertebrate community structure.

CHANGES IN LAND use within stream catchments lead to fundamental changes in the structure and functioning of aquatic communities (Allan et al., 1997; Hynes, 1975). There have been widespread land use changes for agriculture in many watersheds in North America, and urbanization has led to intensive changes in localized areas (Allan, 1995). On the Precambrian Shield, however, agricultural and urban development is minimal. Here, modifications are relatively few and generally consist of transforming land for mining, forestry, and recreation. One recreational activity that is becoming increasingly popular is golf. Interest in this sport has risen dramatically in North America in recent years, and a continuation in this trend has been forecast (Foote, 1998). In Ontario, Canada, >30% of adult residents played at least one round of golf in 1997, up 25% from 1996. As a result of this trend, numerous new courses are being constructed throughout Ontario, including many on the Precambrian Shield.

Golf courses may have impacts on streams and lakes because they represent a major landscape change and are intensively managed. In many cases, chemical additions on golf courses are similar to, and often greater than, those used in intensive agriculture (Alberta Environmental Protection, 1998; Smith and Bridges, 1996). Numerous lakes and streams characterize the Precambrian Shield, and are generally very low in nutrients and dissolved ions (Dillon et al., 1991; Molot and Dillon, 1991). Presumably, golf course operation will impact these lakes and streams, particularly because courses are usually constructed close to waterbodies. However, the magnitude of this impact is not presently known because there are no published studies on the effects of golf course operation on these systems. To date, golf course studies have focused on regions south of the Shield where surface waters are strongly buffered and soils are much thicker. These studies have evaluated the movement of fertilizers and pesticides to groundwater and surface water (Cohen et al., 1999; Balogh and Anderson, 1992; Ryals et al., 1998; Lewis et al., 2001; Mallin and Wheeler, 2000; Walker and Branham, 1992) while less attention has been given to effects on aquatic biota.

Both water chemistry and biological assessments are used to evaluate impacts on aquatic ecosystems (Hynes, 1963). While observed chemical changes can provide direct evidence of water quality impacts, biological assessments are often used to indicate the relevance of water chemistry changes on the aquatic community (Rosenberg and Resh, 1993). Biota integrate environmental conditions, including short-term changes in water quality that may be missed by intermittent chemical sampling (Cox, 1991). If we wish to maintain healthy, diverse biological communities, it is appropriate to monitor the aquatic community in conjunction with physico-chemical variables. Macroinvertebrates are sensitive indicators of aquatic ecosystem health (Resh et al., 1996) and the invertebrate community can serve as sensitive early warning indicators of impacts that may take years to otherwise become apparent (Raddum and Fjellheim,

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Abbreviations: CA, correspondence analysis; CCA, canonical correspondence analysis; EPT, ephemeroptera, plecoptera, and trichoptera; HBI, Hilsenhoff biotic index.

Table 1. Land use management strategy and physical features of golf course sites.

Golf course site no.	Age	Nitrogen fertilizer application rate	Highway runoff	Pond upstream of site	Proportion of catchment forested	Buffer at site
	yr	kg ha^{-1} yr^{-1}			%	
1	5–10	100–150			~50	x
2	10–20	50–100		x	<10	x
3	10–20	50–100	x	x	<10	x
4	10–20	100–150	x	x	<10	x
5	20–50	<50	x		>90	
6	>50	50–100			~50	x

1984). Lewis et al. (2001) concluded that the impact of golf course runoff on sediment quality can be subtle and may require sensitive biological assessment methods. In addition, macroinvertebrates can be used as surrogate indicators for changes in habitat quality and the fish community.

We evaluated the influence of golf course operation on macroinvertebrate community structure in Precambrian Shield streams by comparing water quality and benthic communities between six streams adjacent to golf courses and seven reference streams in nearby forested catchments. We used a reference condition approach because sites upstream of the golf courses were generally unavailable. This approach involves comparing streams exposed to a potential stress to reference streams that are unexposed to that stress (Bailey et al., 1998; Reynoldson et al., 1997). We hypothesize that there will be measurable differences between reference and golf course streams if golf course use is impacting stream macroinvertebrate communities on the Shield.

METHODS

Study Location and Site Descriptions

The streams sampled were all located on the Precambrian Shield in the District of Muskoka, an area of approximately 6500 km^2 located about 200 km north of Toronto, ON, Canada. The Precambrian Shield covers nearly half of Canada, and is characterized by thin, acidic soils over granitic (silicate) bedrock, with numerous streams and lakes (Jeffries and Snyder, 1983). The dominant surficial geology is minor till plain (continuous moraine deposits >1 m thick) and thin till deposits (<1 m thick), interrupted by rock ridges. Dominant soils in the area are brunisolic or podzolic. Because soils/tills in Precambrian regions are generally very shallow, there are fewer opportunities for sorption and precipitation reactions than in other areas, and the soils may not retain total P as strongly (Dillon and Molot, 1996).

Six streams were sampled on five golf courses (two streams are located on one course) that have been in operation from 5 to >50 yr (Table 1). Seven streams were sampled in nearby reference locations; two of these are upstream of golf course sites. Land use and physical features were determined using maps, field observations, and via a chemical use and management survey of golf course superintendents. The reference catchments are located in forested, undeveloped areas. Beaver dams and small ponds are upstream of three of the reference sites. These were included in our reference data set to incorporate a range of background conditions typical of the Shield. One golf course and one reference stream are third order, one reference stream is second order, and the rest of the sampled streams are first order (based on 1:10 000 topographic maps). Catchment sizes range from approximately 0.08 to 10.25 km^2 for reference streams, and from 0.16 to 8.05 km^2 for golf course streams. Three of the six golf course streams drain golf course properties from source to mouth (i.e., the golf course covers almost 100% of the catchment), whereas the other streams have a substantial proportion of their catchment upstream of the golf courses (from 50 to 96%; Table 1). The golf courses also differed in their landscape management and design features, in particular fertilizer application rates, the presence of ponds upstream and mowing practices (e.g., the presence or absence of vegetated buffer strips along the stream; Tables 1 and 2). Streams 3 and 5 were also influenced by highway runoff. The upstream ponds drain into the streams where present. The two adjoining ponds on Stream 4 [1800 and 2000 m^2 surface area, both 3 m (10 ft) deep] were located directly upstream of the sampling area, whereas the ponds on Stream 2 [two ponds, one at the source of the stream approximately 9300 m^2 and another downstream 4600 m^2, both 2.4 m (8 ft) deep] and Stream 3 [one pond, ca. 4600 m^2 and 1.8–2.4 m (6–8 ft) deep] were further upstream of the sampling sites. Although the locations sampled on Streams 3 and 4 were surrounded by an unmanaged buffer, the stream banks were mowed in upstream areas. Raw numbers and not the summary values for age, N fertilizer application rate, and the proportion of catchment broadly classed as forested were used in the analysis with the environmental data.

Water Chemistry

Stream water samples were collected at the same time as the macroinvertebrates. Water samples were filtered through a 76-μm Nitex mesh into prerinsed bottles and placed in temperature-controlled containers while in transit to the laboratory. A series of water chemistry parameters was measured in the laboratory, using standard analytical methods that are outlined in Ontario Ministry of the Environment (1983). Gran alkalinity, pH, Ca and Cl concentrations, conductivity, dissolved inorganic carbon (DIC), dissolved organic carbon (DOC), K, Mg, Na and sulfate concentrations, ammonia, nitrate, total Kjeldahl nitrogen (TKN), and total P were tabulated to compare reference and golf course streams.

Macroinvertebrate Collection and Processing

Macroinvertebrate samples were collected in the fall of 1999 (mid-October to early November) and the spring (early April to mid-May), summer (late July to mid-August), and fall (early October to early November) of 2000. Several streams were not sampled in the summer due to insufficient flow. Samples were collected and processed according to Ontario Ministry of the Environment rapid bioassessment protocols (David et al., 1998). Three riffle areas were sampled at each stream location wherever possible. Runs (areas with flow rates intermediate between riffles and ponds) were sampled at two golf course and two reference streams where suitable riffles were unavailable. Substrates at each sampling area were characterized by determining the percentage cover of the following

Table 2. Composition (%) of vegetation in a 30 by 30 m buffer at the golf course stream sites. The remainder was made up of cart paths and cleared land.

Golf course site no.	Coniferous trees	Deciduous trees	Wetland	Grass (rough)	Fairway
1	34	2	4	17	36
2	0	16	14	57	3
3	1	38	33	10	15
4	0	1	31	16	47
5	0	2	0	0	98
6	34	4	0	22	40

categories (Gordon et al., 1992): (i) bedrock (sheets of rock); (ii) boulder (large rocks >25 cm); (iii) cobble (6–25 cm); (iv) gravel (2 mm–6 cm); (v) sand (0.1–2 mm); (vi) silt and clay (<0.1 mm); (vii) logs; and (viii) macrophytes.

Beginning at the riffle furthest downstream, an unobstructed 1-m^2 area was arbitrarily selected and invertebrates were collected with a kick-and-sweep technique using a 250-μm mesh D net for 1 min. The debris and associated invertebrates were placed in a labeled bucket and transported to the laboratory. Random subsamples of the material in each bucket were taken and sorted until at least 100 macroinvertebrates were removed and identified to order or family. Three counts of at least 100 invertebrates were thus obtained for each stream on each sampling occasion. A total of 78 reference and 69 golf course samples were collected during the study.

Data Analysis

Several biological indices were calculated using the raw macroinvertebrate data: (i) total number of taxonomic groups; (ii) a variant of the Hilsenhoff biotic index (HBI; Hilsenhoff, 1988) using tolerance values derived by averaging values for genera and species common to the Shield; (iii) percent contribution of the dominant taxon; (iv) percentage Ephemeroptera, Plecoptera, and Trichoptera (EPT) based on the number of individuals; (v) percentage gastropods; (vi) percentage oligochaetes; and (vii) percentage crustaceans. These common indices are described further by Barbour et al. (1992), Resh et al. (1995), and Somers et al. (1998).

Macroinvertebrate index values were analyzed using two-way factorial ANOVA comparing golf course and reference streams (two categories) and the different seasons (three categories; Wilkinson, 1992). The data (except HBI and correspondence analysis abundance axes) were $\log(x + 1)$ transformed to fulfill assumptions of normality and homogeneity of variance. Post-hoc tests were conducted using the Tukey procedure to determine significant differences between each of the golf course streams and the pooled reference stream data.

Following the general ideas of the reference condition approach, indices calculated for the reference streams were used to determine the normal range of variation (the range of values including 95% of the reference area data) and indices for the golf course streams were compared with this normal range (Kilgour et al., 1998). Golf course stream samples falling within the normal range of variation were considered to be similar to the reference stream samples, whereas samples between the 95th and the 99.9th percentiles were considered to be atypical relative to the reference samples, and samples outside the 99.9th percentile were classed as extremely different (David et al., 1998). To evaluate whether differences between the golf course and reference streams may be attributable to substrate differences, the substrate data for the golf course and reference streams were compared using one-way ANOVA and MANOVA.

To examine variation in the benthic communities based on the 147 samples collected, we summarized the data using ordination methods. Detrended correspondence analysis (DCA) was used to assess the relative length of the gradient representing variation in the macroinvertebrate data. Based on the gradient length obtained from this analysis (>3 SD for the first axis), we determined that an ordination using a unimodal response model would be most appropriate to summarize these data (ter Braak and Prentice, 1988). As a result, correspondence analysis (CA) was used to determine the major patterns in the abundances of macroinvertebrate taxa (Hill, 1974). The three replicate samples from each site and location were analyzed separately. Macroinvertebrate taxa were included in ordinations if they were present in at least three samples and made up >1% of the count in at least one sample. Ordination scores from the first four CA axes were used to calculate generalized (or Mahalanobis) distances between each site and the mean of the reference stream sites. Histograms of these distances were used to illustrate macroinvertebrate community differences between the golf course and reference sites. The three samples were also combined to obtain a single count of at least 300 macroinvertebrates for each stream and date, and the CA was repeated. Patterns in macroinvertebrate taxa evident from this analysis were then related to measured environmental variables using canonical correspondence analysis (CCA; ter Braak, 1986). Canonical coefficients and approximate *t*-tests were used to identify variables that explained significant amounts of variation in the macroinvertebrate abundances. Significance of the CCA axes was assessed using Monte Carlo permutations (99 random permutations). Only environmental variables with variance inflation factors (VIFs) ≤5 were used in the CCA to reduce the problem of multicollinearity (ter Braak, 1986). High VIFs (VIF > 20) indicate that the variable is almost perfectly correlated with other variables and has no unique contribution to the regression equation (ter Braak and Šmilauer, 1998). Water chemistry data were not acquired for the fall 2000 sampling period; thus, fall samples were not included in the canonical correspondence analysis.

RESULTS

Water Chemistry and Substrates

Water temperature was generally higher in the golf course streams than in the reference streams, particularly during the summer when maximum temperatures of >20°C were measured in several of the golf course streams (Table 3). Higher temperatures are likely due to reduced shading along the golf course streams, as well as the presence of upstream ponds. The concentrations of DOC were similar between golf course and reference streams. All other water chemistry variables were higher and more variable in golf course streams than in forested reference streams. Differences were significant when two-sample Student's *t*-tests were used to evaluate reference vs. golf course sites for all the other variables except pH, ammonia, total Kjeldahl N, and total P. Higher levels of nitrate and K reflect nutrient inputs as a result of fertilizer application, and soil liming practices likely contribute to higher alkalinity. These data represent lower flow stream chemistry because water samples were collected at the same time as the macroinvertebrates and high flow (e.g., storm and snowmelt) events were avoided for macroinvertebrate sampling. We would expect higher levels of nutrients

Table 3. Average water chemistry characteristics for the reference and golf course sites (excluding fall 2000).

	Reference (n = 18)			Golf course (n = 17)		
Water quality parameter	Mean	SD	(Range)	Mean	SD	(Range)
Temperature, °C	9	5	(2–19)	10	6	(3–22)
pH	5.7	0.5	(4.4–6.4)	7.0	0.4	(6.2–7.7)
Alkalinity, μeq L^{-1}	95	61	(10–280)	992**	686	(123–2380)
Calcium, mg L^{-1}	2.8	1.3	(1.4–6.7)	18.9**	12.1	(3.0–40.0)
Chloride, mg L^{-1}	5.9	11.5	(0.3–37.2)	29.7*	39.3	(0.5–129.0)
Conductivity, μS	42	42	(3–164)	208**	169	(34–561)
Dissolved inorganic C, mg L^{-1}	2.1	1.2	(0.7–6.3)	12.1**	8.1	(1.9–29.0)
Dissolved organic C, mg L^{-1}	9.5	4.0	(5.3–16.7)	9.1	4.3	(3.2–18.5)
K, mg L^{-1}	0.4	0.2	(0.1–0.8)	2.5**	2.2	(0.4–7.9)
Mg, mg L^{-1}	0.8	0.3	(0.3–1.8)	4.7**	3.0	(0.7–10.0)
Na, mg L^{-1}	3.8	6.0	(0.6–21.6)	19.0*	22.7	(0.6–67.2)
Ammonia, μg L^{-1}	40	35	(2–120)	106	207	(4–792)
Nitrate, μg L^{-1}	38	46	(6–194)	315**	442	(52–1850)
Total Kjeldahl N, μg L^{-1}	462	221	(160–840)	712	555	(260–2440)
Total P, μg L^{-1}	21.9	14.2	(5.8–54.0)	28.2	22.7	(8.2–86.6)
Sulfate, mg L^{-1}	5.0	2.2	(1.8–9.6)	15.1*	10.5	(2.7–33.7)

* Two-way t-test significant at $p < 0.05$.
** Two-way t-test significant at $p < 0.01$.

(including P) and dissolved ions in golf course streams during storm events (e.g., Kunimatsu et al., 1999).

Variability in nutrient and dissolved ion concentrations was also observed in the reference streams, in part reflecting the presence of beaver ponds upstream of three of the seven reference sites. Beaver activity alters water and sediment chemistry through modification of stream hydrology and morphology, and influences the character of water and materials transported downstream (Devito and Dillon, 1993; Naiman et al., 1988; Smith et al., 1991). For example, reference sites below beaver ponds had higher mean values of ammonia (68 vs. 18 μg L^{-1}), TKN (588 vs. 362 μg L^{-1}), and total P (32 vs. 14 μg L^{-1}).

The one-way MANOVA revealed that the forested reference streams were similar in substrate composition to the golf course streams when all of the substrate variables were compared simultaneously ($F = 0.96$, $p = 0.93$). Based on the ANOVAs, only the proportion of log cover was significantly higher in the forested reference streams relative to the golf course streams ($F = 5.49$, $p = 0.04$). A Bonferroni correction for multiple comparisons suggests that this significant, individual test is not truly significant when all variables are considered simultaneously.

Macroinvertebrates

Chironomidae were among the 10 most abundant taxa in all streams, as were Oligochaeta and Tipulidae at all except golf course Site 4 (Table 4). On average, the relative abundance of Oligochaeta was higher at golf course Sites 1, 5, and 6 than at other sites. Amphipoda, Hirudinea, Gastropoda, Isopoda, and Turbellaria were rare in the reference streams, whereas they were abundant in samples collected from one or more of the golf courses. Plecoptera were uncommon at three golf course streams, and Ephemeroptera at two.

Common taxa from golf course Streams 3, 4, and 6 in particular were quite different from those collected from reference streams. Sites 2 and 3 were located on different streams on the same golf course and had quite different dominant macroinvertebrate taxa. Site 2 is on a shorter stream with more vegetated buffer per unit length of stream, whereas Site 3 is on a stream that drains a larger portion of the course and is influenced by highway runoff.

The distribution of the various indices for the golf course sites relative to the normal range defined using the reference stream variation indicates that the total number of taxonomic groups was similar (Table 5). However, HBI scores were higher in 23% of the golf course samples than in reference samples, indicating possible organic and/or nutrient enrichment at these sites. Reduced proportions of EPT individuals and higher proportions of oligochaetes and gastropods were observed in 35, 13, and 32% of the golf course samples, respectively. Crustaceans were more abundant in almost 60% of the golf course samples.

The factorial ANOVAs revealed that the season × site interaction was significant for the proportion of EPT

Table 4. The 10 most abundant (ranked from 1 to 10, highest to lowest relative abundance) macroinvertebrate taxa at the reference streams and six operational golf course sites.

		Golf course sites					
Rank	Reference streams	1	2	3	4	5	6
1	Chironomidae	Plecoptera	Trichoptera	Amphipoda	Isopoda	Chironomidae	Isopoda
2	Ephemeroptera	Chironomidae	Chironomidae	Ephemeroptera	Amphipoda	Trichoptera	Chironomidae
3	Trichoptera	Oligochaeta	Tipulidae	Chironomidae	Turbellaria	Ephemeroptera	Oligochaeta
4	Pelecypoda	Trichoptera	Ephemeroptera	Pelecypoda	Gastropoda	Oligochaeta	Tipulidae
5	Oligochaeta	Tipulidae	Simuliidae	Oligochaeta	Pelecypoda	Pelecypoda	Hirudinea
6	Simuliidae	Ephemeroptera	Oligochaeta	Gastropoda	Zygoptera	Simuliidae	Trichoptera
7	Tipulidae	Coleoptera	Gastropoda	Trichoptera	Hirudinea	Tipulidae	Pelecypoda
8	Plecoptera	Simuliidae	Pelecypoda	Tipulidae	Trombidiformes	Plecoptera	Ceratopogonidae
9	Ceratopogonidae	Anisoptera	Anisoptera	Anisoptera	Chironomidae	Amphipoda	Plecoptera
10	Anisoptera	Ceratopogonidae	Ceratopogonidae	Simuliidae	Oligochaeta	Ceratopogonidae	Amphipoda

Table 5. Classification (%) of the macroinvertebrate samples from golf course sites (n = 69) with respect to the normal range of variation of samples collected from forested reference streams (n = 78). Extremely different = outside the 99.9th percentile, atypical = between the 95th and 99.9th percentile, and not different = within the 95th percentiles.

Sample classification	No. taxa	HBI	% Dominant	% EPT	% Gastropoda	% Oligochaeta	% Crustacea
Extremely different	0	7	0	16	10	4	48
Atypical	1	16	7	19	22	9	10
Not different	99	77	93	65	68	87	42

and the abundance CA axis 1 scores (Table 6). The proportion of Gastropoda, the proportion of Crustacea, and the abundance CA axis 1 scores were significantly different between golf course and reference streams, but not between seasons. Hilsenhoff biotic index values and the proportion of EPT were significantly different between seasons, but not between sites. Post-hoc tests indicated that golf course Sites 3, 4, and 6 were significantly different from the reference streams in terms of percentage crustacea, abundance CA axis 1 scores, and percentage EPT ($p < 0.012$); Sites 1 and 4 in terms of HBI scores ($p = 0.003$); and Sites 2, 3, and 4 in terms of percentage gastropods ($p < 0.001$).

We compared upstream and downstream samples at golf course Sites 1 and 5 (Fig. 1). An increase in the proportion of EPT individuals and slight decrease in the proportion of oligochaetes was observed downstream of golf course Site 1, although oligochaetes still made up 20% of the count (Fig. 1A). By constrast, a reduction in the proportion of EPT individuals and an increase in the proportion of oligochaetes and crustacea was observed downstream of golf course Site 5 (Fig. 1B). There are several important differences between these two courses. The upstream site at Course 1 drains a beaver pond, which undoubtedly influences nutrient and ion concentrations during certain periods; by contrast, the upstream site at Course 5 drains a forested area. The stream on golf course Site 5 drains nine holes and is longer than the stream on golf course Site 1, which drains three holes. Moreover, the grass on golf course Site 5 is mowed to the stream bank along the entire length of the stream, whereas a substantial unmanaged buffer and wetland area surrounds the stream at golf course Site 1 (Table 2). The substrates of these streams consisted predominantly of a mix of clay, sand, and silt (fine substrate), with some gravel and cobble (the remainder was made up of logs and macrophytes). However, the substrate composition at sites upstream and downstream of Course 5 was similar, whereas at Course 1 the downstream site had a greater proportion of cobble and gravel relative to fine substrate than the upstream site.

Table 6. Factorial ANOVA results for comparisons between golf course and reference sites, between streams, between seasons, and the season and site interactions.

	Sites		Seasons		Interaction	
Index	F	p	F	p	F	p
% EPT	1.98	0.162	2.13	0.035	4.00	0.048
% Gastropoda	7.05	0.009	0.73	0.395	2.08	0.152
% Crustacea	8.24	0.005	0.008	0.929	0.38	0.537
HBI	0.56	0.458	4.17	0.043	0.47	0.495
Abundance CA axis 1	6.73	0.010	0.53	0.470	4.99	0.027
Abundance CA axis 2	0.26	0.608	1.88	0.173	1.24	0.267

Correspondence Analysis

We collected 25 taxonomic groups and omitted two of these groups (Decapoda and Lepidoptera) from the ordination because they were exceedingly rare. The first two CA axes explained 28% of variance in the macroinvertebrate community among samples (Fig. 2) and the first four axes explained 49% of the variation among the samples. All of the samples from the reference streams were confined to a relatively small area of the CA ordination relative to the golf course sites. A number of golf course samples were positioned among the reference streams, but one series of golf course samples were scattered along CA axis 1 and a second series was separated from the reference streams along CA axis 2. Invertebrate taxa with a high positive score on CA axis

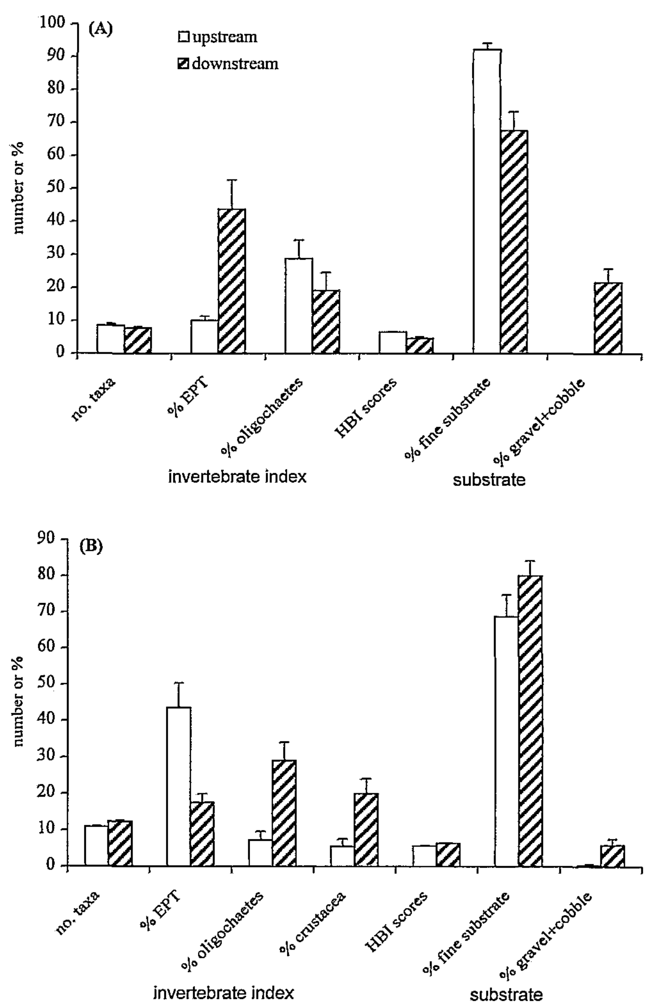

Fig. 1. Macroinvertebrate indices and substrate characteristics (means + 1 SE) at sites upstream and downstream of (*A*) golf course Site 1, and (*B*) golf course Site 5.

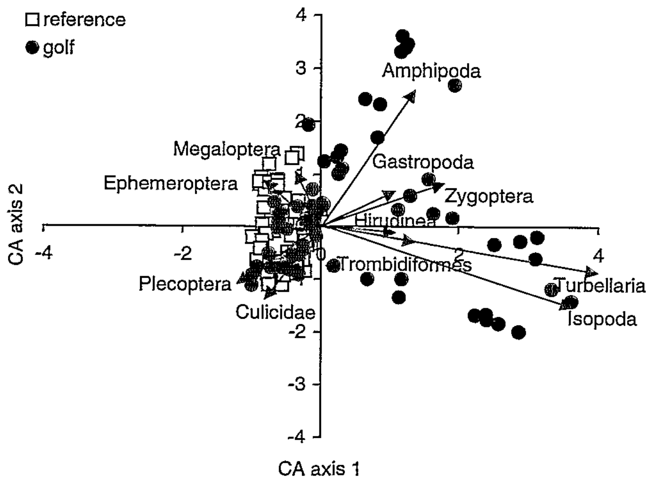

Fig. 2. Correspondence analysis (CA) ordination of the macroinvertebrate data collected in fall 1999 and spring, summer, and fall 2000. Samples from golf course and forested reference streams are separated, and the arrows indicate taxon scores (multiplied by 1.5) that were most indicative of conditions at these sites.

1 (e.g., Turbellaria and Isopoda) characterized one set of golf course samples, from Sites 4 and 6. Golf course samples from Sites 3, 4, and 5 along CA axis 2 had proportionally more Amphipoda. By contrast, Zygoptera, Hirudinea, and Gastropoda were common in a subset of five golf course samples along CA axis 1 (from Sites 4 and 6). Taxa with negative scores on CA axis 1 were most common at the forested reference streams, and some of the golf course sites. These taxa included Ephemeroptera, Megaloptera, Culicidae, and Plecoptera.

The histogram summarizing the distribution of the generalized distances between each sample and the mean of the reference stream samples using the first four CA axis scores revealed that 59% of the golf course samples were outside of the normal range defined by the reference stream samples and thus differed with respect to their macroinvertebrate communities (Fig. 3). Golf course samples that were most different from the reference stream mean ($\geq$8.5 SD) were those collected from Sites 3, 4, and 6. Several samples from other golf course sites were also quite different (Site 5 in particular) to those from reference areas ($\geq$3 SD).

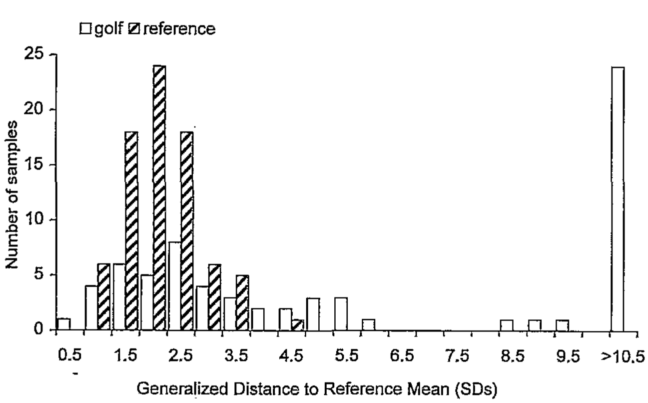

Fig. 3. Frequency distribution of the samples from golf course and reference streams with respect to their distances from the reference site mean macroinvertebrate community based on the four axes of a correspondence analysis (CA) ordination.

Canonical Correspondence Analysis

The CCA identified eight water chemistry variables (temperature, nitrate, TKN, chloride, conductivity, total P, K, and ammonia) that each explained significant ($p <$ 0.05) amounts of variation in the macroinvertebrate data associated with the first three CCA axes (Fig. 4). These environmental variables collectively explained 47% of the variance in the macroinvertebrate community that was summarized by the CCA axes, and the eigenvalues of CCA axis 1 (0.44) and axis 2 (0.18) were similar to those obtained in the CA (0.56 and 0.32). Based on the canonical coefficients and approximate *t*-tests, CCA axis 1 was highly correlated with conductivity and chloride concentration, and CCA axis 2 was highly correlated with ammonia, TKN, and DOC (Fig. 4; Table 7). The CCA axis 3 was highly correlated with nitrate, total P, and K. Overall, golf course Sites 3, 4, and 6 were most different from reference streams in terms of the macroinvertebrate communities. Site 4 varied along CCA axis 2, with high temperatures recorded during the summer, and a high abundance of Amphipoda, relative to other seasons. Site 4 is particularly prone to warming because an unshaded pond is located directly upstream of the sampling site.

The CCA with the physical variables summarized in Table 1 (raw numbers for age, N fertilizer application rate, and the proportion of catchment broadly classed as forested were used in the analysis) identified four variables that each explained significant ($p < 0.05$) amounts of variation in the macroinvertebrate data along the first four CCA axes (Fig. 5; Table 7). The variables collectively explained 27% of the variance in the macroinvertebrate community that was summarized by the

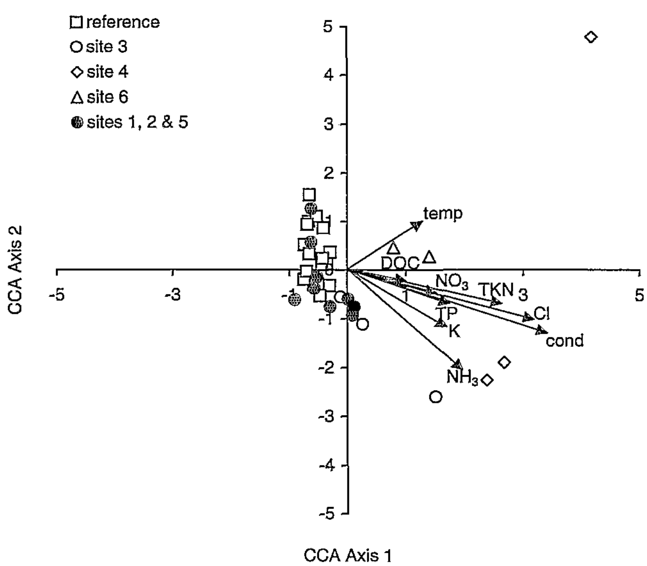

Fig. 4. Canonical correspondence analysis (CCA) diagram showing sample scores obtained in the ordination of macroinvertebrate samples collected from golf course and forested reference streams in fall 1999, and spring and summer 2000. Golf course Sites 3, 4, and 6 were plotted separately from the other golf course sites. Arrows indicate environmental variables (biplot scores were multiplied by 4) that are significantly ($p < 0.05$) correlated with macroinvertebrate distribution. Cond = conductivity, temp = temperature, TKN = total Kjeldahl N, and TP = total P.

Table 7. Coefficients and *t*-values for the canonical correspondence analysis of the macroinvertebrate data using (A) water quality and (B) landscape management and physical features.

Environmental variable	Canonical coefficients				*t*-values			
	Axis 1	Axis 2	Axis 3	Axis 4	Axis 1	Axis 2	Axis 3	Axis 4
A.								
Cl	0.504	0.655	−0.020	−0.048	2.66**	2.35*	−0.06	−0.16
Conductivity	0.599	−0.342	0.126	−0.562	2.45*	−0.95	0.31	−1.50
DOC	−0.042	−1.096	−0.143	−0.056	−0.24	−4.18**	−0.48	−0.21
K	−0.288	0.122	−0.804	0.558	−1.62	0.46	−2.71**	2.05
NH_4–N	−0.609	−2.344	−0.409	0.143	−2.19	−5.72**	−0.88	0.33
NO_3–N	0.069	−0.332	0.975	−0.078	0.47	−1.54	3.98**	−0.35
TKN	0.539	1.990	−0.038	0.106	1.94	4.86**	−0.82	2.74**
TP	0.224	0.609	0.828	−0.629	1.16	2.15	2.57**	−2.12
Temperature	0.098	0.433	−0.572	0.407	0.67	2.02	−2.35*	1.18
B.								
Age	0.181	0.169	0.197	−0.227	1.60	2.30*	2.56*	−3.08**
Fertilizer	0.177	0.368	−0.108	0.277	1.31	4.17**	−1.16	3.13**
Pond	0.159	−0.367	0.382	−0.086	1.11	−3.91**	3.88**	−0.91
Highway	0.432	−0.109	−0.330	−0.023	3.71**	−1.43	−4.14**	−0.30

* Approximate *t*-test significance at $p < 0.05$.
** Approximate *t*-test significance at $p < 0.01$.

CCA axes, and the eigenvalues of CCA axis 1 (0.34) and axis 2 (0.16) were similar to those obtained in CA (0.56 and 0.32). In particular, CCA axis 1 was correlated with the potential influence of highway runoff, whereas CCA axis 2 was correlated with fertilizer application rate and the presence of ponds upstream. Sites 3, 4, and 6 were again most different from the reference streams. The proportion of the catchment that was forested was excluded from the CCA because it was highly negatively correlated with fertilizer application rate, and exceeded our colinearity criterion. The variable presence or absence of a buffer was excluded from the analysis because it did not contribute to the fit of the macroinvertebrate data.

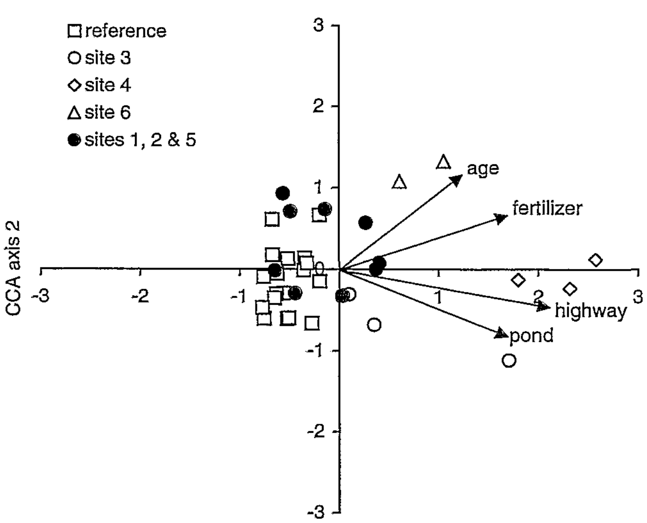

Fig. 5. Canonical correspondence analysis (CCA) diagram showing sample scores obtained in the ordination of macroinvertebrate samples collected from golf course and forested reference streams in fall 1999, and spring and summer 2000. Golf course Sites 3, 4, and 6 were plotted separately from the other golf course sites. Arrows indicate environmental variables (biplot scores were multiplied by 5) that are significantly ($p < 0.01$) correlated with macroinvertebrate distribution.

DISCUSSION

Three of the six golf course streams studied clearly differed from nearby forested reference streams in their macroinvertebrate community composition. Golf course Site 4 had a high fertilizer application rate, received discharge from a pond directly upstream of the site, drained a substantial golf course area, and was potentially further impacted by highway runoff. Sites 2 and 3 were on the same course, but Site 3 had less unmanaged vegetated buffer per unit length of stream, and was potentially impacted by highway runoff. Both 2 and 3 drained a substantial golf course area and received pond discharge upstream, but further upstream than at Site 4. Site 6 was on the oldest course in the study set and received subsurface drainage water from almost the entire golf course area as well as surface runoff from a portion, and had little unmanaged buffer along much of the length of the stream.

There was considerable variability in the macroinvertebrate fauna among the different golf course sites, which was related to land use and golf course management features, and water chemistry variables. The potential impact of highway runoff was one variable that was related to the invertebrate data. Highway runoff can include chemicals that are potentially harmful to aquatic life such as heavy metals, hydrocarbon, and salt used as a deicing agent during winter (Smith and Kastler, 1983). It is thus possible that highway runoff is also affecting the invertebrate community in these streams. However one of the golf course streams that was very different to the reference set (Site 6) was not near a highway.

Of the management features included in the analysis, fertilizer application rate was important, as was the presence of ponds upstream. Fertilizer application rates were based on N fertilizer use, which relates to other fertilizer inputs and generally reflects management intensity for a given course (J.G. Winter, unpublished data, 2000). The ponds that drained into several of the sites are unmanipulated landscape features that are not managed or designed under the specifications developed for wet detention ponds (USEPA, 1999). They are not, for example, periodically dredged to remove accumulated sediments. Consequently, the ponds trap sediments, nutrients, and other chemicals from upstream golf course runoff. Summer anoxic conditions may then promote the release of nutrients from trapped sediments (USEPA, 1999). Discharges from ponds also typically consist of warm water. Other golf course studies, however, have shown that wet detention ponds may considerably lower nutrient outputs (Mallin and Wheeler, 2000). Ponds may thus be useful and advisable from a stormwater management perspective, provided they are designed and managed appropriately. The stream with virtually no unmanaged vegetated buffer was also on the course with the lowest fertilizer application rate, which may have complicated our evaluation of the importance of a buffer zone. However, our qualitative observations (upstream vs. downstream comparisons of Sites 1 and 5 and differences between Sites 2 and 3 on the same course) indicate that the extent of the unmanaged buffer, both directly at the sampling location and upstream, may be another important management feature that merits further evaluation. In other golf course studies, streams with vegetated buffer zones and wooded wetland areas have been found to have lower nutrient output than sites lacking such management practices (Mallin and Wheeler, 2000).

While the number of invertebrate taxa and the contribution of the dominant taxon were similar between golf course and reference streams, we found some differences in the taxa. Although Turbellaria are considered to be sensitive to organic pollution and low oxygen levels (Peckarsky et al., 1990), they were common in golf course streams. Turbellaria prefer moderate rather than low nutrient levels and are generally classed as being dominant in moderately polluted waters (De Lange, 1994). Amphipoda were also more common at the golf course sites, although they are generally found in unpolluted streams (Pennak, 1989). However, several amphipod genera are common in alkaline and brackish waters and have been found to indicate poor water quality conditions (Lenat, 1993). Other groups, such as Isopoda, Gastropoda, Hirudinea, and Trombidiformes, preferring alkaline, nutrient-rich conditions, were more abundant at golf course sites. Isopods feed on dead animal and plant matter and will increase in number under eutrophic conditions, whereas gastropods are herbivores that are likely to flourish in alkaline conditions with abundant periphyton growth (Peckarsky et al., 1990) as seen at several golf course sites. Moreover, since Amphipoda and Isopoda are shredders (Merritt and Cummins, 1996), their abundance in golf course stream may also be attributable to consumption of grass clippings in the streams. Zygoptera and Anisoptera were also common in some of the golf course streams, and are tolerant of organic enrichment, while Ephemeroptera (mayflies) and Plecoptera (stoneflies) are most abundant in cool, unpolluted streams (Lenat, 1993; Peckarsky et al., 1990), and were generally more common in forested reference streams in this study. Observations in field and laboratory studies indicate that stoneflies are sensitive to elevated temperatures, which may account for their scarcity in unforested streams (Quinn et al., 1997). Overall, the higher alkalinity, dissolved ion and nutrient concentrations in the golf course streams favor taxa that are less common in the reference streams.

The rapid bioassessment approach used in this study is intended to be more efficient and cost effective than more detailed taxonomic assessment (Resh et al., 1995). The indices calculated provide an alternative to practical difficulties associated with higher resolution taxonomic comparisons, which may be useful for routine golf course evaluation. Of the simple indices used, the composition measures percentage Crustacea and percentage Gastropoda were most different between golf course and reference streams. Such indices represent a composite of several organisms generally considered to be tolerant of a wide range of pollutants, and in other studies have been found to increase in number in response to increasing impairment (Barbour et al., 1999). In a study of 17 indices for assessing Shield lakes, the index percentage Amphipoda was among those with the greatest power to distinguish between lakes (Somers et al., 1998). Although percentage EPT and the number of taxa have proven to be among the best indicators of water quality in other studies (Barbour et al., 1992; Fore et al., 1996; Kerans and Karr 1994; Sandin and Johnson, 2000) they were not good indicators of golf course impacts on the Shield. The coarse taxonomy inherent in our rapid bioassessment protocol may have contributed to the weaker performance of these metrics (Somers et al., 1998). The HBI and percentage dominant taxon metrics were also not good indicators of golf course impacts, although they have proven useful in other studies (Barbour et al., 1992 and 1999). The HBI was originally designed to evaluate organic pollution (Hilsenhoff, 1988), and was not effective in discriminating between reference and agriculturally impaired sites in Nebraska (Whiles et al., 2000). It may thus not be appropriate for monitoring golf course effects. Although percentage Oligochaeta has been a useful index in some studies (Barbour et al., 1999), it has a variable response to increasing peturbation (Kerans and Karr, 1994) and was not a particularly effective index in this study.

The first axis from the CA ordination of the abundance data was used as a multivariate index, and was able to discriminate between golf course and reference streams. Somers et al. (1998) found that this index was also useful in discriminating between Shield lakes, and recommend that multivariate indices like this be used in addition to simple indices to interpret rapid bioassessment data. Multivariate metrics represent multi-taxa indices that summarize the dominant trends in variation in the benthic community, and are useful for detecting and understanding spatial and temporal trends in the benthic macroinvertebrate fauna (Milner and Oswood, 2000; Norris and Georges, 1993; Somers et al., 1998). Overall, the multivariate approaches used proved effective for evaluating golf course streams.

Golf course runoff to surface waters typically contains pesticides (Ma et al., 1999; Sudo and Kunimatsu, 1992) as well as N and P (Kunimatsu et al., 1999). Oligochaetes and chironomids are reportedly more tolerant of pesticides than many other aquatic invertebrates (Whitten and Goodnight, 1966). While the abundance of these organisms was not generally higher in golf course streams, an increase in the abundance of oligochaetes was observed in our comparison of upstream and downstream sites at golf course Site 5. A wide range of chemicals (e.g., insecticides, fungicides, and herbicides) is applied to the golf courses throughout the year, depending on the pest problem encountered. Pesticides[1] routinely applied to golf courses in Canada include: the fungicides quintozene (PCNB), thiram, and chlorothalonil, iprodine, benomyl, thiophanate-methyl, propiconazole and maneb; the herbicides 2,4-D, mecoprop, dicamba, glyphosate, while MCPA, paraquat, napropamide and chlorsulfuron are used less frequently; and the insecticide malathion most frequently, as well as treatments with diazinon, carbaryl, dimethoate, methoxychlor, propoxur, or pyrethrins (Alberta Environmental Protection, 1998; Anderson et al., 1992). Fungicides in particular are applied at high rates in Canada for snow mold control. Given the lack of published studies on the effect of golf course runoff on stream biota, it is difficult to speculate how these pesticide applications might impact nontarget aquatic organisms. However, there have been toxicological studies on some of the products used. The insecticides diazinon and malathion and the herbicide glyphosate, for example, are toxic to nontarget aquatic invertebrates at environmentally realistic concentrations (Heliövaara and Väisänen, 1993; Stuijfzand et al., 2000; Werner et al., 2000). Furthermore, macroinvertebrate communities were affected by applications of the fungicide carbendazim in freshwater microcosms (Cuppen et al., 2000), while the fungicide propiconazole inteferes with crustacean embryonic development (Kast-Hutcheson et al., 2001). Toxicological effects of pesticides on aquatic invertebrates include a reduction in number of species and abundance, as well as behavioral responses such as increased drift rate (Heliövaara and Väisänen, 1993). These effects have been observed in field studies in agricultural areas, and were related to pesticide contamination (Schulz and Liess, 1999). Pesticide runoff may thus affect the invertebrate communities in these golf course streams, which may contribute to the variance in the macroinvertebrate data that was not explained by the water chemistry variables measured.

The concept of evaluating a potentially impacted site through comparison with reference sites is gaining popularity (Bailey et al., 1998; Reynoldson et al., 1997). In our study, this approach allowed us to evaluate macroinvertebrate communities from streams in situations where sites upstream of golf course impacts were unavailable. The reference conditions in our study are based on the forested streams with acceptable water quality, and our management target is that the invertebrate community in a golf course stream is not different from the reference condition.

Macroinvertebrate community structure was notably different in several of the golf course streams while habitat variables such as substrate type, stream order, and catchment area were similar to the reference set. We thus conclude that there are differences in the benthos that are due to golf course operation. Macroinvertebrates are functionally important in aquatic food webs, and changes in taxonomic composition can affect food web interactions and ecosystem dynamics (Covich et al., 1999). The observed shift from stoneflies and mayflies to gastropods and crustaceans implies impacts (Barbour et al., 1999) that should be avoided, and indicates that better golf course management practices are required at several sites, such as reduced fertilizer application rates and improved pond design.

In summary, we observed differences in the overall benthic macroinvertebrate community at three of the six golf course streams studied, although the causal mechanisms are not known. Our results suggest that golf course land management on the Precambrian Shield can be associated with significant differences in the abundance of certain macroinvertebrate taxa. Studies in the USA have also detected golf course impacts. For example, Lewis et al. (2001) detected sediment contamination by pesticides and metals associated with golf courses, and Mallin and Wheeler (2000) found that some golf courses discharge nutrients at levels great enough to contribute to eutrophication problems in receiving waters. Invertebrate taxa with higher abundances at our golf course sites included Turbellaria, Isopoda, Amphipoda, Zygoptera, and Trombidiformes, while taxa more common at the reference sites included Ephemeroptera, Megaloptera, Culicidae, and Plecoptera. There was less evidence of an effect at half of the golf course streams in our study set, and several factors appear to make Shield streams particularly susceptible to golf course impacts on the macroinvertebrate community: (i) having high fertilizer application rates/intensive management strategies; (ii) receiving pond discharge; (iii) draining a large proportion of the golf course; and (iv) being further potentially impacted by highway runoff.

[1]Pesticide chemical names: quintozene (PCNB), pentachloronitrobenzene; thiram, tetramethylthiuram disulfide; chlorothalonil, tetrachloroisophthalonitrile; iprodine, 3-(3,5-dichlorophenyl)-*N*-(1-methylethyl) 2,4-dioxoimidazoline-1-carboxamide; benomyl, methyl-1-[(butylamino) carbonyl]-*H*-benzimidazol-2-ylcarbamate; thiophanate-methyl, dimethyl 4,4′-(*o*-phenylene)bis(3-thioallophanate); propiconazole, (±)-1-[2-(2, 4-dichlorophenyl)-4-propyl-1,3-dioxolan-2-ylmethyl]-1*H*-1,2,4-triazole; maneb, manganese ethylenebis(dithiocarbamate); 2,4-D, (2,4-dichlorophenoxy) acetic acid; mecoprop, (*RS*)-2-(4-chloro-*o*-tolyoxy)propionic acid; dicamba, 3,6-dichloro-*o*-anisic acid; glyphosate, *N*-(phosphonomethyl)glycine; MCPA, 4-chloro-*o*-tolyloxyacetic acid; paraquat, 1,1′-dimethyl-4,4′-bipyridinium; napropamide, (*RS*)-*N*,*N*-diethyl-2-(1-naphthyloxy)propionamide; chlorsulfuron, 2-chloro-*N*-[[(4-methoxy-6-methyl-1,3,5-triazin-2-yl)amino]carbonyl]benzene-sulfonamide; malathion, diethyl(dimethoxythiophosphorylthio)succinate; diazinon, *O*,*O*-diethyl *O*-2-isopropyl-6-methylpyrimidin-4-yl phosphorothioate; carbaryl, 1-napthyl methylcarbamate; dimethoate, *O*, *O*-dimethyl *S*-methylcarbamoylmethyl phosphorodithioate; methoxychlor, 1,1,1-trichloro-2,2-bis(4-methoxyphenyl)ethane; propoxur, 2-isopropoxyphenyl methyl carbamate.

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REFERENCES

Alberta Environmental Protection. 1998. Factsheet pesticides. Golf course pesticide use and monitoring. Alberta Environmental Protection, Pesticide Management Branch, Edmonton, AB.

Allan, J.D. 1995. Stream Ecology. Structure and function of running waters. Chapman and Hall, London.

Allan, J.D., D.L. Erickson, and J. Fay. 1997. The influence of catchment land use on stream integrity across multiple spatial scales. Freshwater Biol. 37:149–161.

Anderson, A., S. Dean, M. Courneya, and P. Charbonneau. 1992. 1990 Turf industry survey. 1992 Annual Report, Guelph Turfgrass Inst. Guelph, ON.

Bailey, R.C., M.G. Kennedy, M.Z. Dervish, and R.M. Taylor. 1998. Biological assessment of freshwater ecosystems using a reference condition approach: Comparing predicted and actual benthic invertebrate communities in Yukon streams. Freshwater Biol. 39:765–774.

Balogh, J.C., and J.L. Anderson. 1992. Environmental impacts of turfgrass pesticides. p. 221–353. *In* J.C. Balogh and W.J. Walker (ed.) Golf course management and construction. Environmental issues. Lewis Publ., Chelsea, MI.

Barbour, M.T., J. Gerritsen, B.D. Snyder, and J.B. Stribling. 1999. Rapid bioassessment protocols for use in streams and wadeable rivers: Periphyton, benthic macroinvertebrates and fish. 2nd ed. USEPA Rep. 841-B-99-002. USEPA, Office of Water, Washington, DC.

Barbour, M.T., J.L. Pafkin, B.P. Bradley, C.G. Graves, and R.W. Wisseman. 1992. Evaluation of EPA's rapid bioassessment benthic metrics: Metric redundancy and variability among reference stream sites. Environ. Toxicol. Chem. 11:437–449.

Cohen, S., A. Svrjcek, T. Durborow, and N. LaJan Barnes. 1999. Water quality impacts by golf courses. J. Environ. Qual. 28:798–809.

Covich, A.P., M.A. Palmer, and T.A. Crowl. 1999. The role of benthic invertebrate species in freshwater ecosystems. Bioscience 49:119–127.

Cox, E.J. 1991. What is the basis for using diatoms as monitors of river water quality? p. 33–40. *In* B.A. Whitton et al. (ed.) Use of algae for monitoring rivers. Univ. of Innsbruck, Austria.

Cuppen, J.G.M., P.J. Van den Brink, E. Camps, K.F. Uil, and T.C.M. Brock. 2000. Impact of the fungicide carbendazim in freshwater microcosms: I. Water quality, breakdown of particulate organic matter and responses of macroinvertebrates. Aquat. Toxicol. 48: 233–250.

David, S.M., K.M. Somers, R.A. Reid, R.J. Hall, and R.E. Girard. 1998. Sampling protocols for the rapid bioassessment of streams and lakes using benthic macroinvertebrates. 2nd ed. Ontario Ministry of the Environment, Dorset, ON.

De Lange, E. 1994. Manual for simple water quality analysis. International Water Tribunal (IWT) Foundation, Amsterdam, the Netherlands.

Devito, K.J., and P.J. Dillon. 1993. The importance of runoff and winter anoxia to the P and N dynamics of a beaver pond. Can. J. Fish. Aquat. Sci. 50:2222–2234.

Dillon, P.J., and L.A. Molot. 1996. Long-term phosphorus budgets and an examination of a steady-state mass balance model for central Ontario lakes. Water Res. 30:2273–2280.

Dillon, P.J., L.A. Molot, and W.A. Scheider. 1991. Phosphorus and nitrogen export from forested stream catchments in central Ontario. J. Environ. Qual. 20:857–864.

Foote, V. 1998. A new course. Seasons. Summer 1998:17–19.

Fore, L.S., J.R. Karr, and R.W. Wisseman. 1996. Assessing invertebrate responses to human activities: evaluating alternative approaches. J. North Am. Benthological Soc. 15:212–231.

Gordon, N.D., T.A. McMahon, and B.L. Finlayson. 1992. Stream hydrology: An introduction for ecologists. John Wiley & Sons, Chichester, UK.

Heliövaara, K., and R. Väisänen. 1993. Insects and pollution. CRC Press, Boca Raton, FL.

Hill, M.O. 1974. Correspondence analysis: A neglected multivariate method. Appl. Stat. 23:340–354.

Hilsenhoff, W.L. 1988. Rapid field assessment of organic pollution with a family level biotic index. J. North Am. Benthological Soc. 7:65–68.

Hynes, H.B.N. 1963. The biology of polluted waters. 2nd ed. Liverpool Univ. Press, Liverpool, UK.

Hynes, H.B.N. 1975. The stream and its valley. Verh. Int. Ver. Limnol. 19:1–15.

Jeffries, D.S., and W.R. Snyder. 1983. Geology and geochemistry of the Muskoka–Haliburton study area. MOE Data Rep. DR 83/2. Ontario Ministry of the Environment, Dorset, ON.

Kast-Hutcheson, K., C.V. Rider, and G.A. LeBlanc. 2001. The fungicide propiconazole interferes with embryonic development of the crustacean Daphnia magna. Environ. Chem. 20:502–509.

Kerans, B.L., and J.R. Karr. 1994. A benthic index of biotic integrity (B-IBI) for rivers of the Tennessee valley. Ecol. Appl. 4:768–785.

Kilgour, B.W., K.M. Somers, and D.E. Matthews. 1998. Using the normal range as a criterion for ecological monitoring and assessment. Ecoscience 5:542–550.

Kunimatsu, T., M. Sudo, and T. Kawachi. 1999. Loading rates of nutrients discharging from a golf course and a neighboring forested basin. Water Sci. Technol. 39(12):99–107.

Lenat, D.R. 1993. A biotic index for the southeastern United States: Derivation and list of tolerance values, with criteria for assigning water-quality ratings. J. North Am. Benthological Soc. 12:279–290.

Lewis, M.A., S.S. Foss, P.S. Harris, R.S. Stanley, and J.C. Moore. 2001. Sediment chemical contamination and toxicity associated with a coastal golf course complex. Environ. Toxicol. Chem. 20: 1390–1398.

Ma, Q.L., A.E. Smith, J.E. Hook, R.E. Smith, and D.C. Bridges. 1999. Water runoff and pesticide transport from a golf course fairway: Observations vs. Opus model simulations. J. Environ. Qual. 28: 1463–1473.

Mallin, M., and T. Wheeler. 2000. Nutrient and fecal coliform discharge from coastal North Carolina golf courses. J. Environ. Qual. 29:979–986.

Merritt, R.W., and K.W. Cummins. 1996. Trophic relations of macroinvertebrates. p. 647–667. *In* F.R. Hauer and G.A. Lamberti (ed.) Methods in stream ecology. Academic Press, San Diego, CA.

Milner, A.M., and M.W. Oswood. 2000. Urbanization gradients in streams of Anchorage, Alaska: A comparison of multivariate and multimetric approaches to classification. Hydrobiologia 422(423): 209–223.

Molot, L.A., and P.J. Dillon. 1991. Nitrogen/phosphorus ratios and the prediction of chlorophyll in phosphorus-limited lakes in central Ontario. Can. J. Fish. Aquat. Sci. 48:140–145.

Naiman, R.J., C.A. Johnston, and J.C. Kelley. 1988. Alteration of North American streams by beaver. Bioscience 38:753–762.

Norris, R.H., and A. Georges. 1993. Analysis and interpretation of benthic macroinvertebrate surveys. p. 234–286. *In* D.M. Rosenberg and V.H. Resh (ed.) Freshwater biomonitoring and benthic macroinvertebrates. Chapman and Hall, New York.

Ontario Ministry of the Environment. 1983. Handbook of analytical methods for environmental samples. Vol. 1 and 2. Ontario Ministry of the Environment, Toronto, ON.

Peckarsky, B.L., M.A. Fraissinet, M.A. Penton, and D.J. Conklin, Jr. 1990. Freshwater macroinvertebrates of Northeastern North America. Cornell Univ. Press, New York.

Pennak, R.W. 1989. Freshwater invertebrates of the United States. Protozoa to Mollusca. 3rd ed. John Wiley & Sons, New York.

Quinn, J.M., A.B. Cooper, R.J. Davies-Colley, J.C. Rutherford, and R.B. Williamson. 1997. Land use effects on habitat, water quality, periphyton and benthic invertebrates in Waikato, New Zealand, hill-country streams. N. Z. J. Mar. Freshwater Res. 31:579–597.

Raddum, G.C., and A. Fjellheim. 1984. Acidification and early warning organisms in freshwater in western Norway. Verh. Int. Ver. Limnol. 22:1973–1980.

Resh, V.H., M.J. Myers, and M.J. Hannaford. 1996. Macroinvertebrates as indicators of environmental quality. p. 647–667. *In* F.R. Hauer and G.A. Lamberti (ed.) Methods in stream ecology. Academic Press, San Diego, CA.

Resh, V.H., R.H. Norris, and M.T. Barbour. 1995. Design and implementation of rapid assessment approaches for water resource monitoring using benthic macroinvertebrates. Aust. J. Ecol. 20:108–121.

Reynoldson, T.B., R.H. Norris, V.H. Resh, K.E. Day, and D.M. Rosenberg. 1997. The reference condition approach: A comparison of multimetric and multivariate approaches to assess water-quality impairment using benthic macroinvertebrates. J. North Am. Benthological Soc. 16:833–852.

Rosenberg, D.M., and V.H. Resh. 1993. Introduction to freshwater biomonitoring and benthic macroinvertebrates. p. 1–9. *In* D.M. Rosenberg and V.H. Resh (ed.) Freshwater biomonitoring and benthic macroinvertebrates. Chapman and Hall, New York.

Ryals, S.C., M.B. Genter, and R.B. Leidy. 1998. Assessment of surface water quality on three eastern North Carolina golf courses. Environ. Toxicol. Chem. 17:1934–1942.

Sandin, L., and R.K. Johnson. 2000. The statistical power of selected metrics using macroinvertebrates for assessing acidification and eutrophication of running waters. Hydrobiologia 422(423):233–243.

Schulz, R., and M. Liess. 1999. A field study of the effects of agriculturally derived insecticide input on stream macroinvertebrate dynamics. Aquat. Toxicol. 46:155–176.

Smith, A.E., and D.C. Bridges. 1996. Potential movement of certain pesticides following application to golf courses. ACS Symp. Ser. 630:165–177.

Smith, M.E., C.T. Driscoll, B.J. Wyskowski, C.M. Brooks, and C.C. Cosentini. 1991. Modification of stream ecosystem structure and function by beaver (*Castor canadensis*) in the Adirondack Mountains, New York. Can. J. Zool. 69:55–61.

Smith, M.E., and J.L. Kastler. 1983. Effects of rural highway runoff on stream benthic macroinvertebrates. Environ. Pollut. Ser. A 32:157–170.

Somers, K.M., R.A. Reid, and S.M. David. 1998. Rapid biological assessments: How many animals are enough? J. North Am. Benthological Soc. 17:348–358.

Stuijfzand, S.C., L. Poort, G.D. Greve, H.G. van der Geest, and M.H.S. Kraak. 2000. Variables determining the impact of diazinon on aquatic insects: Taxon, developmental stage, and exposure time. Environ. Toxicol. Chem. 19:82–587.

Sudo, M., and T. Kunimatsu. 1992. Characteristics of pesticides runoff from golf links. Water Sci. Technol. 25(11):85–92.

ter Braak, C.J.F. 1986. Canonical correspondence analysis: A new eigenvector method for multivariate direct gradient analysis. Ecology 67:1167–1179.

ter Braak, C.J.F., and I.C. Prentice. 1988. A theory of gradient analysis. Adv. Ecol. Res. 18:271–317.

ter Braak, C.J.F., and P. Šmilauer. 1998. CANOCO reference manual and user's guide for to Canoco for Windows: Software for canonical community ordination (version 4). Microcomputer Power, New York.

U.S. Environmental Protection Agency. 1999. Storm water technology fact sheet. Wet detention ponds. USEPA Rep. 832-F-99-048. USEPA, Washington, DC.

Walker, W.J., and B. Branham. 1992. Environmental impacts of turfgrass fertilization. p. 105–219. *In* J.C. Balogh and W.J. Walker (ed.) Golf course management and construction. Environmental issues. Lewis Publ., Chelsea, MI.

Werner, I., L.A. Deanovic, V. Connor, V. de Vlaming, H.C. Bailey, and D.E. Hinton. 2000. Insecticide-caused toxicity to Ceriodaphnia dubia (Cladocera) in the Sacramento–San Joaquin river delta, California, USA. Environ. Toxicol. Chem. 19:215–227.

Whiles, M.R., B.L. Brock, A. Franzen, and S.C. Dinsmore II. 2000. Stream invertebrate communities, water quality, and land-use patterns in an agricultural drainage basin of northeastern Nebraska, USA. Environ. Manage. 26:563–576.

Whitten, B.K., and C.G. Goodnight. 1966. Toxicity of some common insecticides to tubificids. J. Water Pollut. Control Fed. 38:227–235.

Wilkinson, L. 1992. SYSTAT: Statistics. Version 5.0 ed. SYSTAT, Evanston, IL.