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Abundance, Community Composition, and Site Structure of Lotic Assemblages in Gatineau Park Streams and Streams in the Ottawa-Gatineau Region

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ABUNDANCE, COMMUNITY COMPOSITION, AND SITE STRUCTURE OF LOTIC ASSEMBLAGES IN GATINEAU PARK STREAMS AND STREAMS IN THE OTTAWA-GATINEAU REGION

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ABSTRACT

Sixty-seven sample sites in 45 streams across the National Capital Region and Gatineau Park were sampled between the summers of 2001 and 2008 in order to describe communities within the Park and assess differences in stream fish and invertebrate assemblages along a gradient of urbanization. Additionally the relationships between nutrient concentration and biomass of algae, invertebrates, and fish were quantified, and changes in the size distribution of benthic invertebrates and fish along a gradient of nutrient and anthropogenic impacts were described. Principal Component Analyses indicated that urban and agricultural sites were easily distinguished from forested sites using water quality parameters. Invertebrate and fish assemblages did not differ as clearly among site categories. There were increased nutrient levels in agricultural and urbanized watersheds, however the biomass of primary producers did not proportionally track the increase in nutrients. Slopes of the relationships between biomass and nutrient concentration declined with increasing trophic level, indicating a reduction of the efficiency of energy transfer in more eutrophic systems. Size spectra from nitrogen enriched streams had elevated intercepts and increased residual variance was noted in perturbed watersheds. The current study has provided baseline data for stream communities within and outside of Gatineau Park and in doing so it is clear that Gatineau Park streams are in good, but variable health. In order to monitor change, protect diversity within the Park, and mitigate further stress to these streams, it may be useful to use aspects of the size spectra such as normalized density and the residual variance.

RÉSUMÉ

Au cours de l'été 2001 et 2008, 67 sites ont été échantillonnés dans 45 rivières distribuées dans la région de la Capitale Nationale et dans le Parc de la Gatineau afin d'y décrire les

assemblages de poissons et d'invertébrés à l'intérieur du Parc de la Gatineau et de les comparer avec les communautés prélevées le long d'un gradient d'urbanisation. Les relations entre la concentration en nutriments et la biomasse d'algues, d'invertébrés et de poissons ont également été quantifiées. De plus, les variations dans la distribution de taille des invertébrés benthiques et des poissons le long d'un gradient de nutriments et de pollution d'origine anthropique ont été décrites. Des Analyses en Composantes Principales réalisées avec les variables de la qualité de l'eau ont montré que les sites en milieux urbain et agricole se démarquaient par rapport aux sites en milieu forestier. Cette distinction entre les deux catégories de sites n'était pas aussi marquée pour les analyses effectuées avec les assemblages d'invertébrés et de poissons. Bien qu'une augmentation dans la concentration en nutriments ait été observée dans les bassins versants agricoles et urbanisés, la biomasse des producteurs primaires n'était pas proportionnellement liée à cet enrichissement en nutriments. Les résultats de cette étude ont révélé que les pentes des relations entre la biomasse et la concentration en nutriments étaient moins importantes avec une augmentation du niveau trophique, ce qui indique une réduction de l'efficacité du transfert énergétique dans les systèmes plus eutrophes. Le spectre de taille des rivières enrichies en azote affichait une ordonnée à l'origine plus élevées, et une augmentation de la variance résiduelle a été observée dans les bassins versants perturbés. Cette étude a fournit de l'information de base concernant les communautés des rivières à l'intérieur et à l'extérieur du Parc de la Gatineau. Les résultats suggèrent que les rivières du Parc de la Gatineau sont en bonne santé, mais que leur état est variable. Afin d'être en mesure de suivre les changements dans la qualité de ces écosystèmes, de freiner leur dégradation et d'y protéger la biodiversité, des approches basées sur le spectre de taille peuvent s'avérer utiles et appropriées.

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STATEMENT OF THESIS CONTRIBUTION

The following thesis is presented in manuscript format that will be submitted to journals for publications. As such, there is repetition between chapters, particularly in the methods section to allow for each chapter to stand alone as an individual manuscript. The pronoun we is used throughout as manuscripts will be submitted with more than one author, however, the thesis was completed by me and written by me.

CHAPTER 1 – GENERAL INTRODUCTION

For over forty years, the National Capital Commission has been interested in the conservation of the land within Gatineau Park (Del Degan, Massé et associés Inc., 2010). This has prompted many studies to be undertaken on Park land where habitat alteration has manifested itself in a projected loss of diversity across the region in the future (Del Degan, Massé et associés Inc., 2010). Aquatic environments in the National Capital Region have been subject to a number of habitat alterations through urbanization, wetland destruction, global warming, conversion of forest into farmland, and habitat fragmentation (Del Degan, Massé et associés Inc., 2010). In order to curb the predicted loss of diversity within Gatineau Park and surrounding regions, an ecosystem conservation plan has been implemented by the National Capital Commission.

The ecosystem conservation plan for National Capital Commission Lands has been established for the purpose of long term monitoring and preservation of the biodiversity and ecological integrity within Gatineau Park and surrounding regions (Del Degan, Massé et associés Inc., 2010). The ecosystem conservation plan is a monitoring program that has been aimed at protecting a number of important components common to many habitats such as: vascular plants, bird species, micromammals, freshwater molluscs, anurans (frogs and toads), species at risk, plants at risk, invasive plants, common loon, habitat mosaic, habitat fragmentation, and plant and wildlife potential. The plan outlines objectives which include: reducing the impacts of pressure on ecosystems, maintenance and rehabilitation of processes needed for ecosystem function to remain efficient, maintenance and rehabilitation of plant and animal species, increasing quality of available ecological corridors, conservation of the Park's valued ecosystems, and raising public awareness in order to decrease the impact of recreational activities on the ecological integrity of Gatineau Park. The Gatineau Park Master

Plan was the precursor to the Ecosystem Conservation Plan which will in itself support other studies developed with conservation and recreation within the Park in mind (conservation being of primary importance) (Del Degan, Massé et associés Inc., 2010). The Heritage Conservation Plan, Interpretation Plan, Green Transportation Plan, and Recreational Service Supply Plan will be developed in light of the results from the Ecosystem Conservation Plan and will be influential in the conservation of Park lands. (Del Degan, Massé et associés Inc., 2010). In monitoring the habitats and organisms within the Park, it is hoped that changes will be detected at early stages so that management actions can be implemented to abate further disturbance.

Diversity of flora and fauna within the Park is representative of the area where the lower Laurentians section of the Canadian Shield meets the St. Lawrence Lowlands (Del Degan, Massé et associés Inc., 2010). The Park houses over 1 600 species of plants, over 50 species of fish, 54 mammal species, 232 bird species, 17 amphibians, and 11 reptiles (Del Degan, Massé, 2010). Of these, 131 have been listed as at risk in Canada, Québec, or both (Del Degan, Massé, 2010). It is important to monitor and preserve this vast diversity of species because many organisms found within the Park are not found in any of the surrounding areas in the region. Loss of diversity within the Park could mean a loss of species in the region.

One way to monitor the preservation of habitats is to measure characteristics that are closely linked with important drivers of ecosystem function. Body size, for example, is linked with many biological processes and life history traits which makes it an important aspect of assemblage structure (Blackburn and Gaston 1994, Kozlowski and Gawelczyk 2002, Kamenir et al. 2008, Woodward et al. 2005). The biomass spectrum allows researchers to see how the density of organisms across a range in body sizes within an ecosystem varies (Cyr et al. 1997, Sprules et al. 1991). The shape of the size spectra is generally similar across all spatial and temporal regions and there is a log-linear decrease in density of organisms with increasing body size with a slope of -0.75 (Cyr et al. 1997, Damuth 1981, Knouft 2002, Schmid et al. 2000). Body size is a relevant topic in many fields of ecology and thus predictions surrounding body size can be useful for communities across large scales (Kamenir et al. 2008, Woodward et al. 2005).

The use of body size in ecological research is not a new concept however the underlying concept of the stability of the spectra has remained since the advent of size spectra research. The first spectra were developed from planktonic communities in the pelagic zone of marine systems by Sheldon et al. (1972, 1973). Sheldon was the first to document the overwhelmingly evident trend of consistent biomass estimates across body sizes ranging from bacteria to whales (Kerr and Dickie 2001). It became well understood that this stable body size distribution was driven primarily by predator-prey interactions (Kerr and Dickie 2001). After Sheldon's early work became accepted as ecologically relevant and sound, many others recorded these body size patterns in a plethora of aquatic ecosystems (Kerr and Dickie 2001).

In addition to aspects of the biomass spectrum, many other methods can be used to assess the influence of increased nutrients or human alterations on aquatic environments. Lindeman developed the concept of ecological efficiency in his classic 1942 paper on trophic dynamics which suggested that a system can be quantitatively assessed as the ratio of assimilated energy at two trophic levels (Colinvaux and Barnett 1979). Lindeman's simplified view of energy transfer was a bold move in the field of limnology at the time and although many ecologists were focusing on taxonomic groupings or classes of organisms, Lindeman choose to use the trophic level (Slobodkin 1987) as his principal descriptor. The use of trophic levels did not seem ideal to many researchers especially in aquatic systems where one species can feed at different levels throughout its lifetime and omnivoury blurs the distinct lines of a trophic level (Slobodkin 1987). However, the idea of biological or ecological (Lindeman) efficiency provided the framework for energetic research in the field of limnology for years to come (Slobodkin 1987).

The many studies to proceed Lindeman's paper had a common theme wherein analysis of ecosystems was done from the perspective of groups of organisms sharing and transferring energy resources (Benke et al. 1988). Most early studies on ecological efficiency focused on aquatic ecosystems because the gathering of data on energy transfer between trophic levels was more easily done than in terrestrial systems (Benke et al. 1988). In the 1960's and 1970's ecosystem level research was prevalent and the International Biological Program (IBP) was developed (Benke et al. 1988). A key component of this research effort was to compare energetics across a wide range of ecosystems, however lakes were the primary focus of aquatic systems, and streams were far less studied (Benke et al. 1988). Research in this area continued for over a decade before studies on ecological efficiency began to decline as ecologists turned their attention elsewhere.

To gain an accurate understanding of how assemblages respond to perturbation, it is important to obtain baseline data. Gatineau Park's is comprised of 361 km^2 of relatively pristine land and just northwest of the National Capital Region. It is hedged by the urban centres of Hull and Gatineau. Much of Gatineau Park's land is used for recreational activities such as camping, hiking, and swimming, however these activities are regulated in order to better preserve the natural atmosphere and resources within the Park. Many streams and stream networks ranging from large permanent to small ephemeral exist within Gatineau Park, but few have previously been studied. These smaller streams within the Park can be considered important habitat for young of the year and juvenile individuals from many sport fish taxa that exist in lakes within the Park. Much attention on fish habitat and assemblage organization within the Park has focused on sport fish species in the 55 recognized lakes, and although native minnow presence has been documented in lakes, there have been few comparison studies with urbanized streams. Not only will Gatineau Park serve as a representative of pristine sites for comparison of nutrients, biomass, richness, and size spectra to more urbanized streams, but valuable information of species presence in small streams will be thoroughly documented.

My project is aimed at providing the National Capital Commission with supplementary data for their Ecosystem Conservation Plan on stream environments. The purpose of chapter one was to compare invertebrate and fish communities in streams across the National Capital Region, specifically with respect to evaluating the importance of Gatineau Park in conserving native assemblages and to provide baseline stream community data for the proposed Gatineau Park Ecosystem Monitoring Program which will be implemented in 2010 as part of the 2010 Ecosystem Conservation Plan.

In addition, chapter two used biomass and the biomass spectrum to examine assemblage composition along a gradient of urbanization. The primary objective of this study was to examine how the size spectrum of benthic invertebrate communities varied along an urbanization gradient. The secondary objective of this study was to examine how biomass of periphyton, invertebrates, and fish varied along a gradient of nutrient levels (total phosphorus and total nitrogen). Overall, the study attempted to fill a current void in ecological efficiency research by investigating three trophic levels using a large number of streams existing in watersheds with different land-uses.

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CHAPTER 2 - COMPOSITION AND RICHNESS OF FISH AND INVERTEBRATE ASSEMBLAGES IN SMALL STREAMS WITHIN THE OTTAWA REGION

INTRODUCTION

Gatineau Park is composed of a large portion of protected land that supports a wealth of diversity representative of the area where the Lower Laurentians division of the Canadian Shield meets the St. Lawrence Lowlands (Del Degan, Massé, 2010). There are a number of highly important and sensitive habitats located within the Park including the Eardley Escarpment and Plateau, and the Gatineau Hills (Del Degan, Massé, 2010). All three of these physiographic zones support a unique set of plant and animal species in danger of being lost through the projected deterioration of the Park. Gatineau Park is host to over 1 600 species of plants, over 50 species of fish, 54 mammal species, 232 bird species, 17 amphibians, and 11 reptiles (Del Degan, Massé, 2010). Of these, 131 have been listed as at risk in Canada, Québec, or both (Del Degan, Massé, 2010). It is important to consider that many of these species are not found in the National Capital Region outside of the Park, so conservation of the protected land within the Park is essential to preserve much of the diversity in this area.

In response to growing public concerns about threats to the ecological integrity of Gatineau Park, the National Capital Commission (NCC) has recently developed and implemented the Gatineau Park Ecosystem Conservation Plan (NCC, 2008). The plan outlines a number of objectives which include: reducing the impacts of pressure on ecosystems, maintenance and rehabilitation of processes needed for ecosystem function to remain efficient, maintenance and rehabilitation of plant and animal species, increasing quality of available ecological corridors, conservation of the Park's valued ecosystems, and raising public awareness in order to decrease the impact of recreational activities on the ecological integrity of Gatineau Park.

Although lakes in Gatineau Park have been reasonably well characterized, biotic information on streams is comparatively sparse, and, at present, is insufficient to establish a baseline for systematic surveillance as recommended in the Gatineau Park Ecosystem Conservation Plan (NCC, 2008). To this end, the current study has two major purposes: (1) to compare the general trend of invertebrate and fish communities in streams across the National Capital Region, specifically with respect to evaluating the importance of Gatineau Park in conserving native assemblages; and (2) to provide baseline stream community data for the proposed Gatineau Park Ecosystem Conservation Plan (NCC, 2008) which will be implemented in 2010 as part of the 2010 Ecosystem Conservation Plan.

METHODS

Study area and sample sites

Sixty-seven sites from 45 streams across the National Capital Region (Ottawa, Ontario, Canada) and Gatineau Park were sampled between the summers of 2001 and 2009 for periphyton, benthic macroinvertebrates, and fish to compare the distribution of fish and macroinvertebrate assemblages between pristine and perturbed sites in the National Capital region. I sampled twenty-seven of the total 67 sites in 2008. Eleven sites were sampled during the summer of 2005 (Jennifer Lento, unpublished data), six were sampled in 2003 (Uta Grüenert, unpublished data), and the remaining twenty-three sites were sampled during the summers of 2001 and 2002 (Stephenson, 2007). Methodology used for sampling periphyton, invertebrates, and fish were similar across all years and a general outline will be presented.

Periphyton and Macroinvertebrate Sampling Protocol

Six randomly selected cobbles were collected before fish sampling at each site starting downstream and gradually moving upstream through the 10 metre riffle segment. Above each cobble, current velocity was measured using a pygmy meter at 60% depth of the water column. Each cobble was carefully placed in a whirl pack with a known amount of 95% ethanol (usually between 100-200 mL).

Once in the lab, cobbles were placed in the fridge for 24hrs at which time chlorophyll *a* concentration in the ethanol was estimated spectrophotometerically to obtain an index of periphyton standing stock (Ostrofsky and Rigler 1987). Cobbles were then scrubbed in order to remove all invertebrates from the surface into a sieve of 1 mm mesh size. Material retained by the sieve was preserved with 95% ethanol to await sorting and measuring. After cobbles were scrubbed and dried, I estimated cobble surface area from the mass of aluminium foil needed to wrap the entire surface of the rock with no overlap.

The macroinvertebrates were sorted and identified to family level using keys from McCafferty (1998). Once sorted, organisms were measured for body length, including appendages using OPTIMAS 6.5 imaging software (Media Cybernetics, Silver Springs Maryland). To account for the loss of small macroinvertebrates through the 1mm mesh sieve, a sieve retention model was used to calculate the size specific correction factor (Morin et al. 2004). The model calculates the probability (p) that an organism is retained in a sieve as:

$$\ln(p/(1-p)) = -2.84 + 5.8\log 10(RL) - 3.18\log 10(RL)\log 10(M)$$

where RL is the body length/mesh size and M is the mesh size (mm)

Mass for each individual invertebrate was calculated using length-dry mass regressions developed by Benke et al. (1999). Invertebrate biomass (g DM/m^2) was calculated by dividing the dry mass of organisms collected on each cobble by the surface area of the cobble.

Fish Sampling Protocol

Fish were collected using a Smith-Root LR-24 backpack electrofisher set at 300 V initially and decreased when necessary (if fish/crayfish were dying) during base-flow conditions throughout the summers of 2001-2003, 2005, and 2008. Sample sites were sectioned off using block nets (3 meters long, 0.4 cm mesh for small streams and 15 meters long, 0.4 cm mesh for larger streams). Surface area of the sampled stream segment was estimated by multiplying the length of the segment (10 m) by the average width of the stream (estimated from width measurements taken at five transects of the stream). Fish biomass was estimated using the attrition method: repeated passes were made over the same area until the numbers of fish captured in a pass dropped below 50% of the initial pass. As fish were shocked, they were captured using dip nets and transferred into buckets, or when missed, were washed downstream into the furthest downstream block net. Fish were identified to species level and measured for fork length, total length, and body depth (mm) after each pass using a standard ruler and clipboard. Once fish were processed, they were released downstream of the sample site as to not be included in the next pass. After all three passes were completed the furthest downstream net was checked for fish. Any fish collected from the net were processed in the same manner as described above and were counted as being caught in the first pass.

Fish drymass was estimated for each individual using a length-weight regression from Randall and Minns (2000) or Schneider et al. (2000). Fish data were summed across species-size classes for each pass and site. I then generated the population size at each site using Microfish 3.0 demonstration version (Van Deventer 1989) from maximum likelihood estimates of abundance based on the number of individuals collected and the reduction in fish counts with each additional pass. These population estimates were multiplied by the observed averaged drymass for each taxon-size class combination and then divided by the area of the sampling location to obtain the corrected biomass estimate (in g DM/m^2) for each species-size class combination in each stream.

Water Quality

Water quality was measured at each site one to three times throughout the summers of 2001-2003, 2005, and 2008. Conductivity (Cond, μs/cm), chloride (CL, mg/L) total phosphorus (TP, mg/L), nitrite, nitrate, and total kjehldahl nitrogen were determined by ROPEC laboratories (Ottawa, Ontario, Canada). Total nitrogen (TN, mg/L) was calculated by summing nitrite, nitrate, and total kjehldahl nitrogen for each site and sampling date. Total nitrogen and total phosphorus were averaged across the season and the resultant average values were used in statistical models.

Site Type Designation

Sites were placed into categories corresponding to whether they were located in, or outside the boundaries of Gatineau Park, whether they drained watersheds dominated by one of three land types (forest, agriculture, or urbanized land), and whether they were located on the Canadian Shield. Google Earth® was used to assess the dominant land-use type by visually inspecting the land within each watershed boundary (2010). An estimation was made as to whether watersheds containing each sample site were classified as mostly agricultural, mostly urbanized, or mostly forested land-use. A digital map of the National Capital Region (Natural Resources Canada, Geological Survey of Canada, 2003) displaying areas designated as Canadian Shield was used in conjunction with Google Earth® to determine whether the watershed was located on the Canadian Shield.

Statistical Analysis

ANOVA

In order to test for differences in mean water quality parameters among site categories, an ANOVA was employed in Systat v.12 using the model:

water quality parameter = site category

In the above model, water quality parameters utilized were $\log_{10}$ of: total phosphorus (mg/L), total nitrogen (mg/L), conductivity (µs/cm), and chloride (mg/L). Site categories included: park/non-park, agriculture/urban/forest, and shield/non-shield.

Principal Component Analysis

A PCA was run using Systat v.12 on a correlation matrix for environmental variables, invertebrate biomass, and fish biomass. Only dominant taxa (taxa found at >30% of sites) were included in the ordination because we were interested in the general trends among site types. When all taxa were included in the ordinations very little pattern could be detected most likely because rare taxa reduced the signal to noise ratio. In removing rare taxa these issues were eliminated and general trends were more clearly represented. Forty-eight of the total 70 invertebrate taxa and 25 of the total 33 fish taxa were removed from ordinations. All ordinations were made on log transformed data (log (x+.001g) for invertebrates and log (x+.005g) for fish).

RESULTS

Streams in the National Capital Region were varied in terms of nutrient concentration both in mean and range of values along a gradient of perturbation. In general, nutrient concentration was found to be higher in streams located in perturbed areas (non-park, non-shield, agricultural, and urban watersheds). Total phosphorus varied between 0.005 and 0.6 mg/L for all sites types (Appendix C). Highest total phosphorus values and largest spread of data points were seen in urban, non-shield, non-park sites (Figure 2.1.). Means of total phosphorus were about 3 times higher in non-park sites than park sites ($AveTP_{Park} = 0.03 \pm 0.05$ mg/L, $AveTP_{Non\text{-}Park} = 0.09 \pm 0.12$ mg/L) and were significantly different between all site types ($p_{park/non\text{-}park} < 0.001$, $p_{land\text{-}use} < 0.001$, $p_{shield/non\text{-}shield} < 0.001$) (Table 2.1.).

Total nitrogen varied between 0.02 and 5.23 mg/L with highest values found at urban and agricultural sites (Figure 2.2.). Although agricultural and urban watersheds had similar average TN values ($AveTN_{Urban} = 1.86 \pm 1.16$ mg/L and $AveTN_{Agric} = 1.36 \pm 0.82$ mg/L), higher values were seen in urban watersheds with few data points existing below the mean. Sites outside of Gatineau Park had approximately 4 times higher TN values than did sites within the Park ($AveTN_{Park} = 0.44 \pm 0.31$ mg/L, $AveTN_{Non\text{-}Park} = 1.52 \pm 1.07$ mg/L). Again total nitrogen was significantly different among all site types ($p_{park/non\text{-}park} < 0.001$, $p_{land\text{-}use} < 0.001$, $p_{shield/non\text{-}shield} < 0.001$) (Table 2.1.). Conductivity varied between 53 and 1684 μs/cm in the Ottawa Region with high values being found in urban, non-shield, non-park sites (Figure 2.3.). Spread of data was less in protected sites (park, shield, forest) and significant differences were seen in conductivity among all site groupings ($p_{park/non\text{-}park} < 0.001$, $p_{land\text{-}use} < 0.001$, $p_{shield/non\text{-}shield} < 0.001$) (Table 2.1.). Chloride varied between 0.06 and 385 mg/L with highest values found at urban sites, most likely due to road salt (Figure 2.4.). Chloride values are very low at protected sites and data variability is much greater at perturbed sites with significant differences in chloride values found among all site types ($p_{park/non\text{-}park} < 0.001$, $p_{land\text{-}use} < 0.001$, $p_{shield/non\text{-}shield} < 0.001$) (Table 2.1.).

Richness of fish and invertebrates showed little correlation with environmental variables although both increased with total biomass of the assemblage (Figure 2.8.). An ANCOVA showed that there was no significant difference in the relationship between richness and biomass among dominant land-use types ($p > 0.05$) for both invertebrates and fish. Moreover, sites with high invertebrate richness also showed high fish richness (Figure 2.9.). An ANCOVA showed that there was no significant difference in the relationship between fish and invertebrate richness among dominant land-use types ($p > 0.05$). Since richness should increase with increasing watershed size, this relationship was also explored for 53 of the 67 sites where watershed area was measured (Figure 2.10.). An ANCOVA showed that there was no significant difference in the relationship between richness and watershed area among dominant land-use types ($p > 0.05$) for both invertebrates and fish.

Overall, water quality parameters differed between protected and perturbed groups of sites. Axis I of the principal components ordination explained 66% percent of the variance in sites and was primarily reflected by the gradient in chlorides and conductivity with these two factors being closely correlated (Figure 2.5.). Sites on the right hand side of the ordination had higher chloride and conductivity, whereas sites of the left side of the ordination had low chloride and conductivity values. Axis II reflected more of a gradient in the ratio of total phosphorus to total nitrogen. Although these variables contribute to both ordination axes, total phosphorus and total nitrogen were more clearly associated with axis II than were conductivity and chloride. Positive values on axis 2 corresponded to sites with relatively large amounts of TP given TN values. In this ordination space, sites located in protected (forest) and perturbed (urban/agricultural) watersheds can easily be distinguished from each other, however there was much overlap between urban and agricultural sites. Urban and agricultural sites tended to have high chloride and conductivity and thus were located on the right side of the ordination, whereas forested sites had low chloride and conductivity values and scored low on the first ordination axis. Urban sites tended to have higher total nitrogen and total phosphorus, and although there were some forested sites that had low total nitrogen values, all forested sites generally showed a trend towards low total phosphorus. There was no clear difference in the TN:TP ratio of site groupings and this was reflected in the absence of discrimination between groupings on axis 2. A similar trend was seen for park/non-park sites as was seen in dominant land-use types, however shield/non-shield sites were harder to interpret because site groupings were not distinct. Non-park sites had high total nitrogen, whereas park sites had low total phosphorus.

Among sites, invertebrate assemblages varied mostly in total biomass and in the ratio of sensitive to tolerant taxa. Only 43% of the variation in community composition was explained by the first two axes of the principal component ordination of biomass of dominant invertebrate taxa. The first axis, accounting for 31% of the variation, represented mostly the biomass of the dominant taxa in the region, and was strongly correlated to total biomass (BINV). The second axis, uncorrelated to total biomass, represented the biomass of Ephemeroptera, Plecoptera, and Trichoptera families (EPT taxa generally considered to be sentitive to environmental degradation (Quinn et al. (1997)) relative to biomass of Isopods and Oligochaetes (generally considered to be tolerant (Quinn et al. (1997)). Most of the forested sites in the Park clustered in the upper left quadrant (low overall biomass but relatively high proportion of sensitive taxa) but many of the Park sites bordering the city of Gatineau (DEST) and the municipality of Chelsea (Notch, Mine3, MO, LUS), and those near the high usage area surrounding Lake Philippe (Taylor, Renaud2) had a higher proportion of biomass of tolerant taxa (Figure 2.6.). The scatter seen in Figure 2.6. showed that sites were different in terms of invertebrate composition, however sites groupings do not differ as much. The scatter, or variability, within site groupings is approximately the same as among groupings.

Fish, on the other hand, although they accounted for less variance among sites than environmental variables, showed more of a trend than seen with invertebrates. 53% of the variation in community composition was explained by the first two axes. The first axis explained 37% of the variance in community composition and was strongly correlated with longnose dace, a fish common in streams. The second axis, similarly to what was found with invertebrates, tended to be correlated more closely with sensitive taxa. In this case brook trout (Figure 2.7.). Rockbass and logperch were found together often, while most of the total fish biomass (BFISH) was composed of creek chub and white sucker. Urban and agricultural sites, loosely clustered in the lower right side of the ordination space, were more closely associated with tolerant species such as rock bass and logperch. A number of forested sites, located in the mid and upper left quadrant, were closely associated with brook trout. Forested sites were characterized by low biomass as they were located opposite total fish biomass and the two species representing the highest biomass at sites (creek chub and white sucker). Similar trends were seen for park/non-park and shield/non-shield sites.

DISCUSSION

Streams in the National Capital Region were characterized as having higher nutrient concentrations in urbanized and agricultural watersheds located outside Gatineau Park. Not only were water quality parameters significantly different between site locations (park/non-park, shield/non-shield) and dominant land-use types, but water quality parameters showed a clear trend with protected sites having low chloride, conductivity and total phosphorus. On the other hand, both invertebrate and fish community assemblages showed comparatively little variation among groups of sites. The best correlate of fish and invertebrate richness proved to be the biomass of each respective assemblage (all taxa combined), and larger biomasses tended to be associated with urban and agricultural sites. Moreover, it is clear that biomass was positively correlated with richness. Because of the theory of island biogeography developed in 1967 by MacArthur and Wilson, we expect to see a positive correlation between watershed area and species richness. The positive correlations between watershed area and species richness and between species richness and biomass made it unclear as to whether the trends seen in this study were real, or whether they were artefacts of larger watersheds existing solely in perturbed settings. Figure 2.10. illustrated that the positive trend between watershed area and species richness was consistent among dominant land-use types indicating that the trends described in this study for fish and macroinvertebrates were not artefacts of large watersheds occurring in urban settings only.

The observed differences in water quality between sites were indicative of the different processes that influence water quality in natural and urbanized streams. In comparatively undisturbed areas, stream water quality is altered by processes such as rain-fall, sediment input, and weathering, whereas in anthropogenically altered systems, water quality may be affected by agricultural and industrial practices and urbanization (Qadir et al. 2008, Yayintas et al. 2007). In agricultural watersheds, cattle and fertilizer usage can increase inputs of nutrients such as phosphorus and nitrogen (Nash et al. 2009). In urbanized landscapes, the greater proportion of impervious surfaces and resulting runoff to storm sewers are associated with increased nutrient levels and conductivity (Brabec et al. 2002, Canobbio et al. 2008, Paul and Meyer 2001, Walsh et al. 2005). Wastewater and lawn fertilizers lead to elevated nutrients in streams and it has been shown that nitrogen levels can be affected up to hundreds of kilometres downstream of the source pipe (Paul and Meyer 2001).

In my study, chloride was found to be much higher in urban watersheds than any other watershed type. In fact, in forested, and park watersheds, chlorides were almost negligible. The cause of these high chloride levels could be the combination of two simple factors. First, road salts are applied to roads in the winter months to control road ice and to increase safety of operating motor vehicles in urbanized areas (Ramakrishna and Viraraghavan 2005). This application is known to be detrimental to water quality through increased chloride concentrations (Godwin et al. 2003, Klein 1979, Ramakrishna and Viraraghavan 2005). Salt becomes available to the environment in a number of ways: by dissolving in melting snow and running into water systems and ground water directly, when vehicles spread salt through driving at high speeds making salt available to the water table along road sides, when ice and snow treated with salt is removed and placed in a specific area (Ramakrishna and Viraraghavan 2005). These concentrations of chloride can exceed what is tolerable for sensitive organisms (Klein 1979) and can be carried downstream for long distances (Ramakrishna and Viraraghavan 2005). Although many cities are attempting to decrease salt deposition by using alternative measures (City Of Ottawa, 2010), there is still a higher than normal input of chlorides into streams due to road salt. In addition to road salt use, sample sites from the current study located in Ottawa, Hull and many of the agricultural watersheds south of these areas are located on marine clays of the old Champlain Sea (Crawford 1968). In effect, the higher salt content in these areas may be due to both the higher salt content in the clay and the use of road salt.

By contrast, invertebrate biomass showed little variation among comparatively disturbed versus undisturbed sites, although disturbed (urban, agriculture, non-park) tended to show more variability. As has been reported (Quinn et al. 1997, Quinn and Hickey 1990), there has been some evidence that certain taxa, notably sensitive species such as Plecoptera, predatory Trichoptera, and Ephemeroptera, were more common in comparatively undisturbed sites, while tolerant taxa, notably Chironomids and snails, were more common in urban and agricultural sites. Quinn and Hickey (1990) reported that with increasing levels of development within watersheds, there was an increase in biomass. However only at high levels of catchment development (<70%), was a change in community structure found with more tolerant taxa, and taxa associated with increased algal biomass being found (Quinn and Hickey 1990). Since macroinvertebrate assemblages seem to respond primarily to local conditions such as water chemistry, channel width and depth, and water velocity (Richards et al. 1996; Sponseller et al. 2001), we would expect to see variation in water quality reflected in variation in macroinvertebrate assemblages in the current study. Few studies examine the effects of water quality or land development on individual taxa biomass (Quinn et al. 1997, Quinn and Hickey 1990) while indices such as functional feeding guilds or richness of specific taxa groupings (EPT) are predominantly used as metrics of community health or diversity (Richards et al. 1996; Sponseller et al. 2001). Even so, studies agree that invertebrates respond to local conditions such as nutrient inputs from agricultural or urbanized land-use and increased periphyton biomass. The similar degree of variability in invertebrate assemblages within site groupings and among sites groupings could have led to the lack of response of invertebrate assemblages as seen in the current study. The cause of this result could be because water quality varied not only between groups, but also within them, blurring the lines between the site groupings.

Fish communities showed more variation among sites than benthic invertebrates, but again, were not particularly informative in distinguishing groups of sites. The principal determinant of this stronger relationship between perturbed and protected sites was the presence/absence of brook trout. Brook trout are generally a cold water species that prefer temperatures below 18 °C and water that is fast moving and highly oxygenated (Curry et al. 1997). They do not do well in eutrophic situations. Brook trout were usually found alone at sites and were only found in sites located within Gatineau Park. Although there are some natural populations of brook trout within Gatineau Park, this species is stocked and so their lone presence at sites may have been partially due to the stocking of brook trout that occurs within the Park in many lakes.

There are a number of sites that exist within the Gatineau Park boundaries which have similar invertebrate and fish assemblages to those found in more urbanized or perturbed sites. Sites bordering the city of Gatineau and the municipality of Chelsea and those near the high usage area surrounding Lake Philippe had a higher proportion of biomass of tolerant taxa than sites found within less perturbed areas of Gatineau Park. Although these sites drain watersheds within the forested region of Gatineau Park, most of the streams in the current study listed as being compositionally similar to urbanized streams run through urbanized land for at least some of their extent. In running through perturbed land, water within these streams has the opportunity to pick up nutrients and sediments as run-off or through ground water inputs. This alteration in nutrients, sediments, and oxygen content can create a habitat that is no longer suitable to sensitive taxa like E.P.T taxa and brook trout, however is more inhabitable for tolerant taxa such as chironomids, gastropods, or white suckers (Quinn et al. 1997).

There are a number of fish species of special concern found throughout Gatineau Park. These species include: the stonecat (*Noturus flavus*), yellow bullhead (*Ameiurus natalis*), margined madtom (*Noturus insignis*), spring cisco (*Coregonus artedi*), brassy minnow (*Hybognathus hankinsoni*), and the bridle shiner (*Notropis bifrenatus*). Of these species, only the yellow bullhead was found in streams within Gatineau Park and was found infrequently and in low abundance. This species was not included in the ordination plot because it was uncommon.

The current study has provided important baseline information on water quality, benthic invertebrate communities, and fish community structure for a number of important stream systems in Gatineau Park. This helps to fill some important biological information gaps and more importantly, provides valuable information on which to design and implement a systematic stream surveillance program as identified in the Gatineau Park Ecosystem Conservation Plan. From the results of this study it is evident that small streams within Gatineau Park are in good health, with some variation shown by sites bordering the Park in high impact areas being more similar to perturbed sites in terms of invertebrate and fish composition. In streams of good health, it would be expected to see a mix of sensitive and tolerant taxa and species richness estimates that reflect such assemblage composition. Perturbed sites may be dominated in composition by high biomass of tolerant taxa with few to no sensitive species and high nutrient values. It is my assertion that monitoring these sites more closely in order to mitigate further impact could be a necessary goal for the Gatineau Park Ecosystem Conservation Plan. It could also be useful to study more closely these impacted sites to investigate the cause of changes and potentially to make predictions for future change. In order to properly assess the differences in assemblage composition between urbanized and protected streams in the National Capital Region, it would be useful to sample similar sites in streams located within Gatineau Park and directly outside of the Park in more urbanized landscapes. This has not been fully addressed by the current study, but could open an avenue for future research in this area.

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TABLES AND FIGURES

Table 2.1. – ANOVA results for nutrient values (total phosphorus (mg/L), total nitrogen (mg/L), conductivity (µs/cm), and chloride (mg/L)) among site types (shield/non-shield, agricultural/forest/urban, and park-non-park).

Model terms	p-value	RMS	R2
$\log_{10}$TP ~ Shield/Non-Shield	< 0.001	0.13	0.30
$\log_{10}$TP ~ Dom. Land-use	< 0.001	0.12	0.36
$\log_{10}$TP ~ Park/Non-Park	< 0.001	0.14	0.23
$\log_{10}$TN ~ Shield/Non-Shield	< 0.001	0.18	0.07
$\log_{10}$TN ~ Dom. Land-use	< 0.001	0.10	0.47
$\log_{10}$TN ~ Park/NonPark	< 0.001	0.12	0.39
$\log_{10}$Cond ~ Shield/Non-Shield	< 0.001	0.14	0.33
$\log_{10}$Cond ~ Dom. Land-use	< 0.001	0.09	0.57
$\log_{10}$Cond ~ Park/NonPark	< 0.001	0.11	0.45
$\log_{10}$CL ~ Shield/Non-Shield	< 0.001	0.82	0.36
$\log_{10}$CL ~ Dom. Land-use	< 0.001	0.68	0.48
$\log_{10}$CL ~ Park/NonPark	< 0.001	0.75	0.42

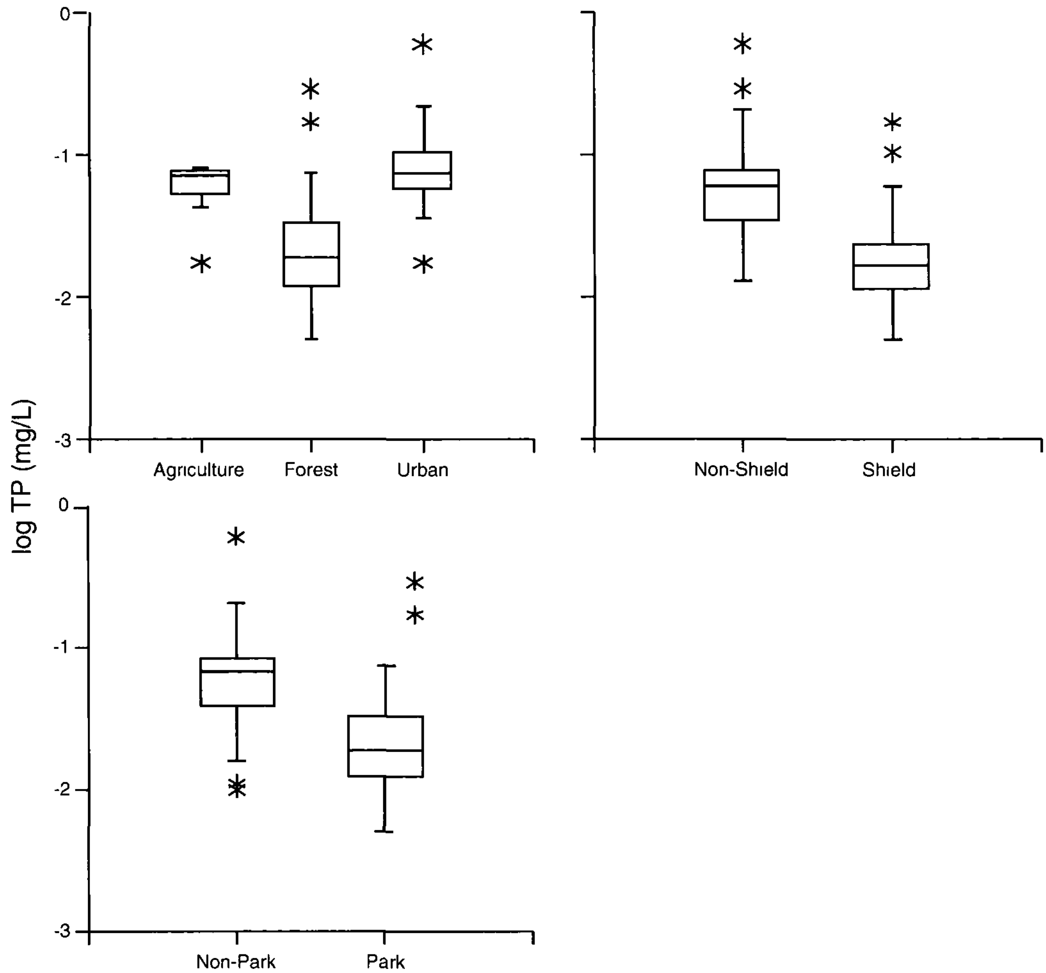

Figure 2.1. Box plots displaying total phosphorus (mg/L) values across site locations and dominant land-use types in streams in the Ottawa region.

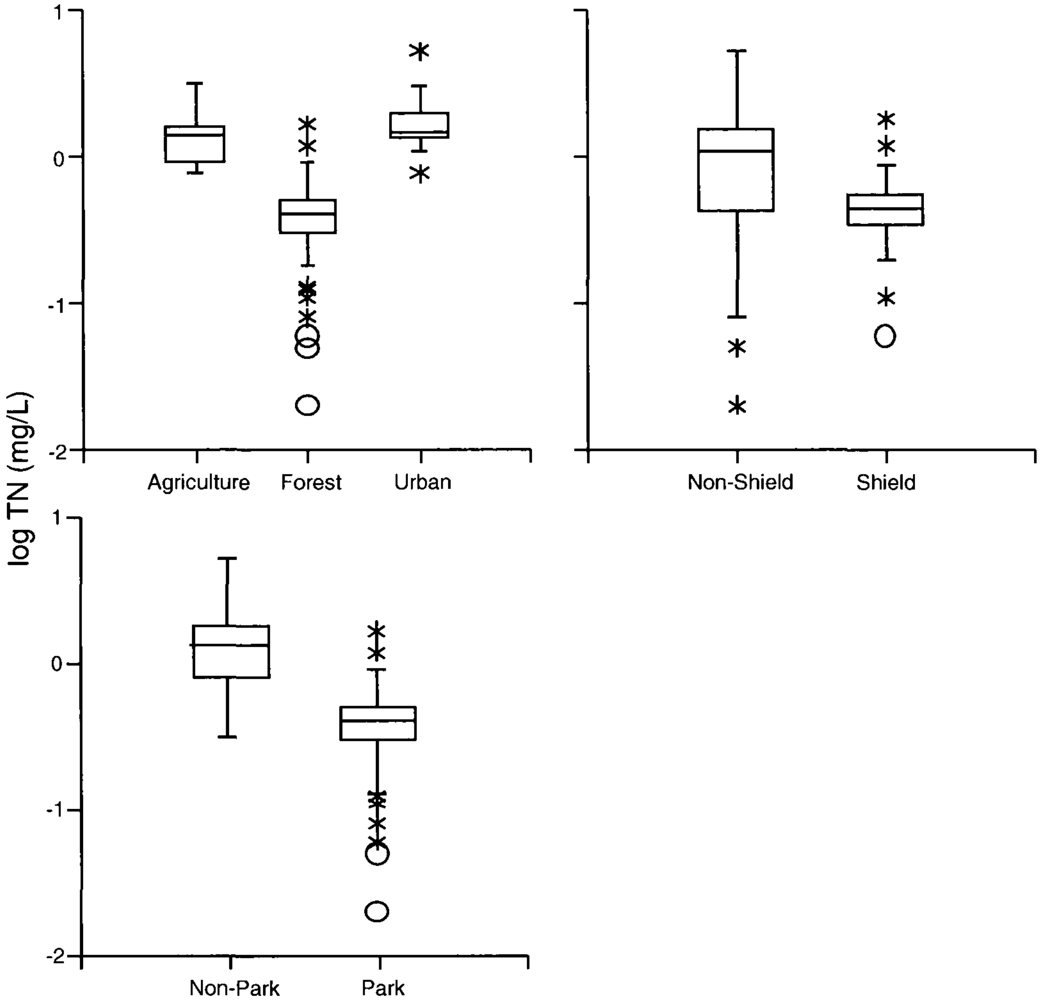

Figure 2.2. Box plots displaying total nitrogen (mg/L) values across site locations and dominant land-use types in streams in the Ottawa region.

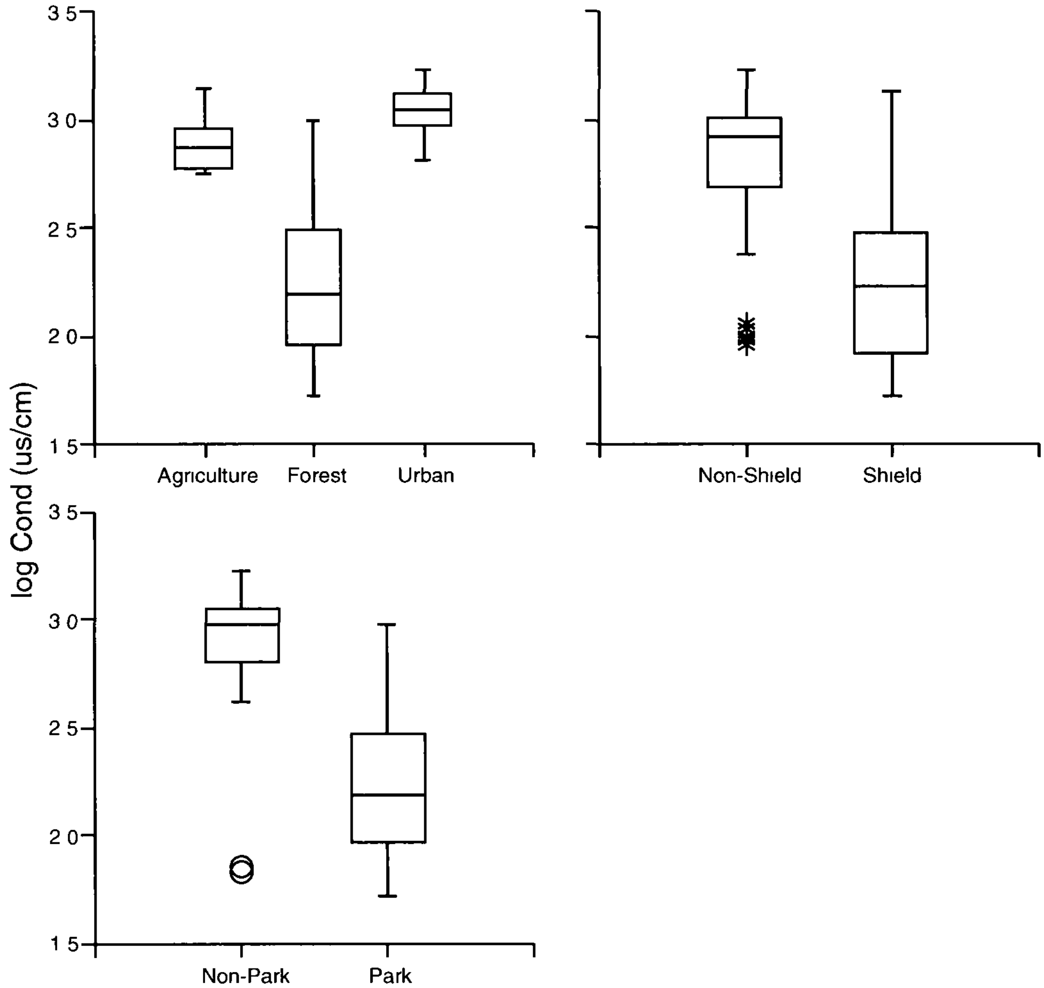

Figure 2.3. Box plots displaying conductivity (μs/cm) values across site locations and dominant land-use types in streams in the Ottawa region.

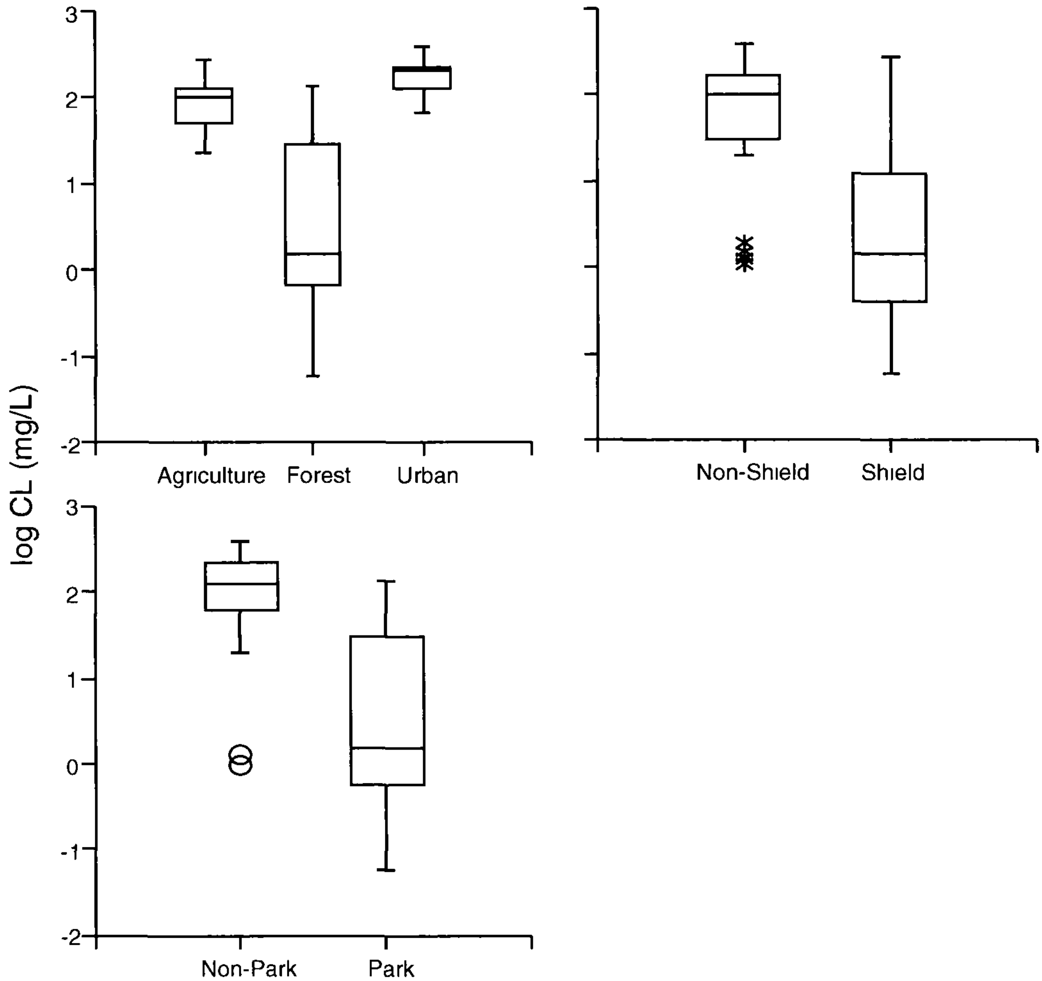

Figure 2.4. Box plots displaying Chloride (mg/L) values across site locations and dominant land-use types in streams in the Ottawa region.

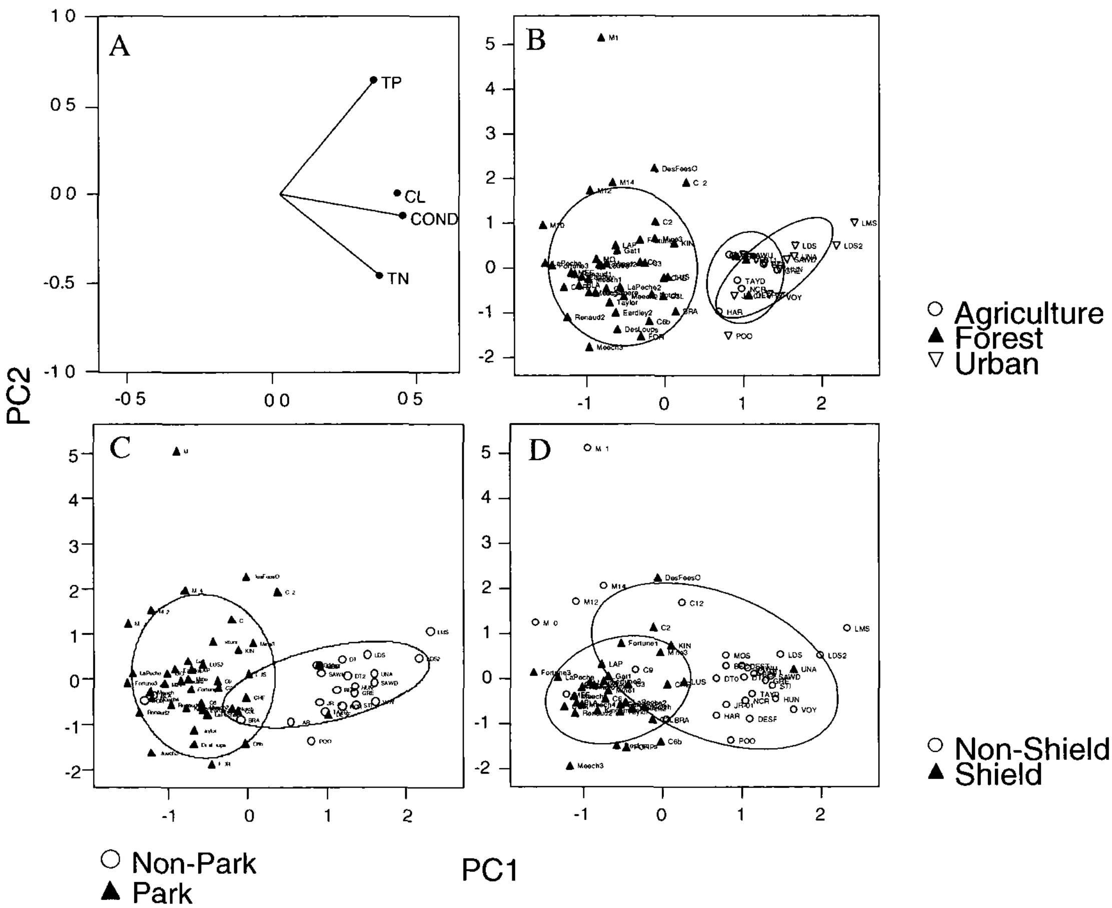

Figure 2.5. Plots of the first two axes of a Principal Component Analysis of environmental variables measured at each of the 67 sample sites. Panel A shows the factor loading plot. Panels B, C, and D display sample sites grouped by land-use, park/non-park, and shield/non-shield respectively.

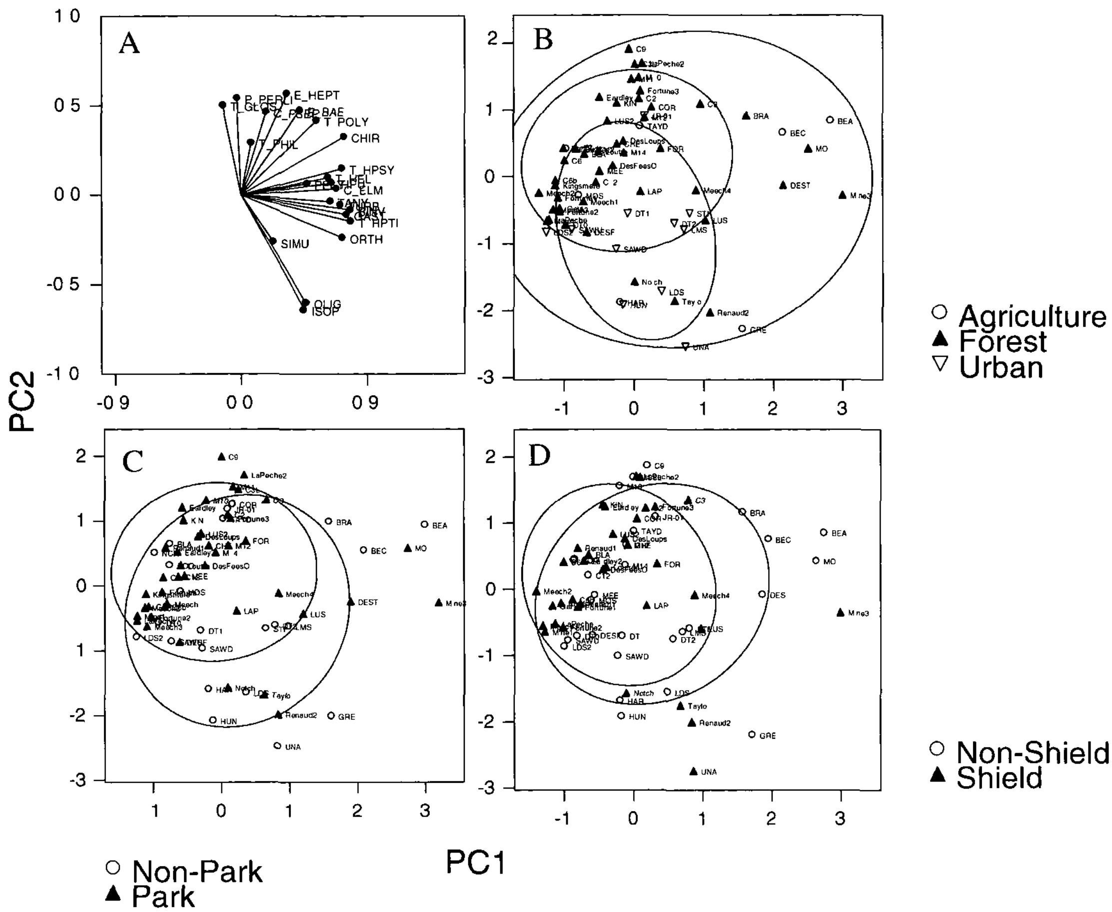

Figure 2.6. Plots of the first two axes of a Principal Component Analysis of dominant invertebrate taxa for 67 sample sites. Only taxa found at >30% of sites were included. Taxa biomass were log transformed (log (x+.001g). Panel A shows the factor loading plot for invertebrate taxa. Panels B, C, and D are display sample sites correlated grouped by land-use, park/non-park, and shield/non-shield respectively.

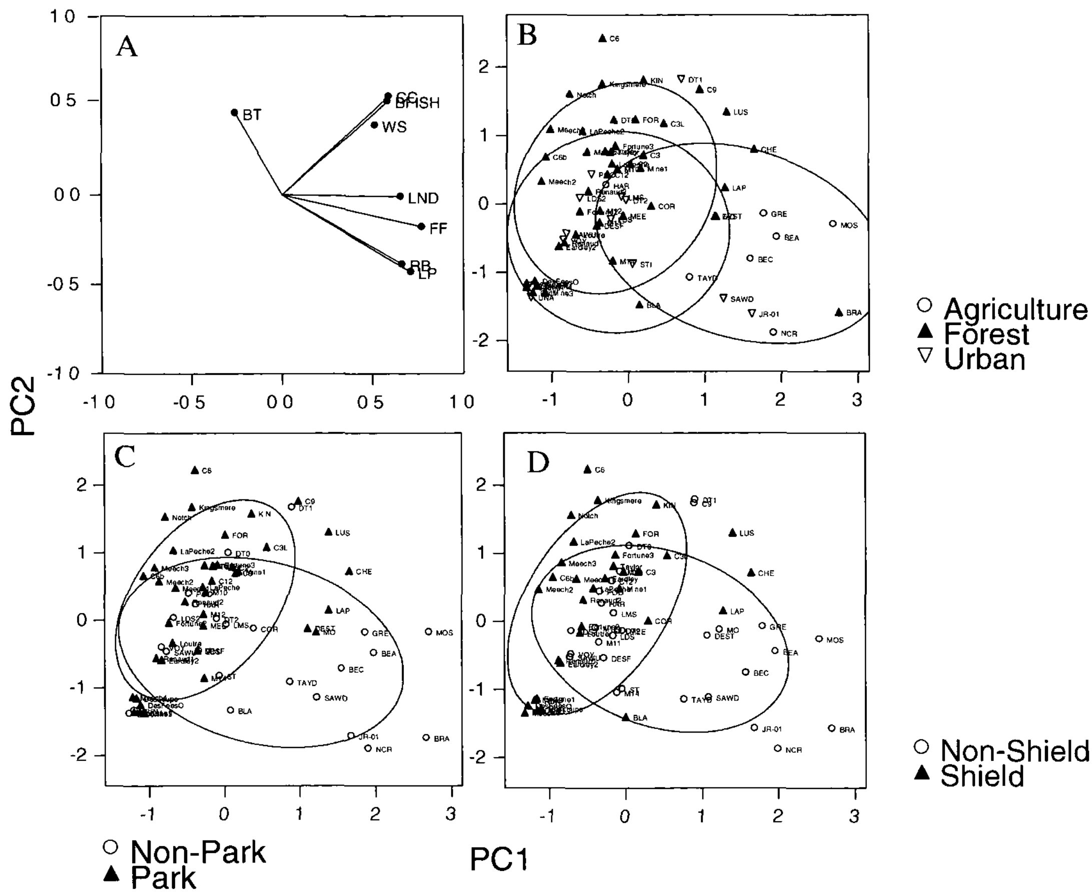

Figure 2.7. Plots of the first two axes of a Principal Component Analysis of dominant fish taxa for 67 sample sites. Only taxa found at >30% of sites were included. Taxa biomass were log transformed (x+.005g). Panel A shows the factor loading plot for fish taxa. Panels B, C, and D display sample sites grouped by land-use, park/non-park, and shield/non-shield respectively.

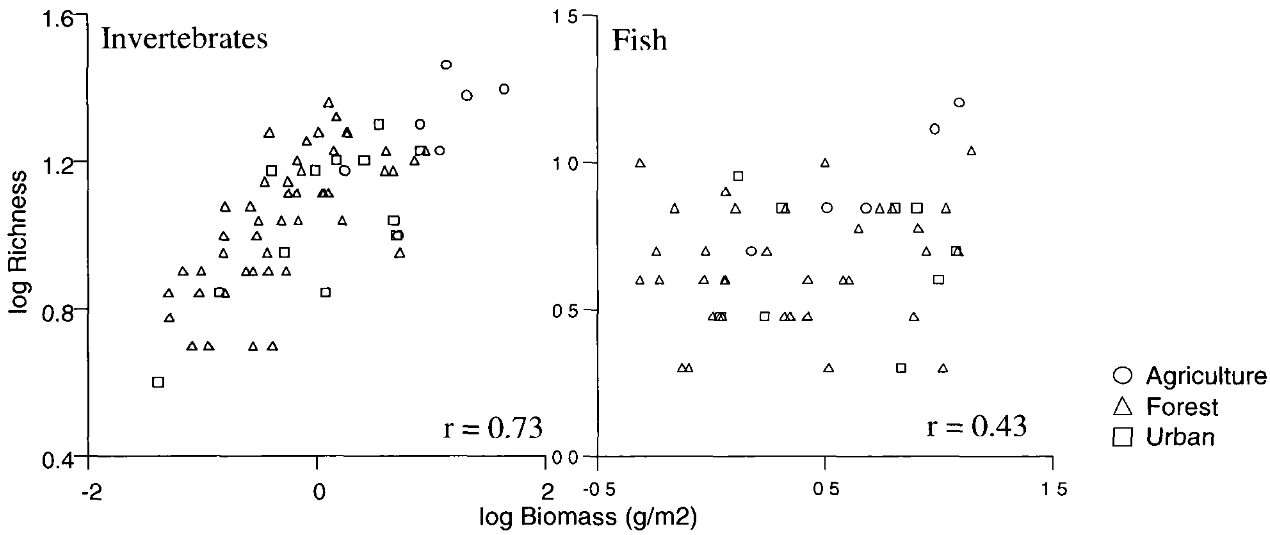

Figure 2.8. Scatter plot showing correlation between richness and biomass for invertebrates and fish for dominant land-use types. Correlation coefficient is shown as r for each relationship.

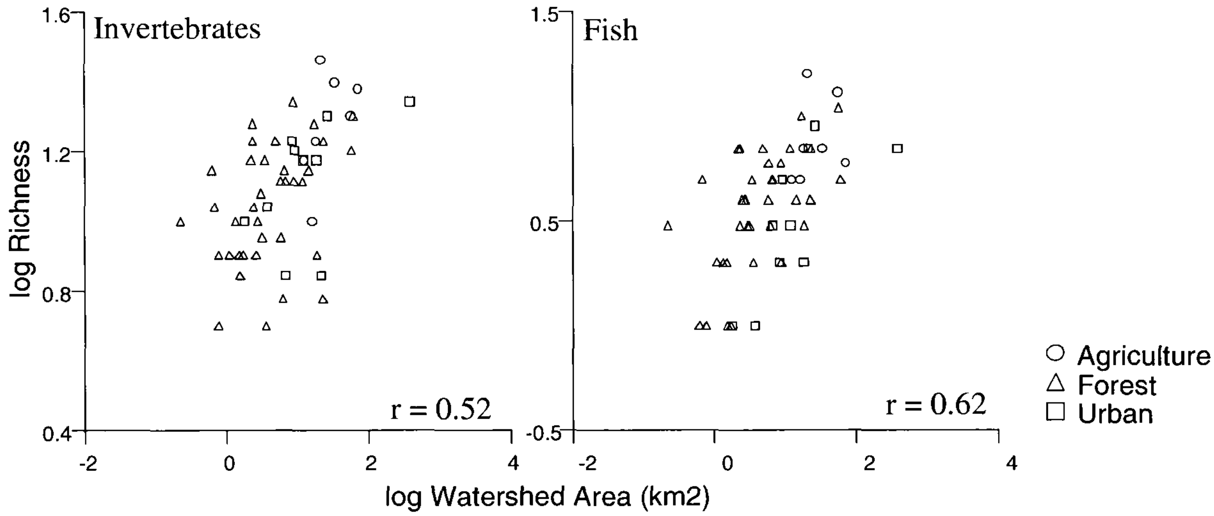

Figure 2.9. Scatter plot comparing invertebrate and fish taxa richness and watershed area. Correlation coefficient is shown as r for each relationship.

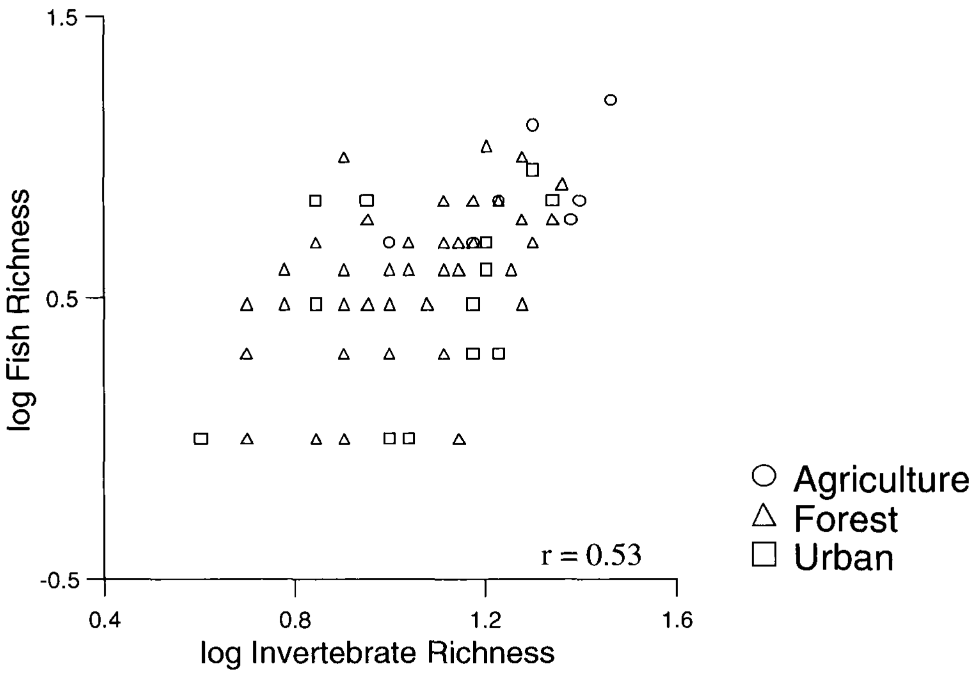

Figure 2.10. Scatter plot displaying the correlation between invertebrate richness and fish richness among dominant land-use types. Correlation coefficient is shown as r.

CHAPTER 3 – STREAM ECOSYSTEM RESPONSE TO INCREASED NUTRIENTS AND PERTUBRATION ALONG AN URBANIZATION AND AGRICULURAL GRADIENT

INTRODUCTION

Retention of nutrients and efficiency of energy transfer between tropic levels is thought to decrease in perturbed ecosystems (Blackburn et al. 1973, de Vries et al. 1998, Jennings and Mackinson 2003, Odum 1985). Anthropogenic eutrophication often leads to major changes in community assemblages when excessive amounts of nutrients are released into aquatic systems, and is often associated with release of toxicants and changes of hydrological regimes related to an increase of impervious surfaces in watersheds. In aquatic systems, it has been widely documented that an increase of nutrients stimulates primary productivity (DeCosta et al. 1983, de Vries et al. 1998, Eloffson et al. 2006, Makarewicz and Likens 1979, van Beusekom et al. 2008) often with concomitant increases in production at higher trophic levels, and decreased overall ecological efficiency (Cannobbio et al. 2009, Dodds et al. 2002, Makarewicz 2009, Miserendino et al. 2008, Walsh et al. 2001). However, comprehensive studies of energetics across multiple trophic levels in streams are sparse (Benke et al. 1988).

Because of the challenges associated with the quantitative estimation of biomass and production at several trophic levels in stream ecosystems, even fewer studies have examined simultaneously more than one or two trophic levels at more than a single site. Consequently, the extent to which ecological or production efficiency depends upon the trophic status of stream ecosystems is largely unknown.

Critical to an accurate estimate of ecological efficiencies, are estimates of production at adjacent trophic levels. Usually, production is estimated as the product of standing stock biomass and instantaneous specific growth rate (Benke 1976). Because biomass can vary by several orders of magnitude whereas variation in growth rate is more constrained, most empirical studies have found that most of the spatial and temporal variance in production can be accounted for by biomass estimates for primary (Morin et al. 1999) or secondary producers (Morin and Dumont 1994). To a first approximation then, biomass can be used as a relative index of production. This index will be particularly appropriate if size spectra of the organisms remain constant at all levels of overall biomass, since size is the main correlate of growth rate (Peters 1983).

The biomass spectrum has been used to summarize complex assemblages and in the simplest definition is the density of organisms across a range in body sizes within an ecosystem (Kerr and Dickie 2001, Sprules et al. 1991). While there is a general tendency towards a log-linear decline in density with body size (Kerr and Dickie, 2001), the invertebrate size spectrum in stream environments is known to be relatively stable across space and time, streams, substrates, seasons, as well as nutrient and productivity gradients (Bourassa and Morin 1995, Cattaneo 1993, Kamenir et al. 2008, Morin 1997, Morin and Nadon 1991). In a study undertaken on the zooplankton size spectra in Lakes Erie and Ontario, it was noted that the stability of the size spectra persisted even during times of great perturbation (reduction in nutrient levels, addition of invasive species, fishing practices, and climate change) possibly because of energetic constraints of planktonic assemblages (Sprules 2008).

Although the shape of the size distribution remains stable other aspects of the spectra have been found to vary. Many studies have found that overall abundance increased along a gradient of enrichment (Boudreau and Dickie 1992, De Eyto and Irvine 2007, Lawton 1990). In addition, the few studies that have examined the magnitude of deviations from linear size spectra, suggest that increased variance is seen in systems with increased anthropogenic perturbation (Sprules and Munawar 1986, Sprules et al. 1988) and that perhaps these deviations from linearity are due to secondary structuring linked to predator prey interactions (Dickie et al. 1987) although both concepts have yet to be explored in stream environments. To my knowledge, this is the first study to assess the residual variability of the expanded stream size spectra.

To date, most studies in stream size distributions have focused on either algal or invertebrate communities (Cattaneo 1993, Morin and Nadon 1991). Rarely have stream size spectra included fish, and even more rarely have they included analysis of multiple trophic levels simultaneously collected. Including multiple trophic levels in descriptions of the spectrum along perturbation gradients could increase our knowledge of how body-size density relationships are structured in pristine and perturbed streams (Knouft 2002).

The current study has two objectives. The first objective is to describe ecological or production efficiency patterns by investigating three trophic levels using a large number of streams existing in watersheds with different land-uses. The second objective of this study is to examine how the size spectrum of benthic invertebrate communities varies along an urbanization gradient paying special attention to deviations from linearity of the size spectra which has never been done in stream environments using the expanded size spectrum.

METHODS

Study area and sample sites

Sixty-seven sites from 45 streams across the National Capital Region (Ottawa, Ontario, Canada) and Gatineau Park were sampled between the summers of 2001 and 2009 for periphyton, benthic macroinvertebrates, and fish to quantify biomass and the size distribution. I sampled twenty-seven of the total 67 sites in 2008. Eleven sites were sampled summer of 2005 (Jennifer Lento, unpublished data), six sites were sampled during the summer of 2003 (Uta Grüenert, unpublished data), and the remaining twenty-three sites during the summers of 2001 and 2002 (Stephenson 2007). Methodology used for sampling periphyton, invertebrates, and fish were similar across all years, as described below.

Periphyton and Macroinvertebrate Sampling Protocol

Six randomly selected cobbles were collected before fish sampling at each site starting downstream and gradually moving upstream throughout the 10 m riffle segment. Each cobble was carefully placed in a whirl pack with a known amount of 95% ethanol (usually between 100-200 mL). In the lab, cobbles were placed in the fridge for 24hrs at which time chlorophyll *a* concentration in the ethanol was estimated spectrophotometerically to obtain an index of periphyton standing stock (Ostrofsky and Rigler 1987). Cobbles were then scrubbed in order to remove all invertebrates from the surface into a sieve of 1 mm mesh size. Macroinvertebrates were placed in sample jars with 95% ethanol to await sorting and measuring. After cobbles were scrubbed and dried, I estimated cobble surface area from the mass of aluminium foil needed to wrap the entire surface of the rock with no overlap.

The macroinvertebrates were sorted and identified to family level using keys from McCafferty (1998). Once sorted, organisms were measured for body length, including appendages, using OPTIMAS 6.5 imaging software (Media Cybernetics, Silver Springs

Maryland). To account for the loss of small macroinvertebrates through the 1mm mesh sieve, a sieve retention model was used to calculate the size specific correction factor (Morin et al. 2004). The model calculates the probability (p) that an organism is retained in a sieve as:

$$\ln(p/(1-p)) = -2.84 + 5.8\log 10(RL) - 3.18\log 10(RL)\log 10(M)$$

where RL is the body length/mesh size and M is the mesh size (mm).

Mass for each individual invertebrate was calculated using length-dry mass regressions developed by Benke et al. (1999). Invertebrate biomass (g DM/m^2) was calculated by dividing the dry mass of organisms collected on each cobble by the surface area of the cobble.

Fish Sampling Protocol

Fish were collected using a Smith-Root LR-24 backpack electrofisher set at 300 V initially and decreased when necessary (if fish/crayfish were dying) during base-flow conditions during the summer of 2001-2003, 2005, and 2008. Sample sites were sectioned off using block nets (3 meters long, 0.4 cm mesh for small streams and 15 meters long, 0.4 cm mesh for larger streams). Stream surface area was estimated by multiplying the length of the sample site (10 m) by the average width of the stream (estimated from width measurements taken at five transects of the stream). Estimates of fish biomass were by the attrition method: repeated passes were made until the numbers of fish captured in a pass dropped below 50% of the initial pass. As fish were shocked, they were captured using dip nets and transferred into buckets, or when missed, were washed downstream into the furthest downstream block net. Fish were identified to species level and measured for fork length, total length, and body depth (mm) after each pass using a standard ruler and clipboard. Once fish were processed, they were released downstream of the sample site as to not be included in the next pass. After all three passes were completed the furthest downstream net was checked for fish. Any fish collected from the net were processed in the same manner as described above and were counted as being caught in the first pass.

Fish drymass was estimated for each individual using a length-weight regression from Randall and Minns (2000) or Schneider et al. (2000) and individuals were grouped into $\log_2$ size classes. Fish data were summed across species-size class for each pass and site. I then generated the population size at each site using Microfish 3.0 demonstration version (Van Deventer 1989) from maximum likelihood estimates of abundance based on the number of individuals collected and the reduction in fish counts with each additional pass. These population estimates were multiplied by the observed averaged drymass for each taxon-size class combination and then divided by the area of the sampling location to obtain the corrected biomass estimate (in g DM/m^2) for each species-size class combination in each stream.

Normalized Size Spectra

To create normalized size spectrum, all organisms were grouped into $\log_2$ size classes. Normalized biomass values were calculated by dividing the biomass in each size class by the width of that size class (for $\log_2$ size class intervals, the width of a size class is the lower limit of the size class interval). Normalized size spectra for all organisms were grouped and plotted as $\log_{10}$ normalized biomass against $\log_{10}$ mass. Where there were multiple years of sampling for the same site, averaged spectra were used.

Water Quality

Water quality was measured at each site two to three times throughout the summers of 2001-2003, 2005, and 2008. Total phosphorus (TP, mg/L), nitrite, nitrate, and total kjehldahl nitrogen were determined by ROPEC laboratories (Ottawa, Ontario, Canada). Total nitrogen (TN, mg/L) was calculated by summing nitrite, nitrate, and total kjehldahl nitrogen for each site and sampling date. Total nitrogen and total phosphorus were averaged across the season and the resultant average values were used in statistical models.

Site Type Designation

In order to compare size spectra, sites were placed into categories corresponding to whether they were located in, or outside the boundaries of Gatineau Park, whether they drained watersheds dominated by one of three land types (forest, agriculture, or urbanized land), and whether they were located on the Canadian Shield. Google Earth® was used to assess the dominant land-use type by visually inspecting the land within each watershed boundary (2010). An estimation was made as to whether watersheds containing each sample site were classified as mostly agricultural, mostly urbanized, or mostly forested land-use. A digital map of the National Capital Region (Natural Resources Canada, Geological Survey of Canada, 2003) displaying areas designated as Canadian Shield was used in conjunction with Google Earth® to determine whether the watershed was located on the Canadian Shield.

Statistical Analysis

General Linear Model

General Linear Models of the form:

$$\log_{10}BM = a + b(\log_{10}TP) + b(\text{trophic level}) + c(\log_{10}TP) * (\text{trophic level}) \quad (1)$$

and

$$\log_{10}BM = a + b(\log_{10}TN) + b(\text{trophic level}) + c(\log_{10}TN) * (\text{trophic level}) \quad (2)$$

were fitted using Systat v.12. In these models, BM represents the biomass (g DM/m^2) at a given trophic level for each site, TN or TP is the average total nitrogen or average total phosphorus (mg/L) for each site, trophic level is either A (for algae), I (for invertebrates) or F (for fish), *a* is the intercept, and *b* and *c* are the coefficients describing the slope.

Simple Linear Regressions

Regression models were fitted to find the slopes of the relationship between biomass of adjacent trophic levels using the models:

$$\log_{10}BM_{\iota} = a + b(\log_{10}BM_{\iota-1}) \quad (4)$$

where BM_{ι} is biomass (g DM/m^2) of either periphyton, invertebrates, or fish, *a* is the intercept, and *b* is the coefficient describing the slope.

Mixed Model

Statistical comparison of size spectra among sites, as a function of nutrient concentration, and among site categories were made using mixed effects modeling techniques as implemented in the nlme R package (Pinheiro et al. 2009) following the model selection approach advocated by Zuur et al. (2009). The log normalized biomass in each i size class in each j site was the dependent variable. A saturated model including the fixed effect of log body mass, log nutrient concentration, and interactions was first fitted tentatively to allow selection of the appropriate random terms represented by random effects intercepts and slopes and appropriate variance structure for the residuals. Intermediate models were of the form:

$$\text{Log (normalized biomass)}_{ij} = \alpha + b_j + \beta \log(M) + c_j (\log M) + \beta 2 \log(\text{nutrient}) + \beta 3 \log(M) \log(\text{nutrient}) + \varepsilon_{ij} \quad (5)$$

where $\varepsilon_{ij} \sim N(0,\sigma^2_k)$.

In these models, α represents the average intercept; β, $\beta 2$, $\beta 3$ represent the fixed effect coefficients for body mass (M), nutrient, and the body mass:nutrient interaction (or a linear change of the size spectrum slope with a change in nutrient concentration) respectively, b_j represents the random intercept and c_j the random slope of the size spectra, and σ^2_k allowed residual variance to vary among the k groups of sites. As the $\beta 3$ coefficient was not significantly different from 0 in all fitted model (p>0.2), this term was dropped from the final models.

RESULTS

Nutrient concentration and biomass of algae, invertebrates, and fish were higher outside the park in sites draining watersheds predominately containing agricultural land located in the Ottawa Valley. Total phosphorus and total nitrogen values ranged between 0.05 and 0.6 mg/L and between 0.015 and 5.23 mg/L respectively among sites. Sites located outside of the Park had higher nutrient values than sites within the Park. Sites draining watersheds predominately containing agricultural land generally had higher TN and TP values than sites that drained watersheds dominated by urbanized land, but both had higher nutrient values than sites draining watersheds dominated by forested land. Chlorophyll a varied between 0.90 and 292.99 g Chla/m^2, while invertebrate and fish biomass varied between 0.04 and 43.65 g DM/m^2 and between 0 and 13.84 g DM/m^2 respectively among sites.

Slopes of the relationship between biomass and nutrients (TN or TP) were significantly different among trophic levels for TN, but not for TP (Table 3.1.). Slopes were steeper for algae, and decreased as trophic level increased (shown separately for sites designated as park/non-park, shield/non-shield, and agricultural/urban/forest) (Figs. 3.1 a-c, Table 3.2.). In addition, the slope of the relationship between periphyton and invertebrates was visibly steeper than for the relationship between invertebrates and fish (Figure 3.2.) indicating increasingly weak relationships between a given trophic level and nutrients as trophic level increases. 95% confidence intervals from regressions for the two plots overlap (Lower $\text{Limit}_{\text{Inv/Peri}} = 0.245$, Upper $\text{Limit}_{\text{Inv/Peri}} = 0.888$, Lower $\text{Limit}_{\text{Fish/Inv}} = -0.104$, Upper $\text{Limit}_{\text{Fish/Inv}} = 0.327$) suggesting that slopes are not significantly different.

Slopes of normalized biomass spectra were remarkably similar across sites (Figures. 3.3. and 3.4.), however intercepts varied (Figure 3.4.). Normalized biomass increased with increasing nutrient concentration (□2 coefficient in Eq. 5 was positive), but this increase was the same across all size classes (□3 coefficient in Eq. 5 was not significantly different from 0) indicating that the average size did not vary with changes in nutrient concentration. Concentrations of TN and TP were strongly and positively correlated among sites. Models using TN or TP were qualitatively similar although TN had a slightly better fit and only these models were presented in Table 3.3. Inclusion of both TN and TP terms in the model did not improve the fit significantly.

Size spectra of sites located in less perturbed areas (sites within Gatineau Park, or draining watersheds dominated by forested land) showed less residual variance than sites located in more urban areas (Table 2.3).The standard deviation of the residuals for the relationship between log normalized biomass (g DM/m^2) and log body size within Gatineau Park was 75% of the standard deviation of the residuals in non-park sites. In addition, the standard deviation of the residuals for the same relationship as above in sites draining watersheds dominated by forested land, were 64% of those draining agricultural land. Sites draining watersheds dominated by agricultural or urbanized land were similar in that the standard deviation of the residuals was 93% of agricultural for urbanized sites.

DISCUSSION

In small streams within the National Capital Region, nutrient concentration is higher in urbanized and agricultural watersheds. Streams in watersheds predominately containing agricultural or urbanized land-use had higher total nitrogen and total phosphorus values than streams draining watersheds dominated by forest. Agricultural watersheds had the highest nutrient values of all three land-use types. Nutrient levels were also found to be higher off the Canadian Shield. However, our study does not allow us to distinguish the effects of land-use and geology because sites located within the relatively pristine forested region of Gatineau Park are also located on the Canadian Shield, whereas most agricultural and urban sites are located south of the shield in the in the clay plains of the former Champlain Sea. This spatial correlation of geology and human activity is not unusual; for example, in a study undertaken in Oslo, Norway, authors found that the task of partitioning decreased water quality to either land-use or geology was difficult due their strong spatial correlation (Reimann et al. 2009). Finally, we found two distinct trends in the relationship between biomass and nutrients in National Capital Region streams. First, slopes were less than 1, indicating that the nutrient assimilation efficiency declines as nutrient levels increase. Second, slopes decreased with increasing trophic level.

Land-use within a watershed can strongly influence the pollutant input into streams (Chang et al. 2008, Hepp and Santos 2009, Nash et al. 2009), however the mechanism driving decreased water quality in agricultural or urban watersheds differ. In the former, increased nutrient input occurs from grazing cattle or from fertilizer spread on crops (Nash et al. 2009). Moreover, grazing cattle alter stream-banks by reducing vegetation and causing erosion (Nash et al. 2009). Both forms of stream-bank alteration can lead to increased sediment and nutrient inputs in streams located adjacent or within watersheds dominated by agricultural land-use.

By contrast, decreased water quality in urban settings usually occurs due to increased imperviousness, which results in reduced infiltration and greater runoff. At times of high rainfall, this modified hydrology causes large flood-like situations to occur which inputs large amounts of water into streams causing nutrient levels and conductivity to change dramatically (Brabec et al. 2002, Canobbio et al. 2008, Paul and Meyer 2001, Walsh et al. 2005). Urbanization leads to both higher nitrogen and phosphorus levels, sometimes exceeding those in agricultural settings (Paul and Meyer 2001).

In addition to land-use, surficial geology influences total nitrogen and total phosphorus levels in the National Capital Region. However, it is difficult to distinguish the effects of land-use and geology because sites located within the relatively pristine forested region of Gatineau Park are also located on the Canadian Shield, since these two factors are highly correlated. All but three sites denoted as being located on the Canadian Shield, are also located within Gatineau Park. In contrast, about half of the sites denoted as not being located on the Shield, are non-park sites, and half are Park sites. In order to properly assess the relative contribution of land-use and geology, it would be necessary to control for the factors of geology or land-use. For example, if all sites were located in the Canadian Shield, but varied in terms of land-use within the watershed that the sample site drained, it would be clear whether land-use alone influenced water quality or fish and invertebrate assemblage composition. By contrast, to find out if geology influences the same factors, it would be essential to sample streams both on and off the Canadian Shield that are drained by watersheds containing forested, agricultural, or urbanized land-use. Although this question was not directly addressed in the current study, this could be an avenue of interest for future researchers.

A relationship between log biomass and log nutrient concentration with a slope of less than one is indicative of decreased ecological or production efficiency with increasing nutrient levels. Internal cycling of nutrients suffers with increasing human disturbance, creating a system where nutrient inputs do not translate into increased biomass (Odum 1985). Several studies have found a decrease of efficiency in systems at higher nutrient levels (Blackburn et al. 1973, de Vries et al. 1998, Jennings and Mackinson 2003, Odum 1985) but a few have found the opposite to be true (Lacroix et al. 1999, Makarewicz and Likens 1979). These contradictory results appear to be evidence of uncertainty surrounding the topic of ecological efficiency however, when the individual studies are examined more closely, problems are encountered. In the case of Lacroix et al. (1999) consistency in slopes between shallow and deep lakes along a gradient of eutrophication were found but slope values were quite dissimilar. Perhaps with a larger sample size, significant differences in slopes may have been found. Makarewicz et al. (1979) also found constant efficiency of the transfer of energy between algae and zooplankton in lakes, however erroneous data analysis was the cause of this result (MacCauley and Downing 1985). The results from my study provide support for the many studies in the past that found ecological efficiency to decrease with increasing insults to ecosystems, and supplements this previous research with additional knowledge of stream ecosystems.

Nutrient levels substantially affect the biomass of organisms in streams. Allen et al. (2006) noted the same pattern and supported it with the textural discontinuity hypothesis which predicted that although the abundance and possibly identities of species may be altered by perturbation the overall pattern (or slope of the size spectrum) will remain stable. Although several studies have found that there was an increase in biomass of organisms with an increase in nutrients or productivity (De Eyto and Irvine 2007, Lawton 1990, Perez-Ruzafa et al. 2002), few studies have examined the size spectrum. The exception is Bourdreau and Dickie's (1992) paper examining the biomass spectrum for invertebrates and fish, however data were extracted and compiled from multiple sources and were not collected from a single sampling region. Even so, Boudreau and Dickie (1992) still found biomass varied positively with nutrient availability. A prior study in streams in the Ottawa-Hull area reported increased macroinvertebrate abundance and average size in streams with higher nutrient concentration (Bourassa and Morin 1995). This pattern was also seen in a study undertaken on fish in the St. Lawrence River (deBruyn et al. 2002) and in planktonic assemblages (Sprules and Munawar 1986).

The current study was the first to examine the effect of nutrients on residual variation in the full size spectra including benthic macroinvertebrates and fish in stream environments. A low amount of variation in residuals suggests tight coupling between producers and consumers (Sprules and Munawar 1986) possibly due to limiting factors (Peterson et al. 1983). In low level systems primary productivity is limited by nutrients and thus a strong relationship exists between the two factors (Peterson et al. 1983). As nutrient concentration increases and assemblages are no longer limited by their abundance, other factors such as grazing become more important at limiting primary productivity (Neckles et al. 1993). In essence systems switch from bottom-up to top-down control as nutrients become less important in limiting their biomass but we do not have proof of this in the current study (Lapointe 1997).

From the current study it is clear that efficiency decreases with increasing nutrients in small streams in the Ottawa Region. This detrimental effect of nutrients on efficiency of energy transfer between trophic levels could be seen downstream as nutrients are transported hydrologically to larger rivers, lakes, and eventually the ocean. Although the slope of the size spectra remains stable among land-use types, there is increased biomass in more urbanized and agricultural watersheds. Moreover, increased residual variability is seen in perturbed watersheds. All of these findings point to the same end: that size can be useful in our understanding of how streams are affected by agricultural and urbanized land-use. Although the size spectra slope remains stable in perturbed watersheds, if we take a closer look, other aspects are altered. Since size is so easy to measure and the size spectra are so easy to calculate, it may be beneficial to incorporate a size based approach in future monitoring efforts of streams in perturbed watersheds.

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TABLES AND FIGURES

Table 3.1. Summary of General Linear Models to test for differences in slopes of the relationship between biomass of periphyton, invertebrates, and fish and average total nitrogen or average total phosphorus concentration for 67 sites in the Gatineau Park and National Capital Region showing the coefficients for each variable term and the interaction term and associated standard error (SE). A significant interaction p-value is indicative of a significant difference in slopes ($\alpha = 0.05$). Residual mean square (RMS), and R^2 for models are also shown. Full models are in the form $\log_{10}BM = a + b(\log_{10}Nuts) + c(\text{trophic level}) + (\log_{10}Nuts) * (\text{trophic level})$, where BM is the organism biomass per site measured in g DM/m^2 for invertebrates and fish, or for periphyton, measured in g $Chla/m^2$, Nuts is either total nitrogen or total phosphorus, and trophic status is denoted as either algae, invertebrates, or fish, *a* is the intercept, and *b* and *c* are coefficients describing the slope. Coefficients and standard errors for each level of the categorical variable were obtained from separate regressions in the form $\log_{10}BM = a + b(\log_{10}Nuts)$.

Log B = Log10TP + trophic level + interaction					
Term	Coefficient	SE	p-value	RMS	R^2
TP	10.80	4.13	0.00	0.38	0.69
Trophic level					
Algae	0.59	0.14			
Invertebrates	0.43	0.20			
Fish	0.00	0.20			
TP:Trophic level	2.91	1.12	0.06		
LogB = Log10TN + trophic level + interaction					
Term	Coefficient	SE	p-value	RMS	R2
TN	19.44	7.04	0.00	0.36	0.50
Trophic level					
Algae	0.79	0.11			
Invertebrates	0.41	0.19			
Fish	0.01	0.19			
TN:Trophic level	4.75	1.72	0.01		

Table 3.2. Intercept, regression coefficients, associated standard errors (SE), residual mean square (RMS), and R^2 value for linear regressions from the relationship between biomass of periphyton, invertebrates, and fish and average total phosphorus or average total nitrogen. Each linear regression fits the model $\log_{10}BM = a + b(\log_{10}Nuts)$, where BM is the biomass of either periphyton, invertebrates, or fish, and Nuts is either total nitrogen or total phosphorus, a is the intercept and b is the coefficient describing the slope.

	T P						T N					
	Intercept	SE	Slope	SE	R2	RMS	Intercept	SE	Slope	SE	R2	RMS
Periphyton	2.1	0.22	0.6	0.14	0.21	0.49	1.4	0.06	0.8	0.12	0.42	0.41
Invertebrates	0.59	0.31	0.43	0.20	0.07	0.67	0.04	0.09	0.41	0.19	0.07	0.69
Fish	0.14	0.31	0.00	0.20	0.00	0.68	0.16	0.10	0.07	0.19	0.00	0.68

Table 3.3. Summary of final mixed effects models fitted to normalized biomass spectra for 67 sites. Each model was fitted in the form: $\log(\text{normalized biomass})_{ij} = \alpha + b_j + \beta \log(M) + c_j (\log M) + \beta 2 \log(\text{nutrient}) + \beta 3 \log(M) \log(\text{nutrient}) + \varepsilon_{ij}$, where $\varepsilon_{ij} \sim N(0,\sigma^2_k)$, α represents the average intercept; β, $\beta 2$, $\beta 3$ represent the fixed effect coefficients for body mass (M), nutrient, and the body mass:nutrient interaction (or a linear change of the size spectrum slope with a change in nutrient concentration) respectively, b_j represents the random intercept and c_j the random slope of the size spectra, and σ^2_k allowed residual variance to vary among the k groups of sites.

Fixed effects					Fixed effects					Fixed effects				
	Coefficient	SE	df	p		Coefficient	SE	df	p		Coefficient	SE	df	p
Intercept	3.68	0.17			Intercept	3.69	0.17			Intercept	3.69	0.17		
logM	-0.63	0.01	989	<0.0001	logM	-0.63	0.01	989	<0.0001	logM	-0.63	0.01	989	<0.0001
logTN	0.18	0.10	53	0.07	logTN	0.18	0.10	65	0.07	logTN	0.18	0.10	65	0.07
Random effects					Random effects					Random effects				
	SD					SD					SD			
Intercept	0.39				Intercept	0.39				Intercept	0.39			
logM	0.05				logM	0.05				logM	0.05			
Residual	0.50				Residual					Residual				
					Park	0.45				Agricultural	0.57			
					Non-Park	0.55				Forest	0.44			
										Urban	0.65			
AIC	1675.35				AIC	1652.02				AIC	1865.62			

Park/non-Park

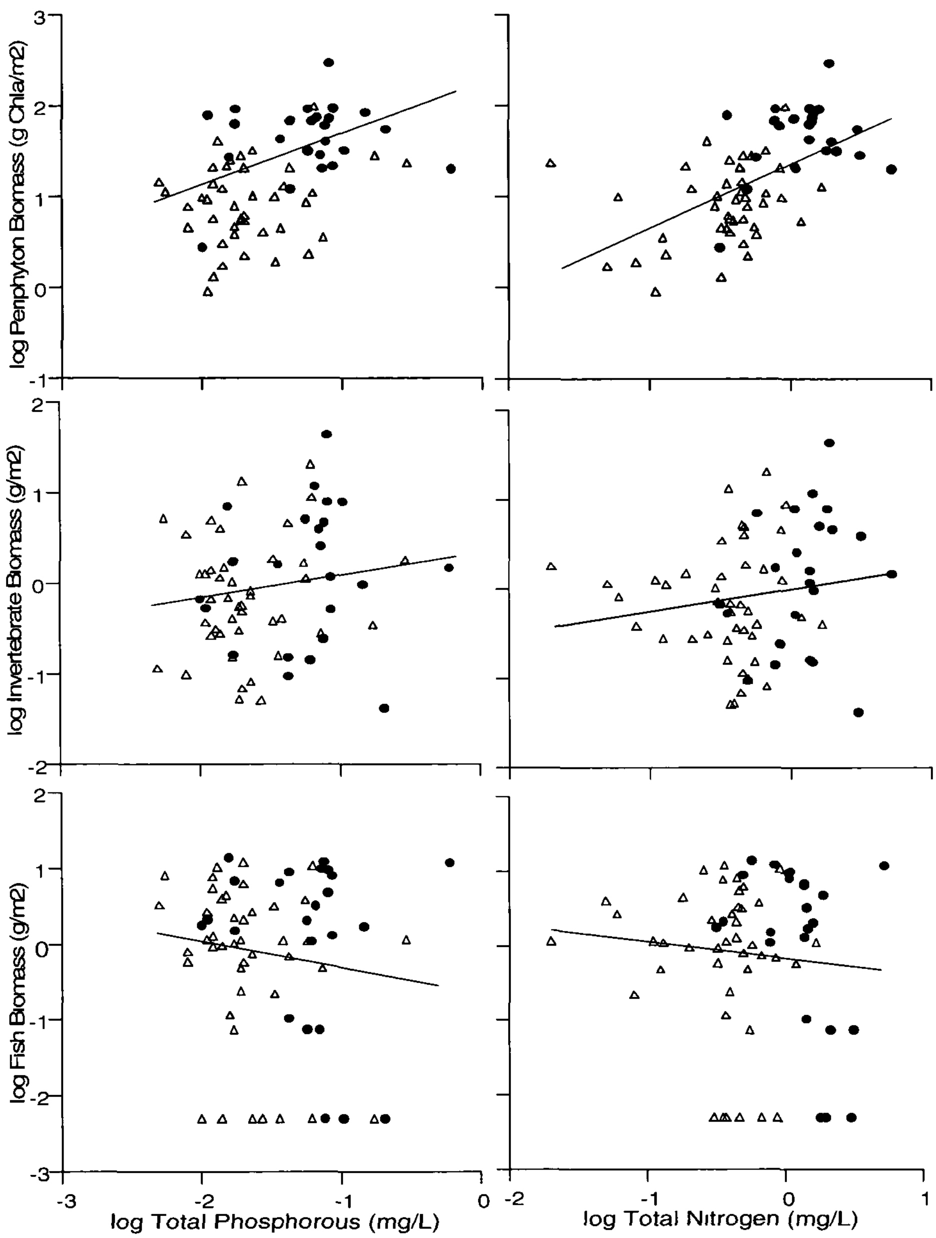

Figure 3.1a. Scatter plots indicating the relationship between periphyton, invertebrate, and fish biomass (g DM/m^2) and water quality parameters (total nitrogen and total phosphorus (mg/L)) shown for site location types (triangles = park, circles = non-park). Smoother for each plot is the linear regression line for each respective relationship.

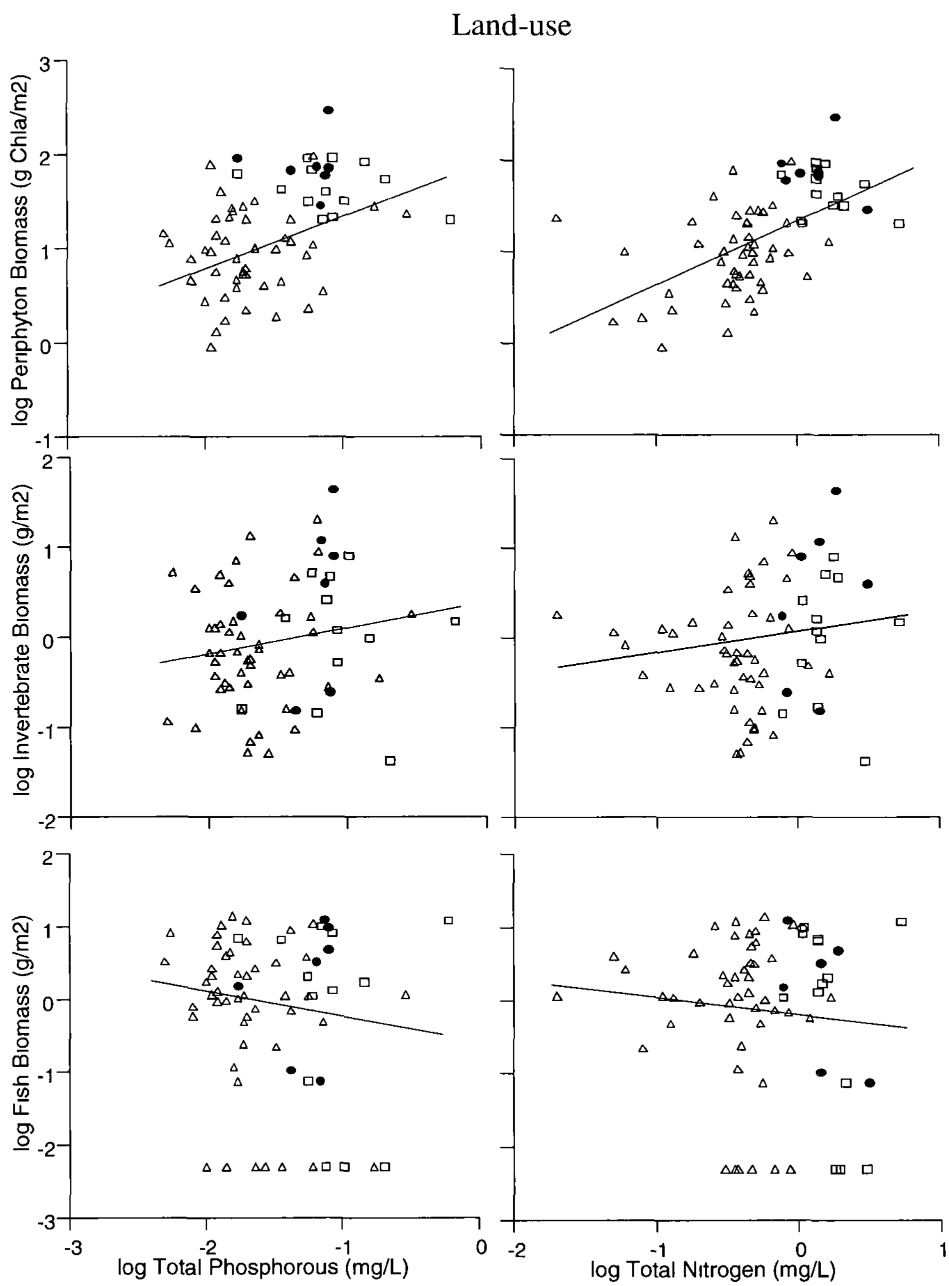

Figure 3.1b. Scatter plots indicating the relationship between periphyton, invertebrate, and fish biomass (g DM/m^2) and water quality parameters (total nitrogen and total thosphorous (mg/L)) shown for site location types (circles = agriculture, squares = urban, triangles = forest). Smoother for each plot is the linear regression line for each respective relationship.

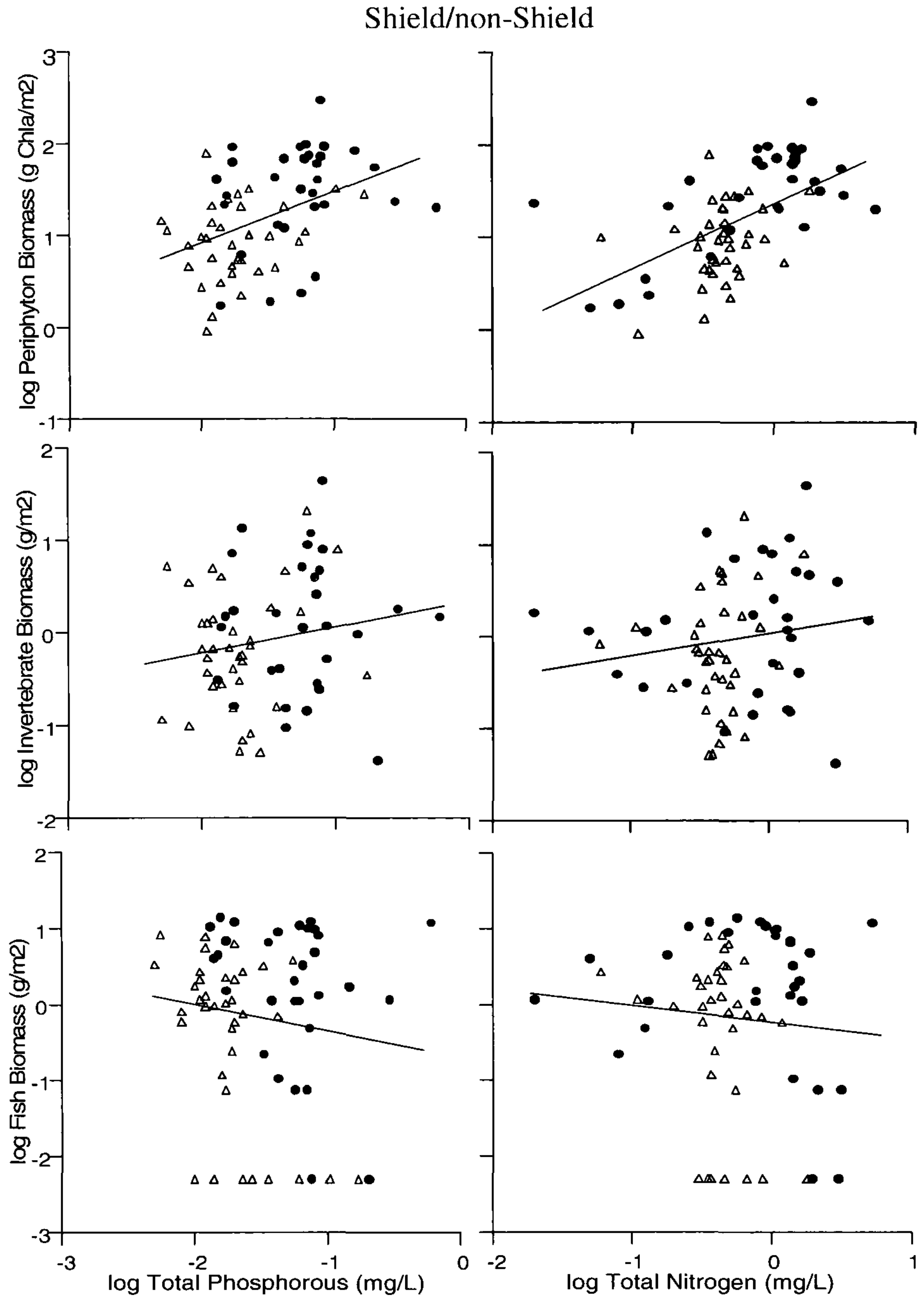

Figure 3.1c. Scatter plots indicating the relationship between periphyton, invertebrate, and fish biomass (g DM/m^2) and water quality parameters (total nitrogen and total phosphorus (mg/L)) shown for site location types (triangles = shield, circles =non-shield). Smoother for each plot is the linear regression line for each respective relationship.

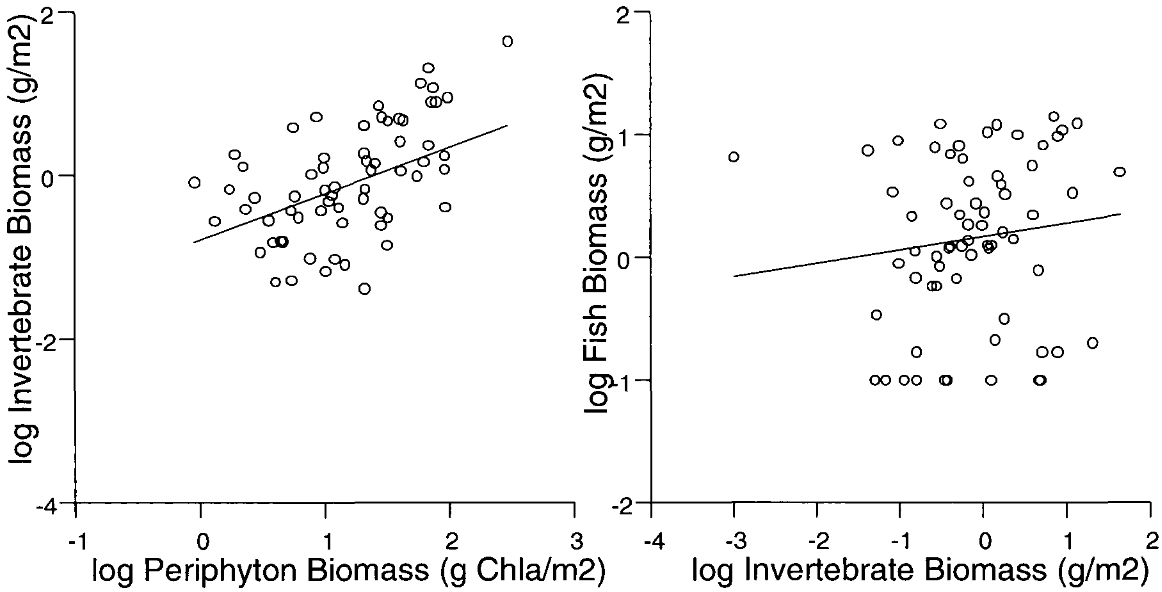

Figure 3.2. Scatter plot illustrating the relationship of organism biomass between trophic levels across sites.

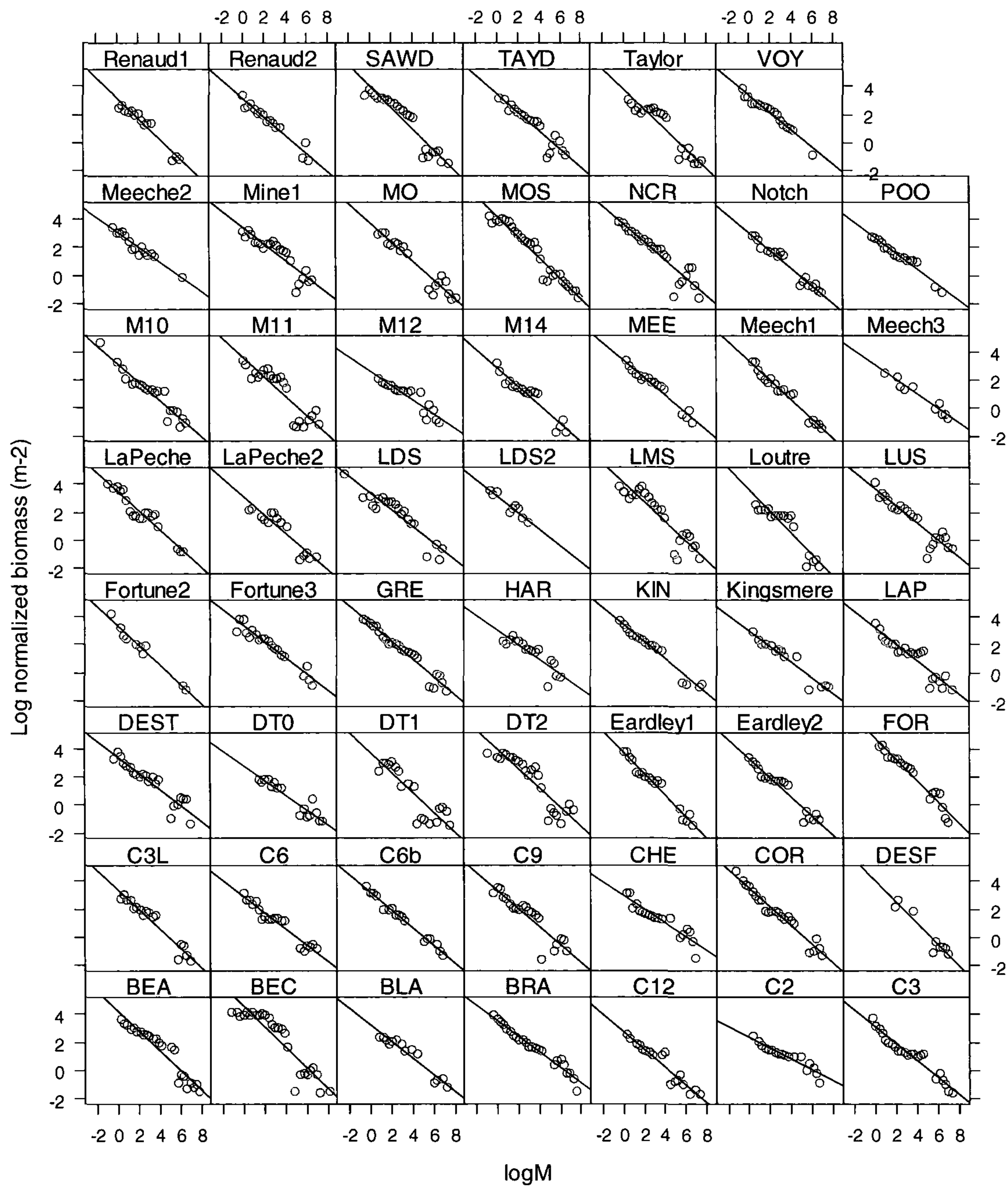

Figure 3.3. Normalized biomass spectra for the 55 sampled sites containing both fish and macroinvertebrates. LogM is the log of upper limit of organism drymass within each size class. The lines are the simple linear regressions fitted separately for each site. Note that the 3 cases where the slope is positive are missing macroinvertebrate data.

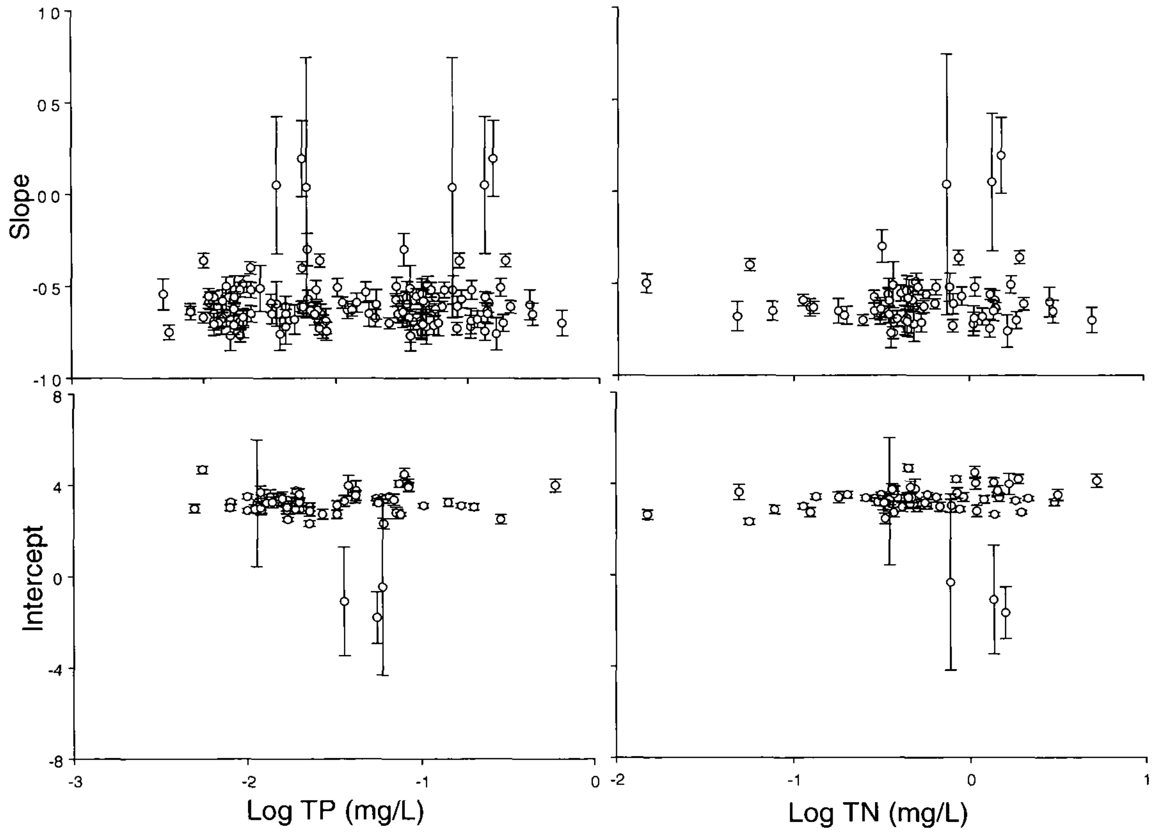

Figure 3.4. Scatter plots displaying the relationship between regression slope and intercept (± SE) of the regression line for each of the 67 normalized size spectra and total phosphorus and total nitrogen (mg/L). Regression models were in the form: $\log_{10}D = a + b(\log_{10}M)$, where D is the normalized biomass (g/m^2), M is the upper limit of organism drymass within each size class, a is the intercept, and b is the coefficient describing the slope.

CHAPTER 4 – GENERAL CONCLUSIONS

In light of the Gatineau Park Ecosystem Conservation Plan (NCC, 2008), the current study has provided important baseline information on water quality, benthic invertebrate communities, and fish community structure for a number of stream systems in Gatineau Park. This information could be used in the future to supplement existing biological information and could serve as the basis on which the design and implementation of a systematic stream surveillance program is developed, as identified in the Gatineau Park Ecosystem Conservation Plan. Moreover, the current study has confirmed the previously observed pattern of declining efficiency with increasing nutrients in small streams. Few stream studies have attempted the measurement of the expanded size spectra including algae, invertebrates, and fish. The more common use of the stream biomass spectrum is focused on either algal or invertebrate communities separately (Cattaneo 1993, Morin and Nadon 1991). The stability of the size spectrum slope remains among land-use types (park/non-park, forest/agricultural/urban, and shield/non-shield) but increased biomass in more urbanized and agricultural watersheds is found. This study is the first to my knowledge, to examine the difference in residual variability of the size spectra between perturbed and protected stream sample sites and increased residual variability in perturbed watersheds is noted (non-park, agricultural, urban).

Principal Component Analysis revealed that water quality parameters differed more between groups (park/non-park, forest/agricultural/urban, and shield/non-shield) than macroinvertebrate or fish composition. Water quality parameters, specifically nutrient inputs have been shown to differ greatly between urbanized and pristine regions. Processes such as rain-fall, sediment input, and weathering alter water quality in natural streams whereas agricultural and industrial practices and urbanization drastically influence water quality in anthropogenically altered streams (Qadir et al. 2008, Yayintas et al. 2007). Nutrient inputs from fertilizer usage cause an increase in phosphorus and nitrogen concentrations in agricultural watersheds. (Nash et al. 2009). Increased impervious cover and the addition of storm sewers and road salting in urbanized watersheds have been noted to cause an increase in nutrients, conductivity, and chloride concentrations (Brabec et al. 2002, Canobbio et al. 2008, Paul and Meyer 2001, Ramakrishna and Viraraghavan 2005, Walsh et al. 2005).

Although water quality parameters differed more between site types than macroinvertebrate and fish composition, fish were marginally better at separating sites than were macroinvertebrates. Somewhat tighter groupings of taxa in protected sites were seen for macroinvertebrates possibly indicating that perturbed areas have more varied assemblages. The lack of trend seen in invertebrates does not match the predicted results reported in previous studies which show that macroinvertebrate assemblages respond primarily to local conditions (Brazner et al. 2005; Richards et al. 1996; Sponseller et al. 2001). However, the scatter of macroinvertebrate assemblages may be due to a grouping of sites that were considered to be within the borders of Gatineau Park, but were located in more highly impacted areas bordering the Park. This grouping of sites tended to behave more like urban than Park sites in terms of assemblage composition. These sites may have obtained nutrient and sediment inputs from the urban centres directly associated with them which in turn influenced more tolerant species presence in those streams. The separation of sites based on fish taxa biomass which is marginally better than for macroinvertebrates appeared to be driven by the presence/absence of brook trout. Overall, macroinvertebrate and fish assemblage composition did not differ substantially between site types, however this result may have been found due to a number of forested, Park sites having similar assemblage composition to urbanized sites.

Several studies have found a decrease of efficiency in systems at higher nutrient levels (Blackburn et al. 1973, de Vries et al. 1998, Jennings and Mackinson 2003, Odum 1985). The results from my study provides support for the concept of decreasing efficiency with increasing nutrient inputs to a system and supplement previous research with additional knowledge of stream ecosystems. Since nutrients can be transferred for long distances downstream from the source of input, negative change manifesting as decreased biomass as trophic level increases is plausible throughout stream reaches and watersheds when a system is perturbed.

Biomass and normalized density of organisms increased with nutrients and varied among site categories with the exception of fish across dominant land-use types. Several studies have found the same trend in biomass of organisms with increasing nutrients or productivity (Allen et al. 2006, De Eyto and Irvine 2007, Lawton 1990, Perez-Ruzafa et al. 2002) but few studies have examined this same trend in the normalized density of the size spectum. The few that have, noted similar trends as seen in my study for invertebrates and fish sampled across multiple regions (Boudreau and Dickie 1992), for macroinvertebrates collected from streams in the Ottawa-Hull area (Bourassa and Morin 1995), for fish in the St. Lawrence River (deBruyn et al. 2002) and in planktonic assemblages (Sprules and Munawar 1986). A different response of small and large organisms to nutrient increase made the size spectra less steep in enriched sites (deBruyn et al. 2002), however this result was not found in the current study. Slopes were consistent among sites and site types indicating that there were differences in the size of organisms at perturbed and unperturbed sites.

The current study was the first to examine the effect of nutrients on residual variation in the full size spectra including benthic macroinvertebrates and fish in stream environments and showed that residual variance increased with perturbation to watersheds. One explanation for this results in the concept of limiting factors. The tight coupling of producers and consumer seen in low nutrient level systems is indicative of a bottom up system where primary productivity is limited by nutrient availability (Peterson et al. 1983). There is a switch from bottom up to top-down control as nutrient levels increase to a point where production becomes limited not by nutrients, but by grazing instead. (Neckles et al.1993). Although, the current study did not show specific evidence of a switch from bottom-up to top-down control, we did show that efficiency of the system decreased as nutrients increased, indicating that it is possible that there was a switch of control type in these streams. In order to further test this hypothesis, it would be necessary to design a study that addresses the switching from bottom-up to top-down control specifically in nutrient-rich streams.

The usefulness of the current study is found mainly in the provision of baseline and supplementary data on stream communities to the construction of effective management strategies in the future in order to protect local diversity in and around Gatineau Park and the City of Ottawa. In doing this, it is possible to suggest that small streams within Gatineau Park are in good, but variable health. There is some evidence of detrimental effects of urbanization seen within Park streams especially those bordering the Park in high impact areas such as Gatineau, Chelsea and streams surrounding Lac Philippe. In order to ensure that the diversity and ecological integrity of Gatineau Park remains, a monitoring plan, focused on these sites may be essential. In studying these streams more closely, it may be possible to mitigate some of the issues, and stop further impact from occurring within the Park. In order to properly assess the differences in biota and water quality between protected and perturbed streams in the National Capital Region, it would be essential to sample similar streams located within the Park and directly surrounding the Park in more urbanized areas. This study was a first attempt at answering this question however future researchers could design a more effective study to address the differences in Park and non-park streams in terms of benthic biota and water quality. In addition, I have suggested that it is possible to use aspects of the size spectrum such as the normalized density and the residual variance to test for differences between perturbed and unperturbed sites. These factors could be used in the future of monitor anthropogenic changes to small streams.

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APPENDIX

Appendix A. List of macroinvertebrate, crayfish, and fish species found within Gatineau Park and the surrounding regions of Gatineau, Hull, and the City of Ottawa ordered taxonomically (*denotes taxa found within Gatineau Park).

Arthropoda		Acronym
Insecta	***Coleoptera***	
	Curculionidae	C_Cur
	Dytiscidae larva	C_DytL
	*Elmidae	C_Elm
	*Elmidae larva	C_ElmL
	Haliplidae	C_Halip
	Hydrophilidae	C_Hydp
	Hydrophilidae larva	C_HydpL
	*Psephenidae	C_Psep
	Unknown adult	COLE
	Diptera	
	Ceratopogonidae	CERAT
	*Chironominae	CHIR
	*Dixidae	D_Dix
	*Empididae	D_Emp
	* Ephydridae	D_Eph
	*Orthocladiinae	ORTH
	*Psychodidae	D_Psy
	*Siamulidae	SIMU
	*Siamulidae larva	SIMU_P
	*Stratiomyidae	D_Strat
	*Tanypodinae	TANY
	*Tipulidae	TIPU
	*Unkown pupa	D_pupa
	Ephemeroptera	
	*Baetidae	E_Bae
	Caenidae	E_Cae
	*Ephemeridae	E_Ephe
	*Heptageniidae	E_Hept
	*Leptophemeridae	E_Lepto
	*Tricoythidae	E_Trico
	Megaloptera	
	*Corydalidae	CORY

		Odonata	
		Anisoptera	O_Anis
		*Calopterygidae	O_Cal
		*Gomphidae	O_Gomp
		Orthoptera	
		Unknown family	ORTH
		Plectoptera	
		*Chloroperlidae	P_Chl
		*Leuctridae	P_Leu
		*Nemouridae	P_Nem
		Peltoperlidae	P_Pelt
		*Perlidae	P_Perli
		*Perlodidae	P_Perlo
		Unknown adult	PLEC
		Trichoptera	
		*Brachycentridae	T_Bra
		*Glossomatidae	T_Glos
		*Helicopsychidae	T_Hel
		*Hydropsychidae	T_Hpsy
		*Hydroptilidae	T_Hpti
		*Lepidostomatidae	T_Lepi
		*Leptoceridae	T_Lept
		*Limniphilidae	T_Limn
		*Philopotamidae	T_Phil
		*Polycentropodidae	T_Poly
		*Psychomyiidae	T_Psy
		*Rhyacophilidae	T_Rhya
		*Unknown pupa	T_pupa
		Unknown adult	TRICH
	Arachnida	***Hydracarina***	
		*Unknown family	HYDR
		Acari	
		Unknown family	ACAR
Crustacea			
	Malacostraca	***Amphipoda***	
		Unknown family	AMPH
		Isopoda	
		*Unknown family	ISOP
		Decapoda	
		Cambaridae	
		**Cambarus robustus*	C

	Astacidae	
	**Orchonectes obscurus*	Oo
	**Orchonectes rusticus*	Or
	**Orchonectes virilis*	Ov
Annelida		
Clitella	***Oligochaeta***	
	*Unknown family	OLIG
	Hyrudinea	
	*Unknown family	HYRU
Mollusca		
Bivalvia	*Unknown family	BIV
Gastropoda	*Unknown family	GAST
Turbellaria		TURB
Chordata		
Actinopterygii	***Cypriniformes***	
	Castomidae	
	**Catostomus commersonii*	WS
	Cyrpinidae	
	**Notropis cornutus*	CS
	Notropis heterolepis	BNS
	**Phoxinus eos*	NRD
	**Phoxinus neogaeus*	FSD
	Pimephales notatus	BMN
	Pimephales promelas	FM
	**Rhinichthys cataractae*	LND
	**Semotilus atromaculatus*	CC
	Semotilus corporalis	FF
	Fundulidae	
	Fundulus diapahnus	BK
	Gadiformes	
	Lota lota	B
	Esociformes	
	Esocidae	
	**Esox Lucius*	NP
	Umbridae	
	**Umbra limi*	CMM
	Gasterosteiformes	
	Gasterosteodae	
	**Culaea inconstans*	BS

Perciformes	
Centrarchidae	
Ambloplites Rupestris	RB
**Lepomis gibbosus*	PS
**Lepomis macrochirus*	BG
**Micropterus dolomieui*	SMB
**Micropterus salmoides*	LMB
Percidae	
**Etheostoma exile*	ID
**Etheostoma nigrum*	JD
Percina caprodes	LP
Sander vitreus	W
Salmoniformes	
Salmonidae	
**Salvelinus fontinalis*	BT
Siluriformes	
Ictaluridae	
Noturus insignis	MT
Scorpaeniformes	
Cottoidae	
**Cottus bairdii*	MS
Siluriformes	
Ictaluridae	
**Ameiurus natalis*	YBH
**Ameiurus nebulosus*	BBH

Appendix B. General site information including site code and name, site location in decimal degrees, the person who sampled each site and the year sampled, as well as site classification by dominant land-use type (forest, agriculture, urban), and site location (shield/non-shield or park/non-park).

Site Code	Site Name	Latitude	Longitude	Sampler	Year	Land-use	Valley/Shield	Park/Non-Park
BEA	Bear Brook Creek	45.37504	-75.45916	Stephenson	2001/2002	Agriculture	Non-Shield	Non-Park
BEC	Beckett's Creek	45.51889	75.35889	Lento	2005	Agriculture	Non-Shield	Non-Park
BLA	Blackburn Creek	45.64131	-75.81520	Stephenson	2001/2002	Forest	Shield	Non-Park
BRA	Brassel's Creek	44.98498	-75.80118	Stephenson	2001/2002	Forest	Non-Shield	Non-Park
C12	Chelsea Creek	45.47927	-75.74585	Gruenert	2003	Forest	Non-Shield	Park
C2	Chelsea Creek	45.50971	-75.81232	Gruenert	2003	Forest	Shield	Park
C3	Chelsea Creek	45.50210	-75.81137	Gruenert	2003	Forest	Shield	Park
C3L	Chelsea Creek	45.50139	75 80833	Hamilton	2008	Forest	Shield	Park
C6	Chelsea Creek	45.49722	75 81389	Hamilton	2008	Forest	Shield	Park
C6b	Chelsea Creek	45.49167	75 83194	Hamilton	2008	Forest	Shield	Park
C9	Chelsea Creek	45.49306	75.79167	Hamilton	2008	Forest	Non-Shield	Park
CHE	Chelsea Creek	45.50209	-75 81147	Stephenson	2001/2002	Forest	Shield	Park
COR	Corriveau	45.68447	-75.73946	Stephenson	2001/2002	Forest	Shield	Non-Park
DESF	Des Fees	45 43708	-75.75575	Stephenson	2001/2002	Forest	Non-Shield	Park
DesFeesO	Des Fees	45.48333	75.79167	Hamilton	2008	Forest	Shield	Park
DesLoups	Des Loups	45.66667	76 22083	Hamilton	2008	Forest	Shield	Park
DEST	Des Trembles	45.42857	-75.76112	Stephenson	2001/2002	Forest	Non-Shield	Park
DT0	Des Trembles	45.43333	75.80000	Lento	2005	Forest	Non-Shield	Non-Park
DT1	Des Trembles	45.41778	75 75611	Lento	2005	Urban	Non-Shield	Non-Park
DT2	Des Trembles	45.45000	75 78333	Lento	2005	Urban	Non-Shield	Non-Park
Eardley	Eardley	45.57917	76 08750	Hamilton	2008	Forest	Shield	Park
Eardley2	Eardley	45 57500	76 08750	Hamilton	2008	Forest	Shield	Park
FOR	Fortune	45 51949	-75 84833	Stephenson	2001/2002	Forest	Shield	Park
Fortune1	Fortune	45.51944	75.85278	Hamilton	2008	Forest	Shield	Park
Fortune2	Fortune	45 51278	75.85139	Hamilton	2008	Forest	Shield	Park
Fortune3	Fortune	45.50694	75.88056	Hamilton	2008	Forest	Shield	Park
Gat1	Gatineau	45.48333	75.86250	Hamilton	2008	Forest	Shield	Park
GRE	Green Creek	45.42044	-75.59221	Stephenson	2001/2002	Agriculture	Non-Shield	Non-Park
HAR	Harwood Creek	45.38370	-75.97086	Stephenson	2001/2002	Agriculture	Non-Shield	Non-Park
HUN	Hunt Club Creek	45 34653	-75.69626	Stephenson	2001/2002	Urban	Non-Shield	Non-Park
JR-01	Jock River	45.25999	-75.70963	Stephenson	2001/2002	Urban	Non-Shield	Non-Park
KIN	Kingsmere	45.48028	-75.84905	Stephenson	2001/2002	Forest	Shield	Park
Kingsmere	Kingsmere	45.47778	75.87500	Hamilton	2008	Forest	Shield	Park
LAP	La Peche	45.62119	-76.09989	Stephenson	2001/2002	Forest	Shield	Park
LaPeche	La Peche	45.17500	76.17500	Hamilton	2008	Forest	Shield	Park
LaPeche2	La Peche	45.62778	76.10556	Hamilton	2008	Forest	Shield	Park
LDS	Leamy Downstream	45.46240	-75.72984	Lento	2005	Urban	Non-Shield	Non-Park
LDS2	Leamy Downstream	45.41667	75.75000	Lento	2005	Urban	Non-Shield	Non-Park
LMS	Leamy Midstream	45.46736	-75.74700	Lento	2005	Urban	Non-Shield	Non-Park
Loutre	Lac Loutre	45.66111	76.20556	Hamilton	2008	Forest	Shield	Park
LUS2	Leamy Upstream	45.48056	75.80000	Hamilton	2008	Forest	Shield	Park
M10	Meech Creek	45.56411	-75.87960	Gruenert	2003	Forest	Non-Shield	Park
M11	Meech Creek	75.75611	75 88333	Lento	2005	Forest	Non-Shield	Park
M12	Meech Creek	45.58271	-75.89584	Gruenert	2003	Forest	Non-Shield	Park
M14	Meech Creek	45.60067	-75.89120	Gruenert	2003	Forest	Non-Shield	Park
MEE	Meech Creek	45.57095	-75.88722	Stephenson	2001/2002	Forest	Non-Shield	Park
Meech1	Meech Creek	45.54444	75.91667	Hamilton	2008	Forest	Shield	Park
Meech2	Meech Creek	45.54306	75.91111	Hamilton	2008	Forest	Shield	Park
Meech3	Meech Creek	45.53333	75.90000	Hamilton	2008	Forest	Shield	Park
Meech4	Meech Creek	45.52778	75.88611	Hamilton	2008	Forest	Shield	Park

Site Code	Site Name	Latitude	Longitude	Sampler	Year	Land-use	Valley/Shield	Park/Non-Park
Mine 3	Mine Road	45 47361	75 79306	Hamilton	2008	Forest	Shield	Park
Mine1	Mine Road	45 48139	75 80556	Hamilton	2008	Forest	Shield	Park
MO	Meech Outflow	45 53333	75 86667	Lento	2005	Forest	Non-Shield	Park
MOS	Mosquito Creek	45 28295	-75 66827	Lento	2005	Agriculture	Non-Shield	Non-Park
NCR	North Castor Creek	45 29557	-75 50547	Stephenson	2001/2002	Agriculture	Non-Shield	Non-Park
Notch	Notch Road	45 48472	75 81250	Hamilton	2008	Forest	Shield	Park
POO	Poole Creek	45 26243	75 92558	Stephenson	2001/2003	Urban	Non-Shield	Non-Park
Renaud1	Lac Renaud	45 60139	76 01944	Hamilton	2008	Forest	Shield	Park
Renaud2	Lac Renaud	45 59861	76 01944	Hamilton	2008	Forest	Shield	Park
SAWD	Sawmill Creek	45 38999	-75 67619	Lento	2005	Urban	Non-Shield	Non-Park
SAWU	Sawmill Creek	45 33787	-75 63348	Stephenson	2001/2002	Urban	Non-Shield	Non-Park
STI	Stillwater Creek	45 34969	-75 82456	Stephenson	2001/2002	Urban	Non-Shield	Non-Park
TAYD	Taylor Drain	45 13985	75 72669	Stephenson	2001/2002	Agriculture	Non-Shield	Non-Park
UNA	Unamed	45 45973	-75 72426	Stephenson	2001/2002	Urban	Shield	Non Park
VOY	Voyageur Creek	45 46505	-75 54846	Stephenson	2001/2002	Urban	Non-Shield	Non-Park

Appendix C. Water quality for 67 sites in the National Capital Region including total phosphorus (mg/L), total nitrogen (mg/L), conductivity (μs/cm), chloride (mg/L), and periphyton biomass (mg Chla/m^2). Standard deviations were taken from 2-3 counts.

Site	Total Phosphorus (mg/L)	Total Phosphorus SD	Total Nitrogen (mg/L)	Total Nitrogen SD	Conductivity (us/cm)	Conductivity SD	Chloride (mg/L)	Chloride SD	Periphtyon Biomass (mg Chla/m2)	Periphyton Biomass SD
BEA	0.08	0.058	1.87	1.15	741.11	103.74	103.89	21.48	293.00	317.76
BEC	0.08	0.034	1.06	0.64	564.29	65.03	43.14	12.82	71.87	46.73
BLA	0.01	0.006	0.35	0.05	67.89	14.63	0.96	0.25	78.77	1.65
BRA	0.02	0.007	0.57	0.21	414.00	93.05	19.57	11.54	27.12	13.45
C12	0.07	0.000	0.12	0.01	603.00	603.33	96.00	1.00	3.53	1.34
C2	0.02	0.004	0.06	0.02	500.00	500.00	93.00	1.00	9.97	0.34
C3	0.01	0.000	0.11	0.01	533.00	533.33	47.67	1.00	0.90	0.71
C3L	0.01	0.001	0.44	0.03	357.00	11.55	40.76	2.15	20.87	1.91
C6	0.01	0.002	0.35	0.04	79.00	17.32	44.70	3.09	13.88	6.50
C6b	0.01	0.009	0.49	0.04	360.00	32.15	42.42	3.24	7.69	8.09
C9	0.02	0.005	0.18	0.02	240.00	23.09	48.48	2.31	21.47	6.73
CHE	0.02	0.012	0.50	0.13	285.00	78.42	42.46	12.00	2.20	6.07
COR	0.01	0.004	0.31	0.07	71.56	21.12	1.30	0.31	2.75	4.28
DESF	0.04	n/a	1.66	n/a	960.00	n/a	135.00	n/a	13.00	7.24
DesFeesO	0.17	0.177	0.46	0.89	137.00	60.28	1.78	0.33	28.26	7.64
DesLoups	0.01	0.002	0.87	0.07	270.00	13.65	0.68	0.03	9.76	3.72
DEST	0.06	n/a	0.91	n/a	888.00	n/a	90.00	n/a	97.20	56.91
DT0	0.04	n/a	0.49	0.02	985.00	21.21	123.50	2.12	12.06	4.71
DT1	0.09	0.026	1.07	0.07	920.00	n/a	125.50	13.44	21.47	7.96
DT2	0.07	0.019	1.09	0.16	890.00	28.28	115.50	24.75	20.25	28.90
Eardley	0.02	0.003	0.37	0.01	90.00	6.66	0.89	0.30	5.76	3.59
Eardley2	0.02	0.002	1.19	0.02	103.00	9.50	0.61	0.61	5.35	1.95
FOR	0.01	0.001	0.45	0.01	320.00	n/a	26.00	n/a	11.29	9.78
Fortune1	0.04	0.054	0.35	0.30	136.00	27.43	4.90	0.82	4.46	1.39
Fortune2	0.02	0.017	0.39	0.05	69.00	24.98	6.05	4.01	5.43	2.02
Fortune3	0.02	0.004	0.29	0.02	53.00	5.03	0.06	0.01	7.86	3.96
Gat1	0.03	0.027	0.37	0.15	257.00	28.51	0.30	0.00	4.03	2.81
GRE	0.07	0.034	1.42	0.89	1395.00	328.17	286.94	82.40	74.09	15.54
HAR	0.02	0.031	0.78	0.31	820.00	744.44	149.50	242.69	92.13	65.89
HUN	0.08	0.050	1.95	1.36	1117.50	477.18	225.38	184.15	40.42	7.29
JR 01	0.04	0.017	1.36	0.96	646.00	78.34	65.57	25.37	42.64	24.25
KIN	0.06	0.068	0.64	0.44	200.00	n/a	4.65	0.07	8.51	3.71
Kingsmere	0.01	0.009	0.41	0.06	110.00	5.77	3.47	0.31	9.35	5.27
LAP	0.03	0.028	0.48	0.18	72.78	22.75	1.48	0.76	9.80	6.41
LaPeche	0.01	0.003	0.20	0.03	89.00	5.57	0.13	0.08	12.11	4.23
LaPeche2	0.01	0.002	0.32	0.03	353.00	6.66	1.29	0.44	1.31	1.16
LDS	0.15	0.004	1.46	0.15	1110.00	56.57	230.00	14.14	84.26	33.04
LDS2	0.20	0.009	3.00	1.06	1685.00	49.50	385.00	7.07	55.17	34.24
LMS	0.60	n/a	5.23	n/a	1130.00	n/a	210.00	n/a	20.16	13.80
Loutre	0.02	0.002	0.37	0.10	74.00	6.56	2.37	0.16	25.28	7.21
LUS	0.04	0.003	0.84		290.00	0.00	5.36	0.23	20.69	12.91
LUS2	0.02	0.035	0.30	0.02	247.00	23.80	0.48	0.02	10.11	5.26
M10	0.01	0.003	0.05	0.02	95.00	95.00	1.23	0.06	1.72	0.71
M11	0.29	0.000	0.02	0.03	95.50	3.54	1.30	0.07	23.53	13.69
M12	0.03	0.001	0.08	0.03	106.67	106.67	1.50	0.10	1.90	0.72
M14	0.06	0.000	0.13	0.04	114.00	114.00	1.90	0.00	2.33	1.42
MEE	0.01	0.000	0.26	0.01	98.00	n/a	1.10	0.00	40.80	9.44
Meech1	0.02	0.020	0.53	0.16	72.00	9.45	0.20	0.01	28.33	52.07
Meech2	0.02	0.020	0.67	0.17	297.00	57.46	0.25	0.05	32.04	64.74
Meech3	0.01	0.002	0.45	0.03	287.00	10.00	0.16	0.05	14.55	11.94
Meech4	0.01	0.019	0.46	0.12	155.00	9.81	0.09	0.03	3.02	2.75
Mine 3	0.06	0.078	0.67	0.69	293.00	23.09	1.35	0.08	10.81	1.84
Mine1	0.02	0.011	0.44	0.87	85.00	10.00	2.61	0.92	20.57	22.33
MO	0.02	n/a	0.36	n/a	90.00	n/a	1.10	n/a	6.16	5.11
MOS	0.08	0.029	0.83	0.12	636.67	146.01	60.88	29.00	59.53	21.97
NCR	0.04	n/a	1.43	n/a	995.56	n/a	106.44	n/a	68.57	19.49
Notch	0.02	0.013	0.57	0.05	125.00	25.17	33.51	2.28	3.86	4.97
POO	0.02	0.037	1.37	1.10	821.11	247.21	84.22	27.67	62.53	47.50
Renaud1	0.02	0.010	0.55	0.08	66.00	5.86	0.34	0.15	4.62	2.17
Renaud2	0.01	0.002	0.32	0.01	189.00	10.50	0.12	0.08	4.58	2.64

Site	Total Phosphorus (mg/L)	Total Phosphorus SD	Total Nitrogen (mg/L)	Total Nitrogen SD	Conductivity (us/cm)	Conductivity SD	Chloride (mg/L)	Chloride SD	Periphtyon Biomass (mg Chla/m2)	Periphyton Biomass SD
SAWD	0 09	n/a	1 36	n/a	1327 78	n/a	226 67	n/a	92 97	15 69
SAWU	0 06	n/a	0 77	n/a	974 67	n/a	140 13	n/a	68 56	28 74
STI	0 06	0 078	1 58	0 68	1055 71	358 39	188 57	107 17	91 94	37 47
TAYD	0 07	0 039	3 14	0 83	562 50	648 09	23 24	161 38	28 70	14 08
Taylor	0 01	0 003	0 46	0 04	367 00	7 81	0 54	0 21	5 62	5 61
UNA	0 10	0 006	1 80	n/a	1340 00	0 00	283 33	5 77	32 12	14 85
VOY	0 06	0 032	2 13	1 27	1434 00	818 69	281 70	173 78	31 53	39 62

Appendix D. Invertebrate biomass (±SD) for 67 sites in the National Capital Region. Standard deviation was taken from 2-3 counts.

Site	Total Invertebrate Biomass (g DM/m2)	Invertebrate Biomass SD	Total Crayfish Biomass (g DM/m2)	ACAR	AMPH	ARAC	C_Cur	C_Dytl	C_Elm	C_ElmL	C_Hali	C_Halip	C_Hydp	C_HydpL	C_Psep	CERAT	CHIR	COLE	CORY	D_Deut	D_dix	D_Emp	D_Eph	D_Psy	D_pupa	D_Stra	E_Bae	E_Cae	E_Ephe	E_Hept	E_Lept	E_Trico	GAST	H_Vel	HIRU	HYDA	HYDR	ISOP
BFA	43.649	18.750	n/a	0.000	19.962	0.011	0.000	0.000	3.737	0.000	0.000	0.000	0.000	0.000	0.155	0.000	0.715	0.000	0.017	0.000	0.000	0.000	0.000	0.000	0.037	0.000	0.000	0.097	0.000	0.020	0.000	0.113	9.049	0.000	0.079	0.000	0.000	0.000
BEC	7.940	1.047	n/a	0.000	0.012	0.000	0.000	0.000	0.131	0.000	0.000	0.000	0.000	0.000	0.139	0.000	0.353	0.000	0.000	0.000	0.000	0.010	0.000	0.000	0.000	0.000	0.358	0.000	0.000	0.000	0.000	0.000	0.041	0.000	0.000	0.000	0.000	0.000
BLA	0.533	2.518	n/a	0.000	0.000	0.000	0.000	0.000	0.034	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.011	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.048	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
BRA	7.111	1.970	n/a	0.000	0.000	0.000	0.000	0.000	0.349	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.408	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.369	0.000	0.000	0.047	0.000	0.000	0.220	0.000	0.000	0.000	0.000	0.005
C12	0.280	0.572	n/a	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.082	0.000	0.081	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.003	0.000	0.020	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
C2	0.826	1.632	n/a	0.000	0.000	0.001	0.000	0.000	0.025	0.000	0.000	0.000	0.000	0.000	0.000	0.001	0.033	0.000	0.000	0.000	0.000	0.004	0.000	0.000	0.000	0.000	0.005	0.000	0.000	0.185	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
C3	1.276	1.222	n/a	0.000	0.000	0.001	0.000	0.000	0.068	0.000	0.000	0.000	0.000	0.000	0.084	0.000	0.078	0.000	0.206	0.000	0.000	0.001	0.000	0.000	0.000	0.000	0.037	0.000	0.000	0.024	0.000	0.000	0.045	0.000	0.000	0.000	0.000	0.000
C3l	0.672	0.779	0.161	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.131	0.000	0.045	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.003	0.000	0.014	0.000	0.000	0.038	0.002	0.000	0.000	0.000	0.000	0.000	0.000	0.000
C6	0.266	0.764	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.004	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.006	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
C6b	0.097	0.273	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.013	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.016	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
C9	1.501	0.417	0.010	0.000	0.000	0.000	0.000	0.000	0.010	0.000	0.000	0.000	0.000	0.000	0.008	0.000	0.043	0.000	0.000	0.000	0.000	0.000	0.001	0.000	0.006	0.000	0.091	0.000	0.002	0.008	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
CHL	0.571	1.080	n/a	0.000	0.000	0.000	0.000	0.000	0.040	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.037	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.003	0.000	0.000	0.000	0.002	0.020	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
COR	0.671	1.000	n/a	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.030	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.018	0.000	0.000	0.046	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
DESE	0.410	4.604	n/a	0.000	0.000	0.000	0.000	0.000	0.315	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
DesLeesO	0.350	0.332	0.000	0.000	0.000	0.000	0.000	0.000	0.036	0.000	0.000	0.000	0.000	0.000	0.089	0.000	0.056	0.000	0.000	0.000	0.000	0.000	0.000	0.011	0.001	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
DesLoups	1.268	1.040	0.557	0.000	0.000	0.000	0.000	0.000	0.022	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.007	0.000	0.116	0.000	0.000	0.000	0.000	0.000	0.001	0.000	0.005	0.000	0.000	0.236	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
DLSE	8.922	3.678	n/a	0.007	0.000	0.000	0.000	0.000	2.744	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.170	0.000	0.076	0.000	0.000	0.000	0.000	0.000	0.011	0.000	0.037	0.000	0.000	0.182	0.000	0.000	0.412	0.000	0.000	0.000	0.000	0.000
DT0	0.094	0.550	n/a	0.000	0.000	0.000	0.000	0.000	0.058	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.007	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.003
DT1	0.517	1.650	n/a	0.000	0.000	0.000	0.000	0.000	0.073	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.034	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.004	0.000	0.000	0.000	0.000	0.000	0.016	0.125	0.000	0.000	0.000	0.011
DT2	2.593	0.797	n/a	0.000	0.000	0.000	0.043	0.000	1.744	0.000	0.000	0.000	0.000	0.000	0.000	0.003	0.051	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.013	0.000	0.000	0.001	0.000	0.000	0.000	0.000	0.000	0.000	0.001	0.000
Eardley	0.556	0.280	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.003	0.000	0.000	0.000	0.000	0.000	0.000	0.066	0.000	0.016	0.000	0.000	0.000	0.000	0.000	0.005	0.000	0.001	0.000	0.000	0.026	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Eardley2	0.489	0.414	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.012	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.001	0.000	0.000	0.000	0.000	0.072	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
FOR	5.282	5.101	n/a	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.311	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.188	0.000	0.096	0.000	0.000	0.465	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Fortune1	0.158	0.238	n/a	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.011	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.001	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Fortune2	0.051	0.166	n/a	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.006	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Fortune3	1.038	0.370	0.000	0.000	0.000	0.000	0.000	0.000	0.024	0.000	0.000	0.000	0.000	0.000	0.187	0.000	0.174	0.000	0.069	0.000	0.000	0.000	0.000	0.000	0.002	0.000	0.000	0.000	0.000	0.004	0.024	0.002	0.000	0.000	0.000	0.000	0.000	0.000
Gat1	0.050	0.286	0.889	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.007	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.001	0.004	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
GRE	11.879	2.048	n/a	0.002	0.000	0.000	0.000	0.000	0.050	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.085	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.035	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.113	0.000	0.000	0.000	0.000	0.374
HAR	1.737	1.229	n/a	0.000	0.450	0.000	0.000	0.050	0.044	0.000	0.046	0.113	0.000	0.000	0.000	0.000	0.021	0.379	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.002	0.000	0.000	0.000	0.000	0.093	0.000	0.000	0.000	0.000	0.063
HUN	*4.729*	*1.187*	*n/a*	*0.000*	*0.000*	*0.000*	*0.000*	*0.000*	*0.000*	*0.000*	*0.000*	*0.000*	*0.000*	*0.000*	*0.000*	*0.000*	*0.000*	*0.000*	*0.000*	*0.000*	*0.000*	*0.000*	*0.000*	*0.000*	*0.001*	*0.000*	*0.000*	*0.000*	*0.000*	*0.000*	*0.000*	*0.000*	*0.000*	*0.000*	*0.000*	*0.000*	*0.000*	*0.133*
JR 01	1.618	1.210	n/a	0.003	0.000	0.000	0.000	0.000	0.003	0.000	0.000	0.000	0.000	0.000	0.016	0.000	0.134	0.000	0.000	0.000	0.000	0.005	0.000	0.000	0.000	0.000	0.054	0.000	0.023	0.046	0.000	0.026	0.000	0.000	0.000	0.000	0.000	0.000
KIN	1.679	1.185	n/a	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.009	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.075	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Kingsmere	0.369	3.165	0.021	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.002	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
LAP	1.856	2.624	n/a	0.000	0.000	0.000	0.000	0.000	0.030	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.013	0.000	0.039	0.000	0.000	0.000	0.000	0.000	0.003	0.000	0.002	0.002	0.000	0.000	0.011	0.000	0.000	0.000	0.000	0.000	0.000	0.000
LaPeche	0.727	0.495	0.000	0.000	0.000	0.000	0.000	0.000	0.027	0.000	0.000	0.000	0.000	0.000	0.040	0.000	0.007	0.000	0.006	0.000	0.000	0.000	0.000	0.000	0.002	0.000	0.016	0.000	0.000	0.000	0.007	0.000	0.000	0.000	0.000	0.000	0.000	0.000
LaPeche2	0.279	1.059	0.096	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.004	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
LDS	0.538	0.388	n/a	0.000	0.000	0.000	0.000	0.000	0.065	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.012	0.000	0.000	0.000	0.000	0.003	0.000	0.000	0.001	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.050	0.000	0.009	0.000	0.000	0.017
LDS2	0.041	0.084	n/a	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.004	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
LMS	1.337	0.411	n/a	0.000	0.000	0.000	0.000	0.000	0.454	0.000	0.024	0.000	0.009	0.000	0.000	0.000	0.046	0.000	0.001	0.000	0.000	0.000	0.000	0.000	0.003	0.000	0.004	0.000	0.000	0.000	0.000	0.000	0.016	0.000	0.047	0.000	0.000	0.008
Loutre	1.400	0.823	1.671	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.050	0.000	0.042	0.000	0.000	0.000	0.000	0.001	0.000	0.000	0.002	0.000	0.006	0.000	0.000	0.063	0.000	0.005	0.000	0.000	0.000	0.000	0.000	0.000
LUS	4.642	3.136	n/a	0.000	0.000	0.000	0.000	0.000	1.046	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.223	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.125	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.064
LUS2	0.068	0.413	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.008	0.000	0.000	0.000	0.005	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
M10	0.687	1.836	n/a	0.000	0.000	0.002	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.010	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.003	0.000	0.000	0.173	0.000	0.000	0.123	0.000	0.000	0.000	0.000	0.000
M11	1.146	1.861	n/a	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.007	0.000	0.013	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.018	0.000	0.000	0.098	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
M12	1.813	1.277	n/a	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.003	0.000	0.027	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.001	0.000	0.002	0.004	0.000	0.162	0.000	0.000	0.202	0.000	0.000	0.000	0.000	0.000
M14	0.387	0.503	n/a	0.000	0.001	0.000	0.000	0.000	0.110	0.000	0.000	0.000	0.000	0.000	0.104	0.000	0.005	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.021	0.000	0.000	0.022	0.001	0.000	0.006	0.000	0.000	0.000	0.000	0.000
MEE	1.132	0.922	n/a	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.006	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.004	0.000	0.058	0.000	0.000	0.091	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Meech1	0.243	0.716	n/a	0.000	0.000	0.000	0.000	0.000	0.000	0.002	0.000	0.000	0.000	0.000	0.000	0.000	0.006	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.008	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Meech2	0.300	0.554	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.020	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.001	0.000	0.004	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Meech3	0.081	1.193	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.002	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.014	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Meech4	0.114	1.409	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.011	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.047	0.000	0.000	0.000
Mine1	4.039	1.601	n/a	0.000	0.000	0.000	0.000	0.000	0.259	0.125	0.000	0.000	0.000	0.000	0.000	0.000	0.107	0.000	0.000	0.000	0.000	0.002	0.000	0.000	0.000	0.000	0.003	0.000	0.000	0.000	0.000	0.000	0.135	0.000	0.000	0.000	0.000	0.007
Mine3	0.377	0.858	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.004	0.000	0.002	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
MO	0.310	0.766	n/a	0.000	0.000	0.000	0.000	0.000	0.045	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.006	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.003	0.000	0.003	0.000	0.000	0.167	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
MOS	7.203	1.170	n/a	0.004	0.227	0.000	0.000	0.000	0.912	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.240	0.000	0.000	0.001	0.000	0.000	0.000	0.000	0.006	0.000	0.367	0.000	0.000	0.011	0.000	0.000	0.232	0.000	0.005	0.000	0.000	0.166
NCR	20.690	3.556	n/a	0.001	0.000	0.000	0.000	0.000	2.837	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.772	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.152	0.000	0.131	0.000	0.000	0.154	0.000	0.000	0.674	0.000	0.000	0.000	0.000	0.097
Notch	0.153	0.265	n/a	0.000	0.000	0.000	0.000	0.000	0.021	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.016	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.002	0.000	0.018	0.000	0.000	0.000	0.015	0.000	0.000	0.000	0.000	0.000	0.000	0.000
POO	0.405	0.232	n/a	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.029	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.001	0.000	0.000	0.000	0.000	0.000	0.006	0.000	0.000	0.000	0.000	0.239
Renaud1	0.160	0.371	1.307	0.000	0.000	0.000	0.000	0.000	0.028	0.006	0.000	0.000	0.000	0.000	0.000	0.000	0.027	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.005	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Renaud2	0.155	0.270	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.003	0.000	0.031	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.001	0.000	0.003	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
SAWD	2.345	0.799	n/a	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.002	0.000	0.000	0.007	0.000	0.009	0.000	0.000	0.000	0.000	0.000	0.017	0.000	0.000	0.000	0.002	0.000	0.000	0.000	0.010	0.000	0.017	0.000	0.000	0.253
SAWU	1.162	n/a	n/a	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.014	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.002	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
SET	0.176	2.242	n/a	0.000	0.006	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.007	0.006	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.007	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.011
TAYD	5.123	1.703	n/a	0.000	0.000	0.000	0.000	0.000	0.130	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.013	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.099	0.000	0.000	0.000	0.000	0.000	0.303	0.000	0.000	0.000	0.000	0.011
Taylor	3.945	1.363	0.352	0.000	0.000	0.000	0.000	0.000	0.106	0.029	0.000	0.000	0.000	0.000	0.000	0.000	0.042	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.002	0.000	0.004	0.000	0.000	0.120	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
UNA	4.960	0.645	n/a	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.005	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.070	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.076	0.000	0.000	0.000	0.000	0.000
VOY	7.917	0.629	n/a	0.000	0.000	0.000	0.031	0.000	0.000	0.000	0.000	0.000	0.000	0.008	0.000	0.003	0.013	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.011	0.000	0.001	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.045	0.000	0.000	6.062

Appendix D. Fish biomass (±SD) for 67 sites in the National Capital Region. Standard deviation was taken from 2-3 counts.

Site	Total Fish Biomass (g DM/m2)	B	BBH	BC	BG	BK	BM	BNM	BNS	BS	BT	CC	CS	CMM	D	FF	FM	fry	FSD	ID	JD	LMB	LND	LP	MS	MT	NRD	PS	RB	SMB	SLS	STS	W	WS
BFA	4.79	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.69	0.00	0.00	0.00	1.51	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	1.61	0.55	0.00	0.00	0.00	0.44
BFC	9.57	2.31	0.00	0.00	0.00	0.00	0.00	0.12	0.19	0.00	0.00	1.91	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.37	0.00	0.75	0.89	0.02	0.00	0.00	0.00	0.15	0.18	1.82	0.67	0.00	0.00
BLA	2.11	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.71	0.40	0.00	0.00	0.00	0.00
BRA	13.84	0.00	0.00	0.00	1.98	0.00	0.00	0.00	0.00	0.24	0.00	3.41	0.00	0.00	0.00	0.94	0.00	0.00	0.00	0.00	1.34	0.00	0.22	1.46	0.00	0.00	0.00	0.68	2.76	0.82	0.00	0.00	0.00	0.00
C12	0.48	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.07	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.10	0.00	0.01	0.00	0.00	0.09	0.16
C2	2.64	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.91	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.70	0.00	0.00	0.00	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00
C3	1.15	0.22	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.21	0.10	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.31	0.00	0.00	0.00	0.20	0.05	0.00	0.00	0.00	0.00	0.00	0.06
C31	1.27	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	11.54	0.25	0.00	0.00	0.00	0.00	0.00	0.63	0.00	0.00	0.00	8.30	0.00	0.00	0.00	8.81	0.00	0.00	0.00	0.00	0.00	0.00	0.02
C6	7.79	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	5.83	7.11	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
C6b	0.79	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	3.25	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
C9	4.45	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	3.69	1.02	0.00	0.00	0.00	0.00	0.00	0.47	0.00	0.00	0.00	22.06	0.00	1.07	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.12
CHI	6.27	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.56	0.00	0.00	1.73	1.04	0.00	0.00	0.00	0.00	0.00	0.00	1.58	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.34
COR	1.75	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.00	0.00	0.65	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.01	0.00	0.00	0.00	0.00	0.00	0.06	0.00	0.00	0.00	0.00	0.00
DESF	1.11	0.00	0.00	0.00	0.00	0.46	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.66	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
DesFecsO	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
DesLoups	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
DEST	10.76	0.00	0.00	0.00	0.00	0.08	0.00	0.28	0.00	0.00	0.00	1.19	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	8.65	0.56	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
D10	8.82	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.22	0.00	6.93	0.00	0.00	0.00	0.00	0.00	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.60	0.00	0.00	0.00	0.00
D11	7.99	0.00	0.00	0.00	0.00	0.00	0.00	0.92	0.00	0.08	0.00	2.32	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	3.47	0.00	0.52	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.67
D12	9.94	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.24	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	9.08	0.00	0.62	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Eardley	1.13	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	4.09	0.78	0.00	0.00	0.00	0.00	0.00	1.83	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Eardley2	0.57	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
FOR	8.14	0.00	0.00	0.00	0.21	0.00	0.00	0.00	0.00	0.00	0.00	1.49	0.00	0.00	6.29	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.09	0.00	0.00	0.00	0.00	0.00	0.05
Fortune1	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Fortune2	0.24	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.24	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Fortune3	2.22	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	6.15	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	15.30	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Gat1	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
GRF	3.23	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.20	0.00	0.00	0.00	0.68	0.00	0.00	0.00	0.00	0.00	0.00	0.93	0.02	0.00	0.00	0.00	0.00	0.13	0.00	0.00	0.00	0.00	0.21
HAR	1.50	0.00	0.00	0.00	0.00	0.05	0.00	0.00	0.00	0.70	0.00	0.36	0.00	0.00	0.00	0.00	0.00	0.39	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
HUN	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
JR01	6.43	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.05	0.00	0.00	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	3.72	0.87	0.00	0.00	0.00	0.00	0.18	1.58	0.00	0.00	0.00	0.00
KIN	3.80	0.00	0.00	0.00	0.13	0.00	0.00	0.00	0.00	0.00	0.00	2.42	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.24
Kingsmere	2.65	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	15.00	0.29	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.22	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
LAP	3.15	0.00	0.00	0.00	0.00	0.10	0.00	0.00	0.00	0.00	0.00	0.38	0.00	0.00	0.05	0.06	0.00	0.00	0.00	0.11	0.00	0.00	2.16	0.13	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.16
LaPeche	0.94	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	2.21	0.00	0.68	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
LaPeche2	0.92	0.00	0.65	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.63	0.40	0.00	0.35	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
LDS	1.71	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.70	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
LDS2	7.35	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
LMS	11.89	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.55	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	6.70	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Loutre	0.11	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.05	0.00	0.00	0.00	0.05
LUS	0.63	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.27	0.00	8.36	0.00	0.00	0.00	0.24	0.00	0.00	0.00	0.00	0.00	0.00	0.59	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	2.41
LUS2	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
M10	4.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.61	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
M11	1.15	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.17	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	3.85	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
M12	0.21	0.00	0.00	0.00	0.00	0.00	0.00	0.17	0.00	0.00	0.00	0.27	0.31	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.39	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
M14	1.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.16	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
MF1	10.37	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Meech1	0.48	0.00	0.00	0.00	0.14	0.00	0.00	0.00	0.00	0.00	0.11	0.24	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Meech2	0.74	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.74	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Meech3	3.28	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	3.28	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Meech4	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Mine1	2.12	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.16	0.00	1.37	0.13	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.66	0.00	0.00	0.00	0.23	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Mine3	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
MO	12.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.14	0.00	0.00	0.12	0.00	0.00	0.00	3.46	0.00	0.00	0.00	0.00	0.00	0.00	6.66	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
MOS	12.29	0.00	0.00	0.17	0.00	0.00	0.51	0.68	0.42	0.07	0.00	2.49	0.00	0.00	0.00	0.26	0.00	0.01	0.00	0.00	0.00	0.00	0.89	0.26	0.25	0.00	0.00	0.00	0.64	0.13	0.00	0.17	0.00	1.14
NCR	0.10	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.73	0.00	0.00	0.00	0.88	0.00	0.00	0.00	0.00	0.00	0.00	9.74	0.86	0.00	0.00	0.00	0.00	0.08	0.00	0.00	0.00	0.00	0.00
Notch	1.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	4.53	0.61	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
POO	6.81	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Renaud1	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Renaud2	0.58	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.58	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
SAWD	1.30	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.00	0.00	0.17	0.00	0.00	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	2.61	0.06	0.55	0.00	0.00	0.00	0.30	0.00	0.00	0.00	0.13	0.00
SAWU	1.09	0.00	0.00	0.00	0.00	0.04	0.00	0.00	0.00	1.12	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
STI	2.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.09	0.00	0.00	0.00	0.00	0.67	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.23	0.06	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
TAYD	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.06	0.00	0.00	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.71	0.17	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Taylor	5.52	0.00	0.00	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	1.47	0.00	6.03	0.00	0.00	0.00	0.00	0.82	0.00	0.00	0.00	0.00	0.00	0.00	0.00	3.35	0.07	0.00	0.00	0.00	0.00	0.00	0.00
UNA	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
VOY	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Appendix E. Map illustrating the boundaries of Gatineau Park and sample sites sampled by myself in 2008 and 2009.