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THE ASSESSMENT OF SEDIMENT PCB CONTAMINATION AND BIOLOGICAL IMPACTS IN LYONS CREEK EAST (NIAGARA RIVER AREA OF CONCERN)

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NWRI Contribution No. 06-414

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by

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SUMMARY

This report describes sediment and biota quality in Lyons Creek East (Niagara River Area of Concern). Previous studies have shown elevated levels of polychlorinated biphenyls (PCBs) in the sediment and detrimental effects on biota in the creek. As part of the GL2020 Action Plan, the National Water Research Institute of Environment Canada applied BEAST (Benthic Assessment of Sediment) methodology to Lyons Creek. Sampling focused mainly on the area between the Lyons Creek pumping station at the Welland Canal to Highway 140; this area was identified as having the highest levels of PCBs based on a preliminary chemical screening performed by the Ontario Ministry of Environment (MOE). Four neighbouring creeks, similar in morphology to Lyons Creek, were sampled as reference locations.

BEAST methodology involves the assessment of sediment quality based on a multivariate technique using data on benthic community structure, the functional responses of laboratory organisms in toxicity tests, and the physical and chemical attributes of the sediment and overlying water. Data are compared to biological criteria developed previously for the Laurentian Great Lakes. As there is the presence of a persistent biomagnifiable toxicant (PCBs) in the sediments of Lyons Creek, its bioavailability and potential for effects on fish and wildlife through biomagnification was assessed. This involved (a) analyses of the relationships of PCBs in benthic invertebrates to those in sediment, and (b) predictions of concentrations of PCBs in receptor species (representative consumers of benthic invertebrates and their predators) using screening-level trophic transfer models. A decision-making framework for sediment contamination, developed by the Canada-Ontario Agreement Sediment Task Group, was applied to the study to arrive at a decision on sediment quality for each site.

In October of 2002 and 2003, Environment Canada collected sediment for physico-chemical analyses and laboratory toxicity tests, overlying water, and benthic invertebrates for community structure analysis at 15 sites in Lyons Creek and 6 sites in neighbouring reference creeks. Benthic invertebrate tissue samples were collected at 11 of the 15 Lyons Creek sites and at 4 of the 6 reference sites. Sediment and biota samples were analyzed for PCBs (and other organic contaminants) and a series of physico-chemical variables were measured in the sediment and overlying water. Exposed and reference sites were compared in terms of PCB concentrations in

sediment and invertebrates. Relationships between PCBs in benthic invertebrates and PCBs in sediment were evaluated by regression analysis. Physico-chemical sediment and water variables were included as additional predictors. Concentrations of PCBs in the tissues of fish and wildlife receptors (Brown Bullhead, Carp, Bluegill, Largemouth Bass, Goldeneye, and Mink) were predicted by multiplying measured body concentrations in the resident invertebrates by relevant biomagnification factors obtained from a review of pre-existing studies. The predicted concentrations in the fish receptors were compared guidelines derived for the protection of fish consuming wildlife and to actual concentrations observed in sport fish collected in the creek by the MOE.

Total PCBs in the top 10 cm of sediment in Lyons Creek range from 0.016 to 12.55 µg/g, and are greater than the Canadian sediment quality guidelines for PCBs and greater than concentrations observed in reference creeks (range: 0.003 to 0.016 µg/g). The highest sediment PCB concentrations in Lyons Creek are upstream of Highway 140.

There is strong evidence of toxicity at 3 of the 15 Lyons Creek sites; these sites are located upstream of Highway 140. Toxicological response is most strongly related to a combination of metals or PAHs depending on the endpoint.

The BEAST model could not be used for the assessment of the Lyons Creek community structure since the current reference database consists of nearshore lake sites and does not contain habitat characteristics and community structure data for connecting channels or small streams or creeks in Southern Ontario. An Analysis of Variance (ANOVA) (with and without adjustment for covariates) was therefore used to compare Lyons Creek communities to those at the neighbouring reference creeks. There is a significantly lower abundance of odonates, low taxon diversity and the absence of key invertebrate families at 1 site, located upstream of Highway 140.

Total PCBs in benthic invertebrates in Lyons Creek range from 0.02 to 53 µg/g, and are elevated above reference creeks (range: 0.05 to 0.40 µg/g). PCB concentrations in the benthos are greater than the International Joint Commission objective for protection of wildlife consumers of aquatic

biota at 9 of the 11 Lyons Creek sites where tissue was collected. Sediment PCBs is strongly predictive of PCBs for 2 of the 4 invertebrate taxa collected (analysed without allowing gut clearance). Invertebrate PCB concentrations, expressed in toxic equivalent quantities for the dioxin-like PCBs, show at least 1 invertebrate taxon above the Canadian tissue residue guideline for the protection of wildlife consumers of aquatic biota at 9 of the 11 Lyons Creek sites; all 4 invertebrate taxa are above the guideline at 4 sites.

The decision-making framework indicates that management actions are required for one site (upstream of Highway 140) due to elevated sediment PCBs, toxicity, benthos alteration, and the potential for biomagnification. Management actions are also likely required for sites in the vicinity of Highway 140 (due to elevated PCBs observed in fish collected in this area). The reasons for sediment toxicity need to be determined for three sites.

The area from Ridge Road to Highway 140 is the most critical area of the creek. The highest sediment, invertebrate, (and fish) PCB concentrations occur in this area. Toxicity, altered benthic communities and potentially adverse effects due to biomagnification are also observed in this area.

APPENDIX 2 – Abstract

Milani, D., and L.C. Grapentine. 2006. The Assessment of Sediment PCB Contamination and Biological Impacts in Lyons Creek East (Niagara River Area of Concern)

Abstract

As part of the Niagara River Remedial Action Plan, tributaries of the river, including Lyons Creek, were identified as part of the Area of Concern. Lyons Creek was bisected when the Welland Canal was constructed. Lyons Creek East extends approximately 20 km from the Welland Canal to the Welland River. Previous studies have shown elevated levels of polychlorinated biphenyls (PCBs) in the sediment and detrimental effects on biota in the creek, specifically in the upper reaches of the creek. In the fall of 2002 and 2003, Environment Canada sampled 15 sites, with detailed sampling efforts focusing on the upper reaches, or the area between the Lyons Creek pumping station at the Welland Canal to Highway 140. Four neighbouring creeks, similar in morphology to Lyons Creek, were sampled as reference locations. Included in the assessment were analyses of: physico-chemical properties of the surficial sediment and overlying water; resident benthic invertebrate tissue; benthic invertebrate community structure, and; laboratory sediment toxicity tests. A risk-based, decision-making framework for the management of contaminated sediment, developed by the Canada-Ontario Agreement Sediment Task Group, was applied to the Lyons Creek study. The overall assessment of each Lyons Creek site was achieved by integrating the information obtained both within and among the following four lines of evidence: sediment chemistry, sediment toxicity, benthic community structure and the potential for PCB biomagnification. Concentrations of PCBs in the tissues of fish and wildlife receptors for Lyons Creek (Brown Bullhead, Carp, Bluegill, Largemouth Bass, Goldeneye, and Mink) were predicted by multiplying measured body concentrations in the resident invertebrates by relevant biomagnification factors obtained from a review of pre-existing studies. The predicted concentrations in the fish receptors were compared guidelines derived for the protection of fish consuming wildlife. Collections of resident sport fish in the creek by the Ministry of Environment provided ground-truthing of the model, as well as demonstrating actual bioaccumulation of PCBs in higher trophic level organisms.

The upper portion of the creek has the highest levels of PCBs and metals in the sediment, higher than sediment quality guidelines and higher than reference creek concentrations. The highest PCB concentrations in benthic invertebrates also occur in the upper reaches of the creek, and are elevated above reference creek concentrations. Acute toxicity is evident at 3 sites between the canal and Highway 140, and generally, Lyons Creek communities are similar to those at reference, with 1 site upstream of Highway 140 having a depauperate community compared to reference creeks. PCBs were predicted to bioaccumulate in higher trophic level receptors to concentrations that are not protective of adverse effects at between 2 and 11 sites. The upper area of the creek from Ridge Road to Highway 140 has the highest sediment, benthic invertebrate, and fish PCB concentrations; laboratory toxicity, altered benthic communities and potentially adverse effects due to biomagnification are all observed within this area. Management actions are recommended for 1 or 2 sites and the reasons for sediment toxicity need to be determined for 3 sites.

FRENCH VERSION

Milani, D., et L.C. Grapentine. 2006. « ».

Résumé

Dans le cadre du Plan d'assainissement de la rivière Niagara, des affluents de cette rivière, dont le ruisseau Lyons, ont été inclus dans le secteur préoccupant. Le ruisseau a été coupé en deux lors de la construction du canal Welland. Le ruisseau Lyons Est s'étend sur environ 20 km, du canal Welland à la rivière Welland. Des études antérieures ont permis de déceler une forte teneur en biphényles polychlorés (BPC) dans les sédiments et des effets délétères sur le biote du ruisseau, en particulier dans les tronçons les plus en amont. À l'automne de 2002 et 2003, Environnement Canada a échantillonné 15 sites en concentrant ses efforts de prélèvement détaillé dans les tronçons d'amont, soit entre la station de pompage du ruisseau Lyons, au canal Welland, et la route 140. Quatre ruisseaux avoisinants, morphologiquement semblables au ruisseau Lyons, ont fait l'objet de prélèvements à titre de sites de référence. L'évaluation comprenait l'analyse des propriétés physico-chimiques des sédiments superficiels et des eaux sus-jacentes, l'analyse de tissus d'invertébrés benthiques résidents, l'analyse de la structure des communautés d'invertébrés benthiques, ainsi que des tests de toxicité des sédiments en laboratoire. Un cadre décisionnel axé sur le risque pour la gestion des sédiments contaminés, mis au point par le Groupe de travail sur les sédiments de l'Accord Canada-Ontario, a été appliqué à l'étude sur le ruisseau Lyons. L'évaluation globale de chaque site du ruisseau Lyons s'est faite en intégrant l'information obtenue de quatre sources de données et des interactions entre ces sources : la chimie des sédiments, la toxicité des sédiments, la structure des communautés benthiques et le potentiel de bioamplification des BPC. On a prédit les concentrations de BPC dans les tissus des poissons et autres récepteurs animaux du ruisseau Lyons (barbotte, carpe, crapet arlequin, achigan à grande bouche, garrot et vison) en multipliant les concentrations observées dans l'organisme des invertébrés résidents par des facteurs de bioamplification obtenus par un examen d'études préexistantes. Les concentrations prédites dans les poissons récepteurs ont été comparées aux valeurs des lignes directrices pour la protection des espèces fauniques consommatrices de poisson. La collecte de poissons visés par la pêche sportive dans le ruisseau par le ministère de l'Environnement a permis une vérification sur le terrain du modèle, en plus de démontrer la bioaccumulation réelle de BPC dans des organismes de niveau trophique supérieur.

C'est dans la portion amont du ruisseau que la teneur des sédiments en BPC et en métaux est la plus élevée; elle dépasse les valeurs seuils des lignes directrices sur la qualité des sédiments et les concentrations observées dans les ruisseaux de référence. C'est aussi dans la portion amont du ruisseau qu'on observe les plus fortes concentrations de BPC chez les invertébrés benthiques; elles dépassent les concentrations observées dans les ruisseaux de référence. Une toxicité aiguë est évidente à 3 sites entre le canal et la route 140; en général, les communautés du ruisseau Lyons sont semblables aux communautés des ruisseaux de référence, un seul site en amont de la route 140 ayant une communauté appauvrie par rapport aux ruisseaux de référence. On prévoyait une bioaccumulation des BPC chez les récepteurs de niveaux trophiques supérieurs à des concentrations qui ne protègent pas les animaux des effets négatifs dans 2 à 11 sites. Le secteur d'amont du ruisseau, du chemin Ridge à la route 140, est celui où les concentrations de BPC dans les sédiments et dans les tissus d'invertébrés benthiques et de poissons sont les plus élevées; dans ce secteur, on observe la toxicité en laboratoire, l'altération des communautés benthiques et des effets négatifs potentiels dus à la bioamplification. Des mesures de gestion sont recommandées pour 1 ou 2 sites, tandis qu'il reste à déterminer les causes de la toxicité des sédiments à 3 sites.

ACKNOWLEDGEMENTS

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The GLSF, a component of the Great Lakes program, provides resources to demonstrate and implement technologies and techniques to assist in the remediation of Areas of Concern and other priority area in the Great Lakes. The report that follows addresses sediment quality issues in Lyons Creek East, Welland, Ontario (Niagara River Area of Concern). Although the report was subject to technical review, it does not necessarily reflect the views of the Sustainability Fund or Environment Canada.

Advice on the study design was provided by Roger Santiago and Janette Anderson (EC-OR), Cheriene Vieira and Rachael Fletcher (MOE) and Jocelyn Baker (NPCA). Field technical support was provided by Craig Logan, Sherri Thompson, Jennifer Webber, Jesse Baillargeon and Corrine Siefker (EC-NWRI), Rachael Fletcher and Steve Petro (MOE), and Al Vieira, Josh Diamond, Annie Michaud, and Stephanie Chase (NPCA). The organic analysis was performed by the MOE Dioxins laboratory (Etobicoke, ON). Station positioning and site maps were provided by the NPCA. Community structure sorting was completed by Sherri Thompson, species identification by Craig Logan, and toxicity tests by Jennifer Webber. The Niagara Seaway Authority provided land access for sample collection.

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ABBREVIATIONS, ACRONYMS AND SYMBOLS

adj	adjusted
AOC	area of concern
BMF	biomagnification factor
BSAF	biota-sediment accumulation factor
CI	confidence interval
CL	confidence limit
dw	dry weight
d/s	downstream
EC	Environment Canada
FCM	food chain multiplier
GL	Great Lakes
GLWQA	Great Lakes Water Quality Agreement
IJC	International Joint Commission
inv	invertebrate
LC	Lyons Creek
LEL	lowest effect level
max	maximum
min	minimum
MOE	Ministry of the Environment (Ontario)
PAH	polycyclic aromatic hydrocarbon
PCB	polychlorinated biphenyl
PEL	probable effect level
QA/QC	quality assurance/quality control
QEW	Queen Elizabeth Way Highway
RAP	remedial action plan
RC	reference concentration
rec	receptor
ref	reference
reg	regression
regr	regression
sed	sediment
SEL	severe effect level
TEF	toxic equivalency factor
TEQ	toxic equivalency unit
TN	total nitrogen
TOC	total organic carbon
TP	total phosphorus
TRG	tissue residue guideline
wt	weight
ww	wet weight
$[x]_i$	concentration of substance x in matrix i

1 INTRODUCTION

1.1 Background and Mandate

In the 1970s, 42 locations in the Great Lakes where the aquatic environment was severely degraded were identified as "problem areas" by the International Joint Commission (IJC). Of these, 17 are along Canadian lakeshores or in boundary rivers shared by the US and Canada. The IJC's Great Lakes Water Quality Board recommended in 1985 that a Remedial Action Plan (RAP) be developed and implemented for each problem area. The RAP approach and process is described in the 1987 Protocol to the *Great Lakes Water Quality Agreement* (GLWQA). The goal is to restore the "beneficial uses" of the aquatic ecosystem in each problem area, which were now called "Areas of Concern" (AOCs). Fourteen possible "impairments of beneficial use", which could be caused by alterations of physical, chemical or biological conditions in the area, are defined in Annex 2 of the GLWQA.

The Canadian government's commitment to the GLWQA was renewed in 2000 with the Great Lakes Basin 2020 (GL2020) Action Plan, under which the efforts of eight federal departments to "restore, conserve, and protect the Great Lakes basin" over the next five years were to be co-ordinated. Environment Canada's contribution included the funding of detailed chemical and biological assessments of sediments in Canadian AOCs. The National Water Research Institute (NWRI) was given the responsibility of conducting and reporting on these assessments.

Under the terms of reference for the NWRI's mandate, the Benthic Assessment of Sediment (BEAST) methodology of Reynoldson et al. (1995; 2000) was to be applied to the AOC assessments. To date, the methodology has involved evaluation of sediment contaminant concentration, laboratory toxicity, and benthic invertebrate community structure (see description below). Recent reviews of the BEAST framework have recommended the inclusion of an additional line of evidence – information on the bioaccumulation of contaminants liable to biomagnify (Grapentine et al. 2002). The study described in this document is an assessment of Lyons Creek East (Niagara River AOC) using these four lines of evidence.

1.2 Canada-Ontario Decision-Making Framework

It is recognized that to make decisions on sediment quality and the need to remediate, four components of information (in addition to knowledge on the stability of sediments) are required: sediment chemistry and grain size, benthic invertebrate community structure, sediment toxicity and invertebrate body burdens (Krantzberg et al. 2000). A risk-based, decision-making framework for the management of contaminated sediment was developed recently by the Canada-Ontario Agreement Sediment Task Group using the above components or lines of evidence. The framework was developed from the Sediment Triad (Long and Chapman 1985; Chapman 1996) and the BEAST (Reynoldson et al. 1995; 2000) frameworks and is described in Grapentine et al. (2002) and Chapman and Anderson (2005). The overall assessment of Lyons Creek sites is achieved by integrating the information obtained both within and among the above four lines of evidence. The biomagnification line of evidence is required in Lyons Creek due to the presence of PCBs and the objective is to determine if PCBs from sediments in Lyons Creek bioaccumulate in the tissues of resident benthic invertebrates and if PCBs could potentially be transferred through benthic invertebrates to fish, wildlife or humans.

1.3 Lyons Creek East

With the construction of the Welland Ship Canal bypass in the late 1960s / early 1970s, Lyons Creek was bisected into east and west portions. A condition of the canal's construction was that the portion of Lyons Creek downstream of the canal (Lyons Creek East) would have its flow maintained by pumping water from the canal into the creek at a rate that would maintain the original integrity of the creek. As part of the Niagara River RAP, tributaries of the river, including Lyons Creek, were identified as part of the AOC. Lyons Creek East, extends approximately 20 km from the Welland Canal to the Welland River. The Ministry of Natural Resources has defined the Lyons Creek East area as a significant wetland, consisting of a high diversity of fauna and flora, and meriting a high level of protection from detrimental impacts (Boyd et al., unpublished). A study looking at PCB aroclor patterns in the sediment in Lyons Creek West have lead to suggestions that PCB contamination may be of historical nature (prior to the canal being bisected), and subsequently lead to investigations of Lyons creek East. Studies dating from as early as 1978 (Acres 1978; MOE 1997; 1998; Boyd et al. unpublished) have identified the sediments in the upper reaches of Lyons Creek East to be highly

contaminated with metals and PCBs, and elevated nutrient levels have also been observed. Recent discharge into the Lyons Creek East from industrial sources includes Welland Pipe, which closed in 2003. Process water from the plant passed through an oil/water separator and then went to one of two settling lagoons before being discharged into the creek. Several oil spills have been recorded in the past in the creek (1988, 1989) (Boyd et al. unpublished). Studies have shown that sediments in the creek are toxic to benthic invertebrates and that PCB accumulates in the tissues of benthic invertebrate organisms as well as in fish tissues.

In September and October 2002, the National Water Research Institute (NWRI) of Environment Canada (and the Ministry of Environment) sampled Lyons Creek East to provide information on the degree of PCB contamination. Additional sampling was conducted in 2003 to further delineate the extent of PCB contamination between the pumping station at the Welland Canal and Highway 140 where the highest PCB concentrations were observed from the 2002 sampling. In 2003, sites where tissue was not collected in 2002 were revisited to obtain tissue and new reference creeks were sampled providing additional background conditions for similar unimpacted creeks. This report presents Environment Canada's results of these investigations and provides a spatial description of the state of the sediments in Lyons Creek and the degree of contamination.

2 METHODS

2.1 Sampling Design

Sampling stations were arrayed in a gradient design supplemented with reference sites. The mixed (gradient + control/potential impact) sampling design allowed several types of comparisons for assessing the distribution of PCBs in sediment and biota. The array of the sites also allowed a spatial analysis of PCB conditions, in which locations of elevated PCB in sediment, invertebrates and receptors (predicted from models) were identified. The location of stations were selected on the basis of (a) areas identified by an initial chemical screening survey performed by the Ministry of Environment in September 2002 as requiring further characterization (b) representing a wide range of PCBs levels in sediment (c) representing least contaminated/reference conditions in the area, and (d) overlapping locations of previous studies.

Sediment was obtained from the top 0 - 10 cm layer of creek bed as this layer includes the vertical home range of most benthic invertebrates.

2.2 Sample Collection and Handling

The survey was conducted 17 – 20 October 2002 and 1 – 9 October 2003. Sediment (for chemical and physical analyses and toxicity tests), overlying water and benthic community samples were collected at 21 sites in total (15 Lyons Creek sites and 6 reference creek sites). Benthic invertebrate tissue samples were collected at 11 of the 15 Lyons Creek sites and 4 of the 6 reference sites. Station co-ordinates are given in Table 1 and Lyons Creek site locations are shown in Figure 1. Reference creek locations are shown in Figure 2. Site locations were established using hand held Garmin GPS devices. Location co-ordinates were then verified using georeferenced digital orthographic imagery. Environmental variables measured at each site are provided in Table 2.

Overlying water

Prior to sediment collections, temperature, conductivity, pH and dissolved oxygen were measured in the water column approximately 0.5 m above the bottom using portable field meters (YSI, Orion). Water samples for alkalinity, phosphorus, nitrogen, and ammonia analyses were collected using a van Dorn sampler. Phosphorus samples (125mL) were preserved with 1mL of 30% sulphuric acid. Samples were stored at 4°C.

Benthic invertebrate community structure and sediment physico-chemical samples

A 40 cm × 40 cm mini-box core frame was used to obtain the benthic community and sediment chemistry samples. Benthic community samples were subsampled from the mini-box core frame using 10 cm (6.5 cm diameter) acrylic tubes. The content of the tubes were sieved through a 250-µm mesh screen and the residue on the screen preserved with 5% formalin for later identification. The remaining top 10 cm sediment inside the frame was removed, homogenised in a Pyrex dish, and allocated to containers for chemical and physical analyses of the sediment. At each of 4 sites where a mini-box core frame could not be used (due to site depth), three mini-ponar grabs were collected for benthic community structure analysis and one mini-ponar grab was collected for chemical and physical properties of the sediment. Each community structure

sample was sieved in its entirety and the residue preserved as described above. Samples were stored at 4°C. Benthic community samples were transferred to 70% ethanol after a minimum of 72 hours in formalin.

Benthic invertebrate tissue and sediment organic contaminant samples

A mini-ponar sampler was used to collect the resident benthic invertebrates for tissue organic contaminant analysis. At each site, enough sediment was collected to fill 2 68-L plastic tubs (approximately 10-15 mini-ponars per tub). A small scoop of sediment (top 10 cm) was taken from each ponar grab and set aside in a glass tray. This was repeated until each tub was approximately 2/3 full. Ample site water was added to each tub. The sediment in the glass tray was homogenized and distributed to a pre-cleaned glass amber jar for organic contaminant analysis. Sediment samples were frozen (-20°C).

Invertebrates were removed from the sediment by wet sieving (using water pumped from the Welland Canal) the sediment through 12" stainless steel sieves (500-μm mesh). Macroinvertebrates collected on the sieve were sorted into separate taxa in glass trays using stainless steel instruments. Biota were rinsed with reverse osmosis water, placed in pre-weighed and pre-cleaned (20% HCL, hexane rinsed) 5-mL scintillation vials, and weighed. A layer of parafilm was placed between vial and cap and the biota was frozen (-20°C). Invertebrate samples were later freeze-dried and reweighed. The wet:dry ratios were used in converting invertebrate tissue contaminant concentrations from dry to wet weight values (see Section 2.6.1).

Several distinct invertebrate taxa were collected from each location. Analyses of organic contaminants were performed on samples composited from organisms within each taxon (i.e., taxa were analyzed separately). Due to sample size requirements and time constraints, taxa of similar functional feeding groups were combined. Amphipods and isopods were combined (hereafter referred to as 'amphipod') and damselflies and dragonflies were combined (hereafter referred to as 'odonate'). Invertebrates were not allowed time to clear sediment from their guts because predators consume whole organisms. PCBs associated with sediment, as well as that incorporated into tissues, are potentially available for transfer through the food chain.

Stainless steel sieves and instruments were detergent washed between stations. Homogenizing and sorting trays and scoops were detergent washed, rinsed in 20% HCl, and rinsed with hexane.

Toxicity test samples

Five mini-ponar grabs were collected per site for the laboratory toxicity tests (approximately 2 L sediment per replicate). Each of the five sediment grabs was placed in separate plastic bag, sealed, and stored in a bucket at 4°C.

2.3 Sediment, Biota and Water Physico-Chemical Analyses

Organic contaminants

Analysis of polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs) and organo-chlorines (OCs) was performed on sediment and biota samples by the Laboratory Service Branch of the MOE (Etobicoke, ON), following MOE standard methods (MOE 1993a; 1994; 2003a).

Lipids

Lipid analysis was performed on the biota samples collected in 2003. (Sample size was not sufficient to allow for lipid analysis of the 2002 benthic invertebrate tissue.) Lipids were analyzed by the MOE (Etobicoke, ON) following MOE standard methods.

Overlying water

Analyses of alkalinity, total phosphorus, nitrates/nitrites, ammonia and total Kjeldahl nitrogen (TKN) in water samples were performed by the Environment Canada's National Laboratory for Environmental Testing (NLET) (Burlington, ON) by procedures outlined in Cancilla (1994) and NLET (2000).

Sediment trace metals and nutrients

Freeze dried sediment was analysed for trace elements (hot aqua regia extracted), major oxides (whole rock), loss on ignition (LOI), total organic carbon (TOC), total phosphorus (TP), and total nitrogen (TN) by Caduceon Environmental Laboratories (Ottawa, ON) using standard techniques outlined by the USEPA/CE (1981) or in-house methodologies.

Sediment particle size

Percents gravel, sand, silt, and clay were performed by the Sedimentology Laboratory at NWRI (Burlington, ON) following the procedure of Duncan and LaHaie (1979).

2.4 Toxicity Tests

Four sediment toxicity tests were performed: (1) *Chironomus riparius* 10-day survival and growth test (2) *Hyalella azteca* 28-day survival and growth test (3) *Hexagenia* spp. 21-day survival and growth test, and (4) *Tubifex tubifex* 28-day adult survival and reproduction test. Sediment handling procedures and toxicity test methods are described elsewhere (Borgmann and Munawar 1989; Borgmann et al. 1989; Krantzberg 1990; Reynoldson et al. 1991; Reynoldson et al. 1998). All tests passed acceptability criteria based on percent control survival in culture sediment before including in a data set: i.e., ≥ 80% for *H. azteca* and ≥70% for *C. riparius* (USEPA 1994; ASTM 1995); ≥80% for *Hexagenia* spp., and ≥75% for *T. tubifex* (Reynoldson et al. 1998).

Water chemistry variables (pH, dissolved oxygen (mg/L), conductivity (µS/cm), temperature (° C), and total ammonia (mg/L)) were measured in each replicate test beaker on day 0 (start of test) and at completion of the test. Tests were run under static conditions in environmental chambers at 23°C ±1 °C, under a photoperiod of 16L: 8D and an illumination of 500 – 1000 lux, with the exception of the *T. tubifex* test which was run in the dark.

Hyalella azteca 28- day survival and growth test

The test was conducted for 28 days using 2 – 10 day old organisms. On day 28, the contents of each beaker were rinsed through a 250-µm screen and the surviving amphipods counted. Amphipods were dried at 60 °C for 24 hours and dry weights recorded. (Initial weights were considered zero.)

Chironomus riparius 10- day survival and growth test

The test was conducted for 10 days using first instar organisms. On day 10, the contents of each beaker were wet sieved through a 250-µm screen and the surviving chironomids counted.

Chironomids were dried at 60 °C for 24 hours and dry weights recorded. (Initial weights were considered zero.)

Hexagenia spp. 21-day survival and growth test

The test was conducted for 21 days using pre-weighed nymphs (between 5 - 8 mg wet weight/nymph). On day 21, the contents of each jar were wet sieved through a 500-µm screen and surviving mayfly nymphs counted. Nymphs were dried at 60 °C for 24 hours and dry weights recorded. Initial mayfly wet weights were converted to dry weights using the following equation from a relationship for nymphs from the Ecotoxicology Lab that was previously determined by regression analysis: Initial dry weight = [(wet weight + 1.15)/ 7.35]. Growth was determined by final dry weight minus initial dry weight.

Tubifex tubifex 28-day reproduction and survival test

The test was conducted for 28 days using sexually mature worms (gonads visible). On day 28, the contents of each beaker were rinsed through a 500-µm and 250-µm sieve sequentially. The number of surviving adults, full cocoons, empty cocoons, and large immature worms were counted from the 500-µm sieve and the numbers of small immature worms were counted from the 250-µm sieve. Survival and reproduction were assessed using four endpoints: Number of surviving adults, total number of cocoons produced per adult, percent cocoons hatched, and total number of young produced per adult.

2.5 Benthic Invertebrate Taxonomic Identification

Invertebrates in the benthic community samples were sorted, counted and identified to the family level at the Invertebrate Laboratory at NWRI (Burlington, ON). Slide mounts were made for Oligochaetae and identified to family using high power microscopy.

2.6 Data Analysis

2.6.1 Potential for biomagnification

PCB distribution in sediment and biota

Levels of PCBs in Lyons Creek were compared to those in reference creeks. Sites in which concentrations of total PCBs in sediment ($[PCB]_{sed}$) and invertebrates ($[PCB]_{inv}$) were significantly elevated above background levels for the study area were identified by comparing

test site concentrations to the upper 99th percentile for the reference sites. For the benthic invertebrates, this was done separately each invertebrate taxon collected.

Relationships between concentrations of total PCBs in sediment and invertebrates were determined using regression analysis, separately for each invertebrate taxon. The approach was used to estimate the degree to which PCBs in invertebrates is predictable from PCBs in sediment, with and without environmental covariables. Simple linear regression (ordinary least squares) was used for the single predictor ($[PCB]_{sed}$) model. "Best subset" multiple linear regression (Draper and Smith 1998; Minitab 2000) was used for the fitting of multiple predictor models. Environmental variables expected to potentially influence uptake of PCB from sediment by biota such as sediment concentrations of total organic carbon, phosphorus, nitrogen, iron, and manganese; sediment particle size fractions of sand, silt and clay; overlying water conductivity, dissolved oxygen, pH, temperature and nutrients (nitrogen, phosphorus, nitrates/nitrites) were included in the models. To increase normality of data distributions and linearity of relations between variables, some data were transformed: log(x) for PCBs in sediment and invertebrates; log(x) for nutrients, iron and manganese in sediment; and arcsine-square root(x) for the particle size fractions. Normality and linearity of the water column data were not generally improved by transformations, so these were analyzed untransformed.

All models fitted to the data included $[PCB]_{sed}$ as a free predictor (i.e., it was not forced to be in the model). The specific null hypothesis of interest was that "the effect of $[PCB]_{sed}$ on $[PCB]_{inv} = 0$, after accounting for effects of other predictors". For the best subset regressions, models were fitted for all combinations of predictors. Determination of the "best" model was based on several criteria (in roughly decreasing order of importance):

- Maximum $R^2_{adjusted}$;
- Significance of partial F-tests (= t-tests) for predictors (especially $[PCB]_{sed}$);
- Significance of F-test for regression;
- Variance inflation factors (VIFs) for predictors < 10;
- Homoscadastic and normally distributed residuals; and
- Mallow's C_p statistic not >> number of predictors.

Lack-of-fit tests for curvature in response-predictor relationships and interactions between predictors were performed and examined for nonsignificance. Observations having large standardized residuals or large influence on the regression were also considered in model evaluations. The best model was identified based on the overall meeting of these criteria. Both single and multiple predictor models were then examined for the degree to which $[PCB]_{sed}$ predicts $[PCB]_{inv}$, as indicated by the significance of the *t*-test of the coefficient for $[PCB]_{sed}$.

Calculation of receptor tissue PCB concentrations

The concentration of PCBs in selected trophically linked receptor species (i.e., consumers of benthic invertebrates and their predators) was predicted by multiplying measured body concentrations in the resident invertebrates by the food chain multiplier relevant for the receptor:

$$C_{rec} = FCM \times C_{inv}$$

where:

C_{rec} = mean contaminant concentration in the consumer (receptor) species

C_{inv} = mean contaminant concentration in invertebrates

FCM = food chain multiplier

The FCM represents the cumulative biomagnification of a substance from one trophic level to a higher trophic level (USEPA 1997c). Whereas a BMF applies to only one trophic level transfer, a FCM refers to one or more, and may be a multiple of more than one BMF. Thus, FCM = $BMF_1 \times BMF_2 \times BMF_3 \times \ldots \times BMF_n$, where 1, 2, 3,..., n are transfers of one trophic level. Biomagnification factors were literature-derived and receptor PCB concentrations were predicted on a *total* PCB basis. Table 3 shows the BMFs and FCMs used to calculate C_{rec} values. For the Brown Bullhead, carp, Goldeneye, and Bluegill, the BMF = FCM, since they are trophic level 2 receptors. The FCMs for transfer from benthic invertebrates to the mink and bass are estimated by multiplying the BMFs for the serial steps. Low, medium and high FCM values are obtained from use of all minimum, all medium or all maximum estimates for each BMF. For the sunfish, bass, and mink, it is recognized that they could be either trophic level 2 or 3 (sunfish), or trophic level 3 or 4 (bass and mink). However, BMF values were not obtained for the higher of the two

trophic levels for these receptors. A review of information on BMFs was conducted using typical methods of electronic database and chain-of-citation searches. Details on the methods and the results of the review are described in Appendix A.

Invertebrate PCB concentrations used in the predictions of PCB in receptors were the observed $[PCB]_{inv}$ values for taxa collected from the site. These were used to obtain minimum and maximum observed $[PCB]_{inv}$ for the taxa collected from the site. "Medium" $[PCB]_{inv}$ for the site was calculated as the mean of the values. Since fish contaminant data are reported for the most part on a wet weight basis, and the guidelines used in this study are also based on wet weights, PCB concentrations in invertebrates were converted to wet weight values. Biota comprised on average 88.0% water (range 81.7 to 91.7%). The ratio of wet to dry weight was determined for each individual sample submitted for analysis (rather than using an overall average ratio for each taxon). Wet weights were determined using the following conversion:

$[PCB]_{inv}$ (µg/g dry weight) / (ratio of wet: dry weight) = $[PCB]_{inv}$ (µg/g wet weight)

Total PCB concentration in each invertebrate taxon based on wet weight is provided in Appendix B; Table B1.

For each site, minimum, intermediate and maximum concentrations of PCBs for each receptor were predicted by:

$[PCB]_{rec} = FCM \times [PCB]_{inv}$,

using corresponding low, medium and high $[PCB]_{inv}$ and FCMs. From the available values, the lowest and the highest BMFs were used for the minimum and maximum prediction, the mean of the values was used for the intermediate prediction. The predicted PCB concentrations in receptors are generic in that they are not specific to particular tissues.

If the predicted contaminant concentration in the receptor exceeded the IJC objective for PCBs and the maximum reference concentration, a potential risk of adverse effects due to biomagnification was concluded. Alternatively, if the predicted contaminant concentration in the

receptor was less than the guideline or the maximum reference concentration, no potential risk was concluded.

2.6.2 Sediment toxicity

BEAST analysis

The BEAST is a predictive approach for assessing sediment quality using multivariate techniques (Reynoldson et al. 1995; 2000; Reynoldson and Day 1998). The approach utilizes data from nearshore reference sites that were sampled from the Laurentian Great Lakes over a three-year period. Information includes benthic community structure (the type and number of invertebrate taxa present), selected habitat variables, and responses (survival, growth and reproduction) of four benthic invertebrates in laboratory toxicity tests. The reference sites establish normal conditions for selected endpoints, and determine the range of 'normal' biological variability. As a result, expected biological conditions are predicted by applying relationships developed between biological and habitat conditions.

Toxicity data were analysed using by "Semi-strong" hybrid multidimensional scaling (HMDS, Belbin 1993) with Euclidean distance site × site association matrices calculated from standardized data. Principal axis correlation (Belbin 1993) was used to identify relationships between habitat attributes and toxicity responses. Significant toxicity test endpoints and environmental attributes were identified using Monte-Carlo permutation tests (Manly 1991). Test sites were assessed by comparison to confidence bands (90, 99 and 99.9% probability ellipses) derived from reference sites. HMDS, principal axis correlation, and Monte-Carlo tests was performed using the software PATN (Blatant Fabrications 2001). Probability ellipses were produced using the software SYSTAT (Systat Software Inc. 2002).

Sediment toxicity and contaminant relationships

The BEAST assessment does not incorporate any information on organic contaminants in the sediment (organic contaminant concentrations were not measured in Great Lakes reference sediments). Therefore, additional analyses of relationships between sediment toxicity and contaminant concentrations for Lyons Creek sites were conducted to aid in identifying causes of toxicity (e.g., organic contaminants, inorganic compounds, sediment grain size).

Relationships between sediment toxicity and sediment contamination for Lyons Creek sites were assessed graphically and by regression analysis. Initially, to examine general and dominant patterns in the data, comparisons between the toxicity responses and contaminant conditions were made based on integrative, compound variables (from either summation or multivariate ordination of measurement variables). After this, to better detect less dominant (though significant) relationships between two or a few variables, analyses were conducted using the original measurement variables (i.e., toxicity endpoints and concentrations of individual compounds).

The sediment toxicity data for Lyons Creek sites were ordinated again by HMDS, as a single group and without the reference site data. To identify and relate the most important of the toxicity endpoints to the HMDS axes, principal axis correlation was conducted. Extractable concentrations in sediment of 9 metals (As, Cd, Cr, Cu, Fe, Hg, Ni, Pb, and Zn) were ordinated by principal components analysis (PCA). Data for all variables were log(x)-transformed. The eigenanalysis was performed on the correlation matrix. Total PCB and PAH variables were integrated by summing the concentrations of the individual congeners.

Both the integrated descriptors of sediment toxicity (axes scores from the HMDS) and individual toxicity endpoints (arsine square root(x)-transformed for survival endpoints and log(x)-transformed for growth and reproduction endpoints) were plotted against the integrated contaminant descriptors as well as individual log(x)-transformed sediment contaminants, sediment nutrient variables, and grain size. To determine whether toxicity was better explained by joint consideration of the contaminant descriptors, multiple linear regression involving the contaminant descriptors as predictors was calculated with each toxicity descriptor as the response variable. The degree to which individual sediment variables account for toxicity was assessed by fitting regression models using "best subset" procedures (Draper and Smith 1998; Minitab 2000). Models were fitted for (a) all combinations of metals (b) all combinations of nutrients and grain size (c) total PCBs, PAHs, and then (d) all combinations of the best predictors from the three groups (This procedure was used to avoid computational difficulties arising from working with 18 predictors simultaneously.) The best models were those having maximum explanatory power

(based on $R^2_{adjusted}$), minimum number of nonsignificant predictors, and minimum amount of predictor multicollinearity.

2.6.3 Benthic alteration

The BEAST method has been used to assess the condition of benthic invertebrate communities and at a number of Great Lakes AOCs, e.g., Collingwood Harbour, St. Lawrence River (at Cornwall), Bay of Quinte, Peninsula Harbour and Hamilton Harbour (Reynoldson et al. 1995; Reynoldson 1998; Reynoldson and Day 1998; Milani and Grapentine 2004; 2005; 2006). A limitation to the use of the method, however, is that it can only be applied with confidence to test sites within the range of habitats and geographic areas contained within the reference database. The current database consists mainly of nearshore lake sites and does did not contain habitat characteristics and community structure data for small streams or creeks in Southern Ontario. The BEAST analysis is more sensitive to changes in abundance than richness, and typically species richness is greater in riverine or stream systems. Therefore, this reference condition approach using lake reference sites is not suitable to Lyons Creek community assessment. (The BEAST approach is applicable for the Lyons Creek toxicological assessment since species responses (ten test endpoints) at reference sites were not found to be significantly correlated with any habitat characteristic, and therefore the range of response in each endpoint represents the natural variability.)

Using the mean values of abundance counts for invertebrate taxon, the biological structure of the data was examined using ordination (HMDS) applied to a Bray-Curtis distance matrix. Analyses were performed at the family level, as this taxonomic detail is shown to be sensitive for the determination of stress (Reynoldson et al. 2000). Principal axis correlation (Belbin 1993) was used to identify significant families and habitat attributes. Using the ordination axes scores from the HMDS, sites were also compared by Analysis of Variance with adjustments for covariates (ANCOVA) using general linear model (Minitab 2000). Comparisons to control using the ordination axes scores were made using Bonferroni's and Dunnett's simultaneous test. Pairwise comparisons of the means from all sites were performed using Tukey's test. Site comparisons were also made using taxa richness and log(x)-transformed abundances of the following major

taxon groups found in Lyons Creek: Tubificidae, Chironomidae, Hyalellidae, Gammaridae, Caenidae and Coenagrionidae.

2.7 Quality Assurance/Quality Control

Field

One randomly chosen site (LC29) was designated as a QA/QC station, where triplicate sediment and overlying water samples were collected for determination of within-site and among-sample variability.

Laboratory

The MOE organics laboratory (Etobicoke, ON) conducted determinations of organic contaminants in sediment and benthic invertebrates. Quality control evaluation for these procedures included evaluation of matrix spike recoveries. Matrix spikes were performed on every sample to determine PAH recoveries.

Caduceon Environmental Laboratory (Ottawa, ON) analyzed sediment for trace metals, major oxides and sediment nutrients. Quality control procedures involved control charting of influences, standards, and blanks. Reference material was used in each analytical run. Calibration standards were run before and after each run. Blanks and reference standards were run 1 in 20 samples and duplicates were run 1 in 10 samples.

Benthic community sorting efficiency

To evaluate control measures for benthic invertebrate enumeration (on a monthly basis), a previously sorted sample was randomly selected, re-sorted, and the number of new organisms found counted. The percent of organisms missed (%OM) was calculated using the equation:

$$\% \text{ OM} = \text{number of organisms missed} \div \text{total organisms found} \times 100$$

A desired sorting efficiency is %OM < 5%. If the %OM was > 5%, two more replicate samples were randomly selected and the %OM calculated. The average %OM was calculated based on

the three samples re-sorted, and represents the standard sorting efficiency for that month. The average %OM is based on only one replicate sample if %OM is < 5%).

3 RESULTS

3.1 Sediment and Water Physico-Chemical Properties

Sediment organic contaminants

Total [PCB] at Lyons Creek sites range from 0.02 to 12.5 µg/g; reference site [PCB] range from 0.003 to 0.016 µg/g (Table 4). The highest [PCB] is at site LC03, located immediately downstream of the former Welland Pipe outfall followed by site LC12 (7.4 µg/g), located approximately 500 m upstream of Highway 140. The severe effect level (SEL) for total PCBs (530 x %TOC) is not exceeded at any site. Overall, total PCBs decrease with distance downstream from the pipe (LC03) with the lowest concentration at the site farthest downstream (LC38) and the upstream site (LC01) (Figure 3). With the exception of sites LC01 and LC38, which exceed the maximum reference concentration marginally, all Lyons Creek sites exceed the maximum reference site concentration (indicated by the green dotted line) by between 1 to 3 orders of magnitude (Figure 3). PCB congener data are provided in Appendix C; Table C1.

The isomeric composition of Lyons Creek and reference sediment is shown in Appendix C; Figure C1. Black Creek reference sites are most different from the rest of the sites, consisting primarily of the trichlorobiphenyls (75 to 100%). Lyons Creek sites consist predominantly of the tetra- (30 to 45%) and pentachlorobiphenyls (25 to 35%), with also hexa- and heptachlorobiphenyls present. Reference sites TC40 and UC01 have the highest percentage of the hexa- and heptachlorobiphenyls. The percentage of coplanar to total PCBs ranges from 3 to 10% at Lyons Creek sites and from 0 to 2% at reference sites (Appendix C; Figure C2). Overall, there is an increase in percentage of coplanar PCBs with distance downstream with a spike at sites LC06 and LC22. Coplanar PCBs are very significantly related to total PCBs ($r^2 = 0.942$, $p = \leq 0.001$) (Appendix C; Figure C3).

Sediment PAH and organo-chlorine (OC) pesticide concentrations are provided in Appendix C; Table C2. Total PAHs range from 0.46 to 62.94 µg/g (median 1.17 µg/g) at Lyons Creek sites;

reference site [PAH] range from 0.40 to 1.08 µg/g (median 0.42 µg/g). Total PAHs follow the same pattern as seen with PCBs, with the highest concentration at site LC03 and decreasing concentrations downstream from the Welland Pipe outfall. Site LC03 exceeds the maximum reference site concentration by between 1 to 2 orders of magnitude. The SEL for total PAHs (10000 x %TOC) is not exceeded at any site. The only OC pesticide present in any significant concentration is pp-DDE, which exceeds the LEL criteria of 5ng/g at 13 of the 15 Lyons Creek sites (maximum [DDE] at LC03, 340 ng/g).

Overlying water

Conditions of overlying water 0.5 m above the sediment are similar across Lyons Creek sites for most variables measured (Appendix C; Table C3). Nitrates/nitrites (NO_3/NO_2) and temperature are highest in the upper reach of the river. Reference sites have higher alkalinity, conductivity, and nitrogen (TKN) than Lyons Creek sites. Black Creek reference sites show dissimilarities in NO_3/NO_2 compared to the other reference sites and Lyons Creek sites, with 1 to 2 orders of magnitude differences noted. The range of variables across Lyons Creek sites are: alkalinity 18 mg/L, conductivity 120 µS/cm, dissolved oxygen 3.8 mg/L, NH_3 0.07 mg/L, NO_3/NO_2 0.27 mg/L, pH 2.4, TKN 0.32 mg/L, phosphorus (TP) 0.03 mg/L, and temperature 9.9 °C.

Sediment particle size

Particle size data for Lyons Creek sediment are provided in Appendix C; Table C4. Lyons Creek sediment consists mainly of fines; silt ranges from 33.5 to 83.1% (median 52.5%), and clay ranges from 16.6 to 63.6% (median 45.2%). Overall, reference creek sediments have a slightly higher clay content than Lyons Creek sediment, ranging from 38.4 to 71.1% (median 49.3%) and a lower silt content, ranging from 20.3 to 43.6% (median 34.8%). Reference site BLC02 (Black Creek) has the highest gravel content (6.8%). With the exception of site LC03 (17.9% sand), reference sediment is coarser than Lyons Creek sediment, with sand content ranging from 5.0 to 18.0% (median 15.3%) at reference sites, and from 0.3 to 17.9% (median 1.7%) at Lyons Creek sites.

Sediment nutrients

Total organic carbon (TOC) at Lyons Creek sites ranges from 1.9 to 10.7% (median 5.1%), total nitrogen (TN) ranges from 2480 to 8390 µg/g (median 5030 µg/g), and total phosphorus (TP) ranges from 892 to 3070 µg/g (median 1460 µg/g) (Appendix C; Table C5). Overall, TP concentrations are lower at reference sites (range: 428 to 1040 µg/g, median: 779 µg/g), while TOC and TN at reference sites are similar to Lyons Creek sites (TOC: range 3.2 to 10.6%, median 5.9%, TN: range 1970 to 8420 µg/g, median 4255 µg/g). Total nitrogen exceeds the SEL at 9 of 15 Lyons Creek sites and at 2 of 6 reference sites. The SEL is exceeded for TOC at downstream site LC38 and reference site BEC02 and for TP at LC03.

Sediment trace metals

Overall, most trace metals are higher at Lyons Creek sites than reference sites, especially for zinc (Zn), which ranges from 126 to 7969 µg/g (median 657 µg/g) at Lyons Creek sites, and from 81 to 166 µg/g at reference sites (median 108 µg/g) (Appendix C; Table C5). Site LC03 is consistently highest in most metals. The SEL is exceeded for As, Cu, Ni, and Zn at LC03 and for Zn at LC08, LC10, LC12 and LC14.

3.2 Biomagnification Potential

3.2.1 Benthic invertebrate tissue PCB levels

The lowest [PCB] are found in the reference creek benthos (range 0.05 to 0.40 µg/g, mean 0.18 µg/g), followed by benthos collected from the upstream site LC01 (range 0.23 to 0.68 µg/g, mean 0.41 µg/g) (Table 5, Figure 4). Total $[PCB]_{inv}$ is ~1 to up to ~2 orders of magnitude higher at Lyons Creek sites, ranging from 0.02 to 52.6µg/g; LC12 has the highest concentration (mean 17.4 µg/g), followed by LC17 (mean 3.5µg/g) and LC03 (mean 2.7 µg/g). All four taxa could not be analysed at all sites due to insufficient tissue quantity. There was insufficient oligochaete tissue for the Black Creek reference sites and insufficient chironomid tissue for LC01 and LC03. (Benthic invertebrates were not collected from Beaver Creek (BEC) and sites LC06, LC10, LC22 and LC23.) On a whole-body, uncleared-gut basis, the amphipods accumulate more PCBs at 7 of the 11 Lyons Creek sites (most sites between the pumping station and the railway); oligochaetes accumulate the most PCBs at 3 sites including LC12. A complete list of benthic invertebrate PCB congener concentrations is provided in Appendix D; Table D1.

The isomeric composition of benthic invertebrates is shown in Appendix D; Figure D1. Taxa collected from the reference creeks consist primarily of the lower chlorinated biphenyls (tri- and tetrachlorobiphenyls). The isomeric composition of taxa collected from Black Creek is similar to that seen in the sediment samples. The higher chlorinated biphenyls occur in taxa collected from sites generally between LC08 to LC29. Site LC19 has the highest percentage of penta- to hepta-chlorobiphenyls for chironomids and amphipods, whereas LC14 has the highest percentage for the oligochaetes and odonates. Overall, the isomeric composition of the amphipods is most similar to that seen in the sediment samples.

Comparison of [PCB] to IJC tissue objective and reference maximum

Total [PCB] in benthic invertebrates (wet weight) is shown in Figure 5 and in Table B1; Appendix B. The green dotted lines in Figure 5 represent the maximum reference concentration for each taxa and the red line is the IJC tissue objective for the protection of wildlife consumers of aquatic species (0.1 µg/g ww, IJC 1989).

Chironomid – No data are available for sites LC01 and LC03. Six sites are above the IJC tissue objective for PCBs (sites LC12 to LC19) and all sites are above the maximum reference site concentration. The highest PCB accumulation in the midges is at LC12 and LC17, which show very similar concentrations. Reference and Lyons Creek [PCB] range from 0.012 to 0.024 µg/g and from 0.072 to 0.465 µg/g, respectively (Appendix B; Table B1).

Amphipod – Eight sites are above the IJC objective (sites LC03 to LC19) and all test sites are above the maximum reference concentration except LC38. The highest PCB accumulation is at LC12, followed by LC17 and LC03, where amphipods show similar concentrations. Reference and Lyons Creek [PCB] range from 0.006 to 0.025 µg/g and from 0.010 to 1.386 µg/g, respectively (Appendix B; Table B1). Overall, amphipods accumulate the highest concentrations of PCBs.

Oligochaete – No data are available for reference sites BLC01 and BLC02. Six sites are above the IJC objective (sites LC03, LC12, LC16 to LC18, LC29) and all sites are above the maximum reference concentration except LC01 and LC08. Oligochaetes accumulated the highest PCBs at

LC12. Reference and Lyons Creek [PCB] range from 0.008 to 0.043 µg/g and from 0.033 to 6.149 µg/g, respectively (Appendix B; Table B1).

Odonate – One site (LC12) is above the tissue objective and LC12 and LC16 are above the maximum reference concentration. Reference and Lyons Creek [PCB] are similar, ranging from 0.012 to 0.036 µg/g at reference sites and from 0.003 to 0.055 µg/g at Lyons Creek sites (Appendix B; Table B1). Overall, odonates accumulated the least amount of PCBs of the four taxa.

Coplanar PCBs

Invertebrate [PCB], expressed in toxic equivalent units (TEQ), is shown in Figure 6. The red line is the CCME avian tissue residue guideline (TRG), which in the current study applies to the diving duck receptor (the only wildlife receptor in the study that would feed directly on benthic invertebrates). The avian TRG, derived by Environment Canada, is 2.4 ng TEQ·kg^{-1} diet ww (CCME 2001). The mammalian TRG of 0.79 ng TEQ·kg^{-1} diet ww, while lower, was not used in this case as there is not a direct feeding relationship from invertebrates to the mammal receptor (mink). The TEQ is the summation of 12 co-planar PCB congener's toxic equivalency factor (TEF) × [coplanar PCB]$_{inv}$. The TEFs were developed to compare toxicities of various PCB congeners relative to the most potent PCB inducer in the cytochrome enzyme system (2,3,7,8-TCDD), and based on the World Health Organization, range from 0.00001 to 0.1 for avian (Van den Berg et al. 1998). All Lyons Creek sites except LC01 and LC03 have at least one taxon with a [TEQ] well above the TRG (Figure 6). Sites where all four taxa have [TEQ] above the TRG include LC14, LC16, LC18 and LC38. The high [TEQ] observed at sites LC14, LC16, LC18, LC29 and LC38 are due to the high concentration of PCB 126 in the benthos samples. PCB 126 (as well as PCB 81) has the highest TEF (0.1). The high [TEQ] for site LC12 is due primarily to the high concentration of PCB 105 and PCB 118 in the amphipod and oligochaete samples. No reference site [TEQ] is above the TRG. The percentage of coplanar to total PCBs varies among taxa and sites, with an overall range in biota from 0 to 17% at Lyons Creek sites and from 0 to 12% at reference sites (Appendix D; Figure D2). The pattern observed for sediment (overall increase with distance downstream) is not seen in the benthos. The highest percentage of coplanar PCBs to total PCBs is at LC14 (chironomids – 17%), and for reference sites is BLC01

(odonates – 12%). The highest coplanar PCBs are found in the odonates at 45% of Lyons Creek sites followed by the chironomids at 36% of Lyons Creek sites. (The odonates have the lowest total PCBs at all Lyons Creek sites – see Figures 4 and 5.) Coplanar PCBs are significantly related to total PCBs for all taxa (r^2 = 0.853 to 0.999, p = ≤ 0.001).

3.2.2 Biota-sediment accumulation factors

Biota-sediment accumulation factors (BSAFs) for total PCBs are shown for each taxon in Appendix D; Figure D3 and Table D4. Mean percent lipids (% dry mass) are: amphipods 5.7%, chironomids 13.8%, oligochaete 17.9%, and odonates 7.6%. Lyons Creek BSAFs are lower than reference site BSAFs and are highest overall for the amphipods for Lyons Creek sites (excluding outliers) and overall highest for the odonates for reference sites. Lyons Creek BSAF ranges are: amphipods 0.3 to 10.6 (median 2.4), chironomids 0.04 to 46.3 (median 0.7), oligochaetes 0.01 to 34.6 (median 0.9), odonates 0.001 to 12.1 (median 0.2). Reference creek BSAF ranges are: amphipods 5.1 to 61.9, chironomids 5.2 to 24.2, oligochaetes 1.4 to 10.3, and odonates 5.7 to 85.3. For the oligochaetes, there are only two data points for the reference sites (sites TC40 and UC01). Percentiles could not be computed in this case and therefore the set of data points is not shown in Figure D3.

3.2.3 Relationships between PCB concentrations in tissue and sediment

Concentrations of total PCBs in each invertebrate taxon vs. total PCBs in sediment are plotted in Figure 7, with fitted regression lines using sediment [PCB] alone as the predictor. For the chironomid and amphipod, the slopes are significant ($P \leq 0.05$) and the R^2_{adj} values are 0.625 and 0.874, respectively (Table 6). Predictions of $[PCB]_{inv}$ are moderately improved for both taxa with pH in the model (Table 6), bringing the R^2_{adj} values to 0.749 and 0.918 for the chironomid and amphipod, respectively. In both cases $[PCB]_{sed}$ is the strongest predictor ($P \leq 0.001$) and the coefficients for pH are positive. For the oligochaete, the addition of pH (positive regression coefficient), phosphorus in the overlying water (positive regression coefficient) and sand (negative regression coefficient) result in a significant slope, with an R^2_{adj} value of 0.783. For the odonate, the slope is not significant, and no additional predictors improve the model.

3.2.4 Predictions of total PCBs in receptors

Receptors of concern for Lyons Creek

Knowledge of the food web structure of the study area site was needed to determine relevant receptor species (fish, bird, mammal). The identified receptors determined the biomagnification factors (BMFs) to use for predicting receptor total PCB concentrations and the appropriate criteria (e.g., guidelines for protection of wildlife consumers of aquatic biota; human health guidelines for protection from fish consumption) for comparison. Based on generic food webs for the Great Lakes (e.g., Diamond et al. 1994; Russell et al. 1999), information on fauna resident in Lyons Creek East (Boyd et al. unpublished; MOE 2003b) and guidelines from Environment Canada (2001), receptors representative of three trophic levels were selected for biomagnification modelling:

Trophic Level 1 - Benthic invertebrates

Amphipod/Chironomid/Oligochaete/Odonate

Trophic Level 2 - Benthivorous fish

Brown Bullhead/Carp Total PCB levels are found to be at levels that warrant consumption advisories for both these species at Highway 140 (MOE 2003b).

Trophic Level 2 - Benthivorous duck

Goldeneye Lyons creek wetland supports diving duck populations, both migratory and year round residents.

Trophic Level 2 - Planktivorous/Benthivorous fish

Bluegill Total PCB levels are found to be at levels that warrant consumption advisories for the Bluegill at Highway 140 (MOE 2003b).

Trophic Level 3 - Large piscivorous fish

Largemouth Bass Total PCB levels are found to be at levels that warrant consumption advisories for this species both at Highway 140 and downstream of the QEW (MOE 2003b).

Trophic Level 3 - Piscivorous mammal

Mink Mink are associated with numerous aquatic habitats. They are opportunistic feeders and are one of the most sensitive mammals to PCBs (Allan et al. 1991; CCME 2001).

Brown bullhead, Bluegill, and Largemouth bass (as well as other fish species) are collected regularly at Highway 140 and downstream of the QEW as part of the Sport Fish Contaminant Monitoring Program. Sport fish consumption restrictions for total PCBs for the general population begin at levels >0.153 µg/g (restriction to 4 meals per month); complete restriction is advised for levels >1.22 µg/g (MOE 2005).

A model of the feeding relationships linking these receptors with each other and benthic invertebrates and sediment is shown in Appendix A; Figure A1.

Assumptions for potential for biomagnification

For the prediction of PCB concentrations in the tissues of upper trophic level biota, bioaccumulation is considered to occur predominantly through dietary pathways. This is suggested by several experimental and modelling studies (Thomann 1980; Morrison et al. 1997; Madenjian et al. 1998; Russell et al. 1999). Dietary importance is also shown to be more important for PCB congeners with high octanol-water partition coefficients (K_{ow}) (> 6.3) (Morrison et al. 1997; Russell et al. 1999). Biomagnification factors used to derive the FCMs for the models, however, are based on total PCBs due to the lack of available congener specific data. Additionally, in modelling the exposure to and uptake of PCB by receptors, several conservative assumptions (i.e., maximum potential exposure to PCB) are made. These include:

For fish receptor:

- Fish consume invertebrates only from the site; and
- Fish feed on same invertebrate taxa as those collected in field sampling.

For wildlife receptor:

- 100% of the diet is fish;
- Fish are consumed only from the site in question;
- Fish consume invertebrates only from the site; and

- Fish feed on the same invertebrate taxa as those collected in field sampling.

In addition, the flux of PCBs between sediment, water and biota compartments are considered to be in equilibrium.

Presentation of model outcomes

Predicted concentrations of PCBs in each receptor species at each sampling site, calculated by multiplying observed total PCB concentrations in invertebrates (wet weight values from Appendix B; Table B1) by the appropriate FCM (from Table 3), are shown in Table 7 and Figure 8. Receptor PCB concentrations are presented for "minimum", "intermediate" and "maximum" levels of PCB exposure and uptake scenarios. In each subfigure, predicted [PCB] for the six receptors are presented in bar charts comparing reference and test sites. In the bar charts, which have the same logarithmic scales in all subfigures, two criteria concentrations are marked: (1) the 99th percentile of the predicted $[PCB]_{rec}$ for the reference sites, and (2) the IJC tissue objective for the protection of wildlife which consume fish. The tissue objective applies only to the fish receptors, and refers to the concentrations of PCB in the diets of wildlife that consume aquatic biota. The tissue objective for total PCBs is 0.1 µg/g ww (IJC 1989).

Exceedences of criteria

PCBs – minimum Under the minimum uptake and exposure scenario, site LC12 and site LC16 (just slightly) are above the IJC tissue objective for the bullhead and carp whereas only LC12 is slightly above the objective for the bluegill and bass (the "low" FCM estimates for bass and bluegill are lower than those for the carp and bullhead – see Table 3) (Figure 8a,b). Site LC03 is below the tissue objective and reference maximum as the minimum invertebrate tissue value used in the calculation is very low (0.003 µg/g ww for the odonate, Appendix B; Table B1). All reference sites are below the tissue objective. All test sites except LC03 and LC38 are above the predicted reference maximum for each receptor. (Most sites are just slightly above the reference maximum with the exception of LC12 and LC16.)

PCBs – intermediate Under the intermediate uptake and exposure scenario, all test sites exceed the tissue objective for all receptors; reference site exceedences are predicted at 0 sites for the

bluegill, 1 site for the bullhead (just above), and at all 4 sites for the carp and bass (Figure 8a,b). All test sites are above the predicted reference maximum for each receptor.

PCBs – maximum The maximum predictions of $[PCB]_{rec}$ result in all test sites exceeding the tissue objective and the reference sites maximum for all fish receptors (Figure 8a,b). Reference sites also exceed the tissue objective for all fish receptors.

Overall patterns

Beyond the comparisons of predicted $[PCB]_{rec}$ for exposed sites to reference sites and to the IJC tissue objective, patterns are evident in the differences in predicted $[PCB]_{rec}$ among the receptors, and among the three exposure and uptake scenarios.

Among receptors Under the minimum scenario, predicted $[PCB]_{rec}$ for all fish receptors are similar (predicted [PCB] for the diving duck are ~an order of magnitude higher) (Table 7). Under the intermediate and maximum scenarios, predicted $[PCB]_{rec}$ for the bullhead and the bluegill are similar, and are the lowest (both trophic level 2 receptors). The carp and the diving duck (also trophic level 2 receptors) also have similar predicted levels under the intermediate and maximum scenarios but are up to ~ 7× higher than the bullhead and bluegill predictions. Predicted $[PCB]_{rec}$ increases from trophic level 2 to 3, with the highest predictions noted for the bass. For fish receptors, there are differences of up to ~ 75× between bullhead and largemouth bass predictions. The number of sites at which $[PCB]_{rec}$ exceeds the tissue objective is the same (all sites) for the intermediate and maximum scenarios. The number of exposed sites at which predicted $[PCB]_{rec}$ exceeds the maximum of reference site concentrations is the same among receptors. This is because within a series (i.e., any of the minimum/ intermediate/ maximum groups), $[PCB]_{rec}$ all derive from the same $[PCB]_{inv}$ values. Differences among predicted $[PCB]_{rec}$ values reflect differences among uptake pathways in the BMFs from Table 3. The pattern of variability among sites is the same for all receptors within a scenario (i.e., the $[PCB]_{rec}$ values are fully correlated among receptors). Comparisons are not made to the mink since the FCMs are based on lipid normalized BMFs.

Among exposure and uptake scenarios Looking at differences between the minimum, intermediate and maximum exposure and effect scenarios for the *same* receptor, predicted $[PCB]_{rec}$ can range up to four orders of magnitude between the minimum and maximum scenarios (Table 7). The largest range is seen at site LC03 due to the very low minimum $[PCB]_{inv}$ value at this site. The range is especially large for the bass, a trophic 3 receptor that has the largest range in FCMs (Table 3). Under the minimum scenario, the predicted $[PCB]_{rec}$ for LC12 are above the IJC tissue objective for all fish receptors, and just above the tissue objective for LC16 for two fish receptors (bullhead and carp). Under both intermediate and maximum scenarios, all test sites have predicted $[PCB]_{rec}$ greater than the tissue objective for all fish receptors, and reference site predictions are below the objective only for the bullhead and bluegill in the intermediate scenario (Table 7).

3.3 Sediment Toxicity

Mean species survival, growth, and reproduction in Lyons Creek and reference sediments are shown in Table 8. The established numeric criteria for three categories (non-toxic, potentially toxic, toxic) are included for each laboratory species (Reynoldson and Day 1998).
Toxicity is evident at 3 sites: LC03, LC08 and LC12. At site LC03, there is acute toxicity to *Hyalella*, *Hexagenia* and *Chironomus*, and *Tubifex* reproductive impairment (low cocoon and young production). At site LC08, there is acute toxicity to *Hyalella*, *Hexagenia* and *Tubifex*. At site LC12, there is acute toxicity to *Hexagenia* and reduced *Chironomus* survival and growth. Reference site BLC02 (Black Creek), shows an effect on *Tubifex* reproduction, with low cocoon and young production evident (Table 8).

BEAST analysis: comparison to Great Lakes reference sites

The multivariate assessment (ordination) of sites was performed using the integrated survival, growth and reproduction toxicity test endpoints on three axes. Stress values for the ordinations, which indicate how effectively among-site similarities are represented by three axes compared to 10 variables, ranged from 0.08 to 0.09 (which is good). Ordination results for integrated endpoints are summarized in plots with two of the three axes in Appendix E (Figures E1 to E3). (Due to extreme toxicity evident at LC03 and LC08, these sites were assessed separately from the other sites.)

The most highly correlated endpoints include *Hyalella* survival ($r^2 \geq 0.89$), *Chironomus* survival ($r^2 \geq 0.91$) and *Tubifex* young production ($r^2 \geq 0.93$) for ordinations 1 and 2 (Figures E1 and E2), and *Hyalella* survival ($r^2 = 0.88$), *Tubifex* young production ($r^2 = 0.84$) and *Hexagenia* survival ($r^2 = 0.52$) for ordination 3 (Figure E3). The relationship between the habitat variables and toxicity is also shown in the ordinations. The highest correlation is seen for Zn in ordination 3 (Figure E3) ($r^2 = 0.51$), and remaining correlations have $r^2 \leq 0.16$. The departure of site LC12 is associated with decreased *Hexagenia* and *Chironomus* survival (shown as vectors in Figure E2). No habitat variable appears to be correlated with toxicity observed at site LC12. The departure of LC03 and LC08 from reference is most severe, and is likely due to a combination of decreased survival and growth endpoints (endpoints are located along the same vector line as the sites in the opposite direction) as well as reduced *Tubifex* cocoon and young production (Figure E3). These sites are oriented along a gradient of increasing Zn (Figure E3). Zinc is elevated at both sites (LC03: 7969 µg/g, LC08: 1080 µg/g) (Appendix C; Table C5).

Results of the BEAST toxicity assessment are summarized in Table 8. Most Lyons Creek sites (11 of 15) are non-toxic (Band 1), LC14 is potentially toxic (Band 2) and LC03, LC08 and LC12 are severely toxic (Band 4). The severely toxic sites (as well as the potentially toxic site) are located upstream of Highway 140. All reference sites are non-toxic with the exception of BLC02, which is potentially toxic.

Sediment toxicity and contaminant concentrations

Examination of relationships between sediment toxicity and sediment contaminants both graphically and by regression analysis aids in identifying possible causes of toxicity attributable to organic contaminants (as well as inorganic compounds, sediment nutrients and sediment grain size). The ordination of the multiple measurements of sediment toxicity by HMDS for the Lyons Creek and reference sites produced two descriptors of sediment toxicity (Figure 9). The resultant axes represent the original 10-dimensional among-site resemblances well (stress = 0.07). Principal axis correlation produces a vector for each toxicity endpoint along which the projections of sites in ordination space are maximally correlated. With the exception of *Hyalella* growth, all endpoints are significant at (r^2 range: 0.41 to 0.95, $P \leq 0.05$); *Hexagenia* survival being the most significant endpoint. The most significant environmental variables include total

PCBs, total PAHs and Zn (r^2 range: 0.73 to 0.84, $P \leq 0.001$). Most toxicity endpoints are positively correlated with both axes; therefore, the greater the toxicity of a site, the lower its score for Axis 1 and 2 generally. Site LC08 is distinctly separated from the other sites along Axis 1 and LC03 and LC12 are separated from the other sites on both axes and are oriented along a gradient of increasing PCBs, PAHs and Zn.

Integrated toxicity descriptors – contaminant relationships

Nine metals (As, Cd, Cr, Cu, Fe, Hg, Ni, Pb, and Zn) were ordinated by principal components analysis (PCA). The first three principal components account for 71%, 10% and 7% of the total variation, respectively. All measurement variables were negatively loaded for PC1, and loadings are of a similar magnitude. This component – denoted as "metPC1" – is used as a descriptor of general metal contamination. Sites elevated in metals score low for PC1. PCBs and PAHs were integrated by summing the concentrations of the individual congeners.

The integrated descriptors of sediment toxicity (Axis 1 and 2 scores from the HMDS) were plotted against the contaminant descriptors metPC1, total PCBs and total PAHs (the latter two of which were log-transformed to improve linearity) (Appendix E; Figure E4). The strongest relationship by multiple linear regression is for Axis 2, with 63% of the variation explained by PAHs and PCBs.

$$ToxAxis2 = 0.278 - 1.19\ logPAHs + 0.244\ logPCBs\ (p \leq 0.001)$$

Axis 1 ("ToxAxis1") is graphically related to total PAHs ("logPAHs"). This contaminant descriptor accounts for 56% of the variance in the Axis 1 toxicity descriptor.

$$ToxAxis1 = 0.123 - 1.28\ logPAHs\ (p \leq 0.001)$$

Individual toxicity descriptors - contaminant relationships

Relationships among individual measurement variables were evaluated by plotting sensitive endpoints (*Hexagenia* survival and growth, *Hyalella* survival and *Tubifex* young production) against concentrations of PCBs, PAHs and the integrated metal toxicity descriptor (metPC1) (Appendix E; Figure E5), as well as the individual concentrations of metals (As, Cd, Cr, Cu, Fe,

Hg, Ni, Pb, Zn), sediment nutrients (phosphorus, nitrogen and total organic carbon) and particle size (percents clay, sand, silt, and mean particle size) (Appendix E; Figures E6 and E7).

Plots of the four toxicity endpoints (listed above) against the three integrated contaminant descriptors show some significant relationships (Appendix E; Figure E5). Predictor coefficients that are negative indicate that decreased survival, growth or reproduction is related to increased contaminant concentrations.

For *Hyalella* survival, 25.0% of the variability is explained by total PCBs, and 30.9% is explained by PCB 105:

Hyalella survival = 1.09 - 0.0909 log total PCBs ($p = 0.012$)

Hyalella survival = 1.17 - 0.100 log PCB 105 ($p = 0.005$).

For *Hexagenia* survival, 30.0% of the variability is explained by total PCBs:

Hexagenia survival = 1.18 - 0.215 log total PCBs ($p = 0.006$)

For *Hexagenia* growth, 25.6% of the variability is explained by total PAHs and total PCBs and both predictors are significant ($P = 0.021, 0.017$):

Hexagenia growth = 0.641 - 0.912 log total PAHs + 0.246 log total PCBs ($p = 0.043$)

For *Tubifex* young production, 40.4% of the variability is explained by total PAHs:

Tubifex young production = 1.18 - 0.327 log total PAHs ($p = 0.002$).

Plots of the five toxicity endpoints against PAHs, PCBs, and individual metal concentrations, sediment nutrients and particle size also show some relationships that are slightly more significant and explain more of the variability than those above in some cases. Predictor coefficients that are negative indicate that decreased survival, growth or reproduction is related to increased contaminant concentrations, while positive coefficients indicate that decreased survival, growth or reproduction is related to a decreased contaminant or nutrient concentration.

For *Hyalella* survival: 55.6% of the variability is explained by Pb alone:

Hyalella survival = 2.26 - 0.706 log Pb ($p = \leq 0.001$)

For *Hexagenia* survival, 76.0% of the variability is explained by Pb, Cd and Fe. All predictors are significant ($p \leq 0.002$):

Hexagenia survival = 2.87 - 1.90 log Pb - 1.15 log Cd + 2.72 log Fe ($p = \leq 0.001$)

For *Hexagenia* growth, the greatest variability is explained by PCBs and PAHs as above.

Tubifex young production: 60.4% of the variability is explained by Pb, Cd and Zn. All predictors are significant ($p \leq 0.002$):

Tubifex young production = 1.69 - 1.34 log Pb - 1.10 log Cd + 0.543 log Zn ($p = 0.001$)

3.4 Community Structure

Benthic communities at reference and Lyons Creek sites consist predominantly of Chironomidae and Tubificidae, which are present at all sites. At Lyons Creek sites, tubificids range from 543 to 40,712/m^2 and are generally in lower numbers at downstream sites, and chironomids range from 3076 to 92,400/m^2 (Figure 10). At reference sites, tubificids range from 446 to 11,037/m^2, and chironomids from 1210 to 27,322/m^2. Other taxon groups present at the majority of test sites include hyalellid (0 – 2654/ m^2) and gammarid amphipods (0 – 1930/m^2), naidid worms (0 – 6031/m^2), ceratopogonid dipterans (0 – 4825/m^2), caenidae mayflies (0 – 6152/m^2), leptocerid caddisflies (0 – 23703/m^2), and coenagrionid odonates (0 – 1870/m^2) (Appendix F; Table F1). Lyons Creek sites have similar or slightly higher abundances of the dominant macroinvertebrate taxon than the reference creek sites, with some notable absences. Leptocerids are absent at five sites between the Welland pipe outfall and Highway 140 (sites LC03 to LC12), and caenids are absent at two of these sites (LC10 and LC12) (Figure 10). Site LC12, which has the second highest sediment [PCB] (7.4 µg/g) and which is acutely toxic to mayflies (see Table 8) is void of caddisflies, mayflies and amphipods (Figure 10). Taxon richness is generally similar for the reference sites and most test sites, ranging from 17 to 25 (mean 20) for the reference sites and from 11 to 28 for Lyons Creek sites (Figure 10). Site LC12 has the lowest number of taxa (11 taxa) followed by LC08 and LC10 (14 taxa) and LC16 (15 taxa). These sites are all upstream of Highway 140. Another notable difference between test and reference sites is the presence of zebra mussels (Dreissenidae) at site LC01 (2823/m^2). Dreissenids are mostly absent from all

other Lyons creek sites and are present at two reference sites in much lower abundance (36 – 121/m^2) (Appendix F; Table F1).

The HMDS (using invertebrate family data) reveals that three axes define the structure in the data (stress = 0.130, Figure 11). The degree of similarity among sites is indicated by the spatial proximity of sites in ordination space; sites in close proximity are similar in community structure. Families maximally correlated with the ordination axes scores are shown as vectors. Maximally correlated families include Tubificidae ($r^2 = 0.713$), Chironomidae ($r^2 = 0.656$) and Hyalellidae ($r^2 = 0.511$), which are shown as vectors in Figure 11. Higher abundances of Tubificidae and Chironomidae are associated with sites along Axes 1 and 3, respectively; generally sites from the Welland pipe (LC03) outfall to Highway 140 for tubificids and sites close to Highway 140 (e.g., LC17) for chironomids. Higher abundances of amphipods are associated with sites along Axes 2 and 3; generally these include sites downstream of LC16. Sites LC08, LC10 and LC12 are associated with decreased amphipod taxa (sites are oriented along the same vector in the opposite direction of amphipod vector). Black Creek reference site BLC02 is most different from the rest of the reference sites, separated from the other reference sites along the third axis and is oriented along a gradient of increasing NO_3/NO_2. Environmental variables such as Ca, Cu, Cd, are associated with sites along the first axis (sites upstream of Highway 140).

Ordination axes scores were used to compare Lyons Creek sites to reference creek (control) sites. The ANOVA F tests and Bonferroni's test show no significant differences between control and test sites. Pairwise comparisons using Tukey's test also reveal no significant differences between any sites. Site comparisons made using log(x)-transformed abundances of dominant taxon groups found in Lyons Creek (Tubificidae, Chironomidae, Hyalellidae, Gammaridae, Caenidae and Coenagrionidae) reveals a significant difference (ANOVA $p < 0.001$) in the abundance of coenagrionids (odonates). Bonferroni's simultaneous tests found a significantly decreased abundance of odonates at LC12 (no odonates present) ($p < 0.001$), and a significantly increased abundance of odonates at LC03 ($p = 0.047$). Dunnett's simultaneous tests reveals similar results for sites LC12 and LC03 and also found sites LC17 and LC19 to have a significantly greater abundance of odonates than controls ($p = 0.047$). Site comparisons made

using log(x)-transformed taxon richness reveals no significant difference between control and test sites.

3.5 Quality Assurance/Quality Control

Field replication

Three replicate sediment and overlying water samples were collected at LC29. Variability among site replicates in a measured analyte has three sources: natural within-site heterogeneity in the distribution of the analyte in sediment or water, differences in handling among samples, and laboratory measurement error. Among-triplicate variability indicates the overall "error" associated with quantifying conditions at a site based on a single sample. Variability, expressed as the Coefficient of Variation (CV), is shown in Appendix G; Table G1. Differences in variability are seen among the parameters. Overall, variability is low, with CVs ranging from 0.4 to 35.8% (median 2.3%); the highest CV is noted for PAHs.

Caduceon laboratory

Duplicate measurements of sediment metals and major oxides for two sites are shown in Appendix G; Table G2. Variability is low, with CVs ranging from 0.1 to 26% (mean 3%). Matrix spike recoveries and reference standard recoveries are shown in Table G3. Matrix spike recoveries are good, ranging from 89 to 109% (mean 99%). Three mercury reference standards were included in the analysis. Recoveries range from 93 to 111% (mean 101%).

MOE laboratory

Recoveries of matrix spikes, performed on sediment and biota samples are shown in Appendix G; Tables G4 and G5, respectively. Matrix spikes were performed with three PAH compounds: d10-phenanthrene, d12-chrysene and d8-naphthalene. Recoveries for sediment matrix spikes are highest for phenanthrene (range 72 to 140%, mean 97%), followed by chrysene (range 36 to 140%, mean 65%), and naphthalene (range 23 to 120%, mean 60%) (Tables G5 and G6). For biota samples, matrix spike recoveries are similar to that seen for sediment, with recoveries highest for phenanthrene (range 87 to 120%, mean 99%), followed by chrysene (range 52 to 110%, mean 72%), and naphthalene (range 41 to 94%, mean 67%) (Table G6).

Benthic community sorting efficiency

The mean percent community sorting efficiency for Lyons Creek samples, which represents the overall average for one sorter over four months, is 2.6%. This is an acceptable low level, indicating that a good representation of the benthic community was achieved.

4 DISCUSSION

4.1 PCB Concentrations at Lyons Creek Sites Relative to Reference Sites

4.1.1 Sediment

Concentrations of PCBs in the upper 10 cm layer of sediment at Lyons Creek sites (except the site farthest downstream) are greater than $[PCB]_{sed}$ at references sites and are highest between the Welland Canal and Highway 140. Reference creek concentrations (0.003 to 0.016 µg/g) compare to background concentrations of 0.005 to 0.019 µg/g reported for the upper Great Lakes and North Channel (Rowan and Rasmussen 1992). The CCME (1999b) freshwater sediment quality guideline (Probable Effect Level) for PCBs (0.277 µg/g) is exceeded at 13 of 15 Lyons Creek sites (LC03 to LC29). Overall, $[PCB]_{sed}$ declines with distance downstream of the former pipe outfall; $[PCB]_{sed}$ downstream of the QEW are similar to that upstream of the former outfall, and similar to reference creek concentrations.

The MOE collected sediment core samples (sectioned at 0, 25 and 50 cm) from five transects in Lyons Creek in 1991 (MOE 1993b). The maximum concentration reported in surficial sediment (0 cm) was 4.6 µg/g, which is lower than the maximum [PCB] reported in the current study (12.5 µg/g). ([PCB] was also found to increase with sediment depth to 25 cm, and then decreased at 50 cm.) Results are not directly comparable as the top 10 cm sediments were analyzed in the current study; however, the highest [PCB] are consistently in sediments upstream of Highway 140. The MOE reported [PCB] in the top 5 to 10 cm sediment ranging from < trace amount to 6.04 µg/g at five sites collected in Lyons Creek in 1992 (MOE 1998). These five sites were located southwest (upstream) of the Welland pipe outfall to just downstream of Highway 140 and are in close vicinity to sites in the current study. Sediment [PCB] at 3 of 5 sites in 1992 are similar to those in the current study, while at 2 sites, there was ~2 to 4-fold difference in sediment [PCB]. Again, results are not directly comparable due to differences in sampling depth.

4.1.2 Benthic invertebrates

PCB tissue concentration

Sediments are an important source of organic (hydrophobic) compounds such as PCBs to aquatic organisms; therefore, BSAFs are an indication of chemical bioavailability (Niimi 1996). The BSAFs reduce site variability due to differences in total organic carbon concentration and allow differences in PCB bioaccumulation between species to be examined (Ankley et al. 1992). PCBs are taken up by the four invertebrate taxa assessed. Biota-sediment accumulation factors are >1 for all reference sites and are highest for Lyons Creek sites with the lowest sediment [PCB]. Site LC03, which has the highest sediment [PCB], has the lowest BSAFs; LC01 and LC38, which have the lowest sediment [PCB], have the highest BSAFs. Tissue concentrations do not increase as much as sediment concentrations at highly contaminated sites; therefore, high BSAFs at the reference sites and the low BSAFs at the highly contaminated sites are not unusual. For the Lyons Creek sites, BSAFs are overall highest for the amphipods. Niimi (1996) reports a BSAF for *D. hoyi* (amphipod) of 4, very close to the mean value of 3.5 for amphipods collected from Lyons Creek. Ankley et al. (1992) report a mean BSAF (± SD) of 0.87 (±0.38) for oligochaetes collected from the lower Fox River/Green Bay sediment, which is lower than the mean BSAF in the current study for oligochaetes collected from Lyons Creek (4.15 ± 10.16). The large mean BSAF is driven by site LC38, which has low [PCB] in the sediment (0.018 µg/g) and high sediment TOC (10.7%). Site LC38 is located approximately 12 km downstream of Highway 140. If this site were removed from the calculation, the mean BSAF would be 1.10 (±1.16). It should be noted that BSAFS are based on whole-body, uncleared-gut concentrations which could obscure true BSAFs. As the amount of sediment in the gut increases, the measured BSAF will converge to 1. A true BSAF < 1 will be overestimated because the concentration in the sediment is greater than the tissue concentration, whereas a true BSAF >1 will be underestimated because sediment concentrations are lower than that found in the tissue (Bechtel Jacobs 1998). Concentrations of PCBs in benthic invertebrates are elevated above the $[PCB]_{inv}$ for the reference sites at the majority of Lyons Creek sites for 3 of 4 taxa, and benthos collected from LC12 and LC17 are consistently highest in [PCB]. For the odonates, the [PCB] are consistently the lowest. The odonates (samples contained a mixture of dragonflies and damselflies) are predacious invertebrates and will feed on invertebrates as well as small vertebrates such as tadpoles and fish

fry. They likely have less direct contact with sediment than the other taxa analyzed which may explain the lower PCB levels.

The MOE collected oligochaete worm tissue at three transects (T1, T3, and T5) in Lyons Creek in 1992 (MOE 1993b; 1998). In the current study, oligochaete data are available in vicinity of these transects for comparison: LC01 (≈T1), LC08 (≈T3) and LC12 (≈T5). Total PCBs (converted to dry weight) at transects T1 (at pumping station), T3, and T5 are 1.0, 4.3 and 5.7 μg/g, respectively, and increase with distance downstream (~750 m) of the pumping station. In the current study, $[PCB]_{olig}$ at sites LC01 and LC08 are similar (0.34 and 0.31 μg/g, respectively) and are ~an order of magnitude lower than those seen in 1992. $[PCB]_{olig}$ at LC12 (closest to T5) is ~an order of magnitude higher (53 μg/g) than that seen in 1992.

Sediment toxicity

Sediment toxicity tests reveal that the mayfly, *Hexagenia* spp. is most sensitive to Lyons Creek sediments, showing an acutely toxic response at 3 sites (LC03, LC08, LC12), followed by the amphipod *Hyalella*, showing an acutely toxic response at 2 sites (LC03, LC08). The greatest toxicity is observed at site LC03, approximately 4m downstream of the former Welland Pipe outfall, where acute and/or chronic toxicity are evident to all four laboratory organisms. The severely toxic sites have the highest sediment [PCB] (4.7 to 12.5 μg/g); LC03 has the highest $[PCB]_{sed}$, and SELs are exceeded for As, Cu, Ni, and Zn as well. Toxicity to *Tubifex* is observed at two sites (LC03 and LC08), but the modes of toxicity differ. At LC03, the effect is chronic, with low number of cocoons produced per adult, indicating an effect primarily on gametogenesis (cocoon production), and the low number of young (but high hatching rate) suggests a toxic effect on the small individuals. At LC08, the effect on *Tubifex* is primarily acute (35% survival), resulting in a low reproductive output. The use of several species and different physiological endpoints is important in toxicity evaluation as sensitivities will differ among species and sensitivities also tend to be contaminant specific. Toxicity is observed to ~ 750 m downstream (LC12) of the pumping station at the Welland Canal, and no toxicity is observed from ~1450 m downstream (LC14) on. The three severely toxic sites had an oily residue present on the surface water and the sediment had a distinct strong odour of hydrocarbons that was not observed at the

other sites. Elevated zinc is correlated with the location of sites in ordination space, specifically site LC03, where Zn was quite high (7969 µg/g).

Better than 50%, and up to 76%, of the variability in toxicity of Lyons Creek sediments is explained by most regression models. *Hexagenia* survival and individual metal contaminants produce the strongest relationship followed by the toxicity descriptor Axis 2 and integrated contaminants (PAHs and PCBs). Predictors with coefficients indicating decrease in toxicity with increase in contaminant concentration do not suggest causal relationships. These include positive coefficients for the survival, growth and reproduction variables. (A decrease in values for toxAxis1 and toxAxis2 is associated with increasing toxicity generally.) After excluding predictors not indicative of toxicity relationships, toxicity to *Hexagenia* appears to be most strongly associated with Pb and Cd; however, concentrations of these two metals are not high in the sediments (below the LEL or SEL – see Appendix C; Table C5). PAHs are also indicated as potentially toxic in the regressions for toxAxis2. Contaminant mixtures can exhibit various interactive and confounded effects that are complex and difficult to recognize using a correlation/regression approach with a sample size not much larger than the number of contaminants. Further data and experimental evidence would be needed to test whether the contaminants showing the strongest relationships in these analyses are in fact responsible for the sediment toxicity.

The MOE performed sediment toxicity tests (top 5 to 10 cm sediment) with *Hexagenia* spp. and the midge *Chironomus tentans* at five Lyons Creek sites in 1992 (T1 (upstream control), T3, T5, Stn 4, Stn 5) and repeated 2 of the sites in 1996 (T3, T5) (MOE 1993b; 1998). These MOE sites are in closest vicinity to sites LC01, LC06, LC12, LC16, and LC17, respectively, in the current study. In 1992, acute toxicity was observed for *Hexagenia* and *Chironomus* at two sites, T5 (≈LC12) and Stn 4 (≈LC16), with percent survival ranging from 55 to 60% at T5, and from 33 to 60% at Stn 4. Results for T5 are similar to that seen at LC12 in the current study, where percent survival for *Hexagenia* and *Chironomus* range from 46 to 64%. *Hexagenia* and *Chironomus* survival at site LC16 range from 93 to 96%, much higher than that seen at Stn 4 in 1992. An oily sheen and/or a strong oily odour were also noted in the 1992 sampling at all sites except T1. Reduced mayfly and chironomid growth compared to the upstream control site T1 (≈LC01) was

also observed at all sites in 1992. In the current study, lower growth (compared to LC01) is observed at LC12 for the mayfly and midge and at LC16 for the midge only. Site T5, which was acutely toxicity in 1992, showed no evidence of acute toxicity in 1996. The MOE ranked sites according to sediment [PCB] and level of biological effect and attributed differences in results to in-situ heterogeneity or contaminant redistribution over time as there was a 4-fold increase in [PCB] at T3 in 1996 and a 4-fold decrease in [PCB] at T5 in 1996 (MOE 1998).

Community structure

Lyons Creek benthic communities were not compared to the Great Lakes reference communities (BEAST model) because this method can only be applied with confidence to test sites within the range of habitats and geographic areas contained within the reference data set (Reynoldson and Day 1998). The Great Lakes reference database consists of sites restricted to harbours, embayments and nearshore waters of the Great Lakes; there are no sites in connecting channels or small streams/creeks. Therefore, Lyons creek communities were strictly compared to neighbouring reference creek communities. Reference creeks used in the assessment were deemed appropriate for comparison to Lyons Creek based on five parameters: watershed area, stream order, wetland percentage, flow type and sediment type (NPCA 2003).

Overall, abundance and diversity of invertebrate families at Lyons Creek sites are similar or higher to that observed in neighbouring reference creeks, with the average number of organisms/m^2 at Lyons Creek sites ~2 times higher than that at the reference sites. However, site LC12 (severely toxic) has low taxa diversity (less than 2 standard deviations of the reference creek mean), is void of hyalellid and gammarid amphipods (one or both of which are present at all other Lyons Creek sites), caenid mayflies (present at most other sites), and leptocerid caddisflies, and there is a significantly lower abundance of coenagrionids (odonates) (present at all other sites including reference). Additionally, the highest PCB accumulation in benthos occurs at LC12, and this site has the second highest $[PCB]_{sed}$, (after LC03). Site LC08 (also severely toxic) has low taxa diversity (14 taxa) and is void of caddisflies, showing some concordance with toxicity as well. However, other sensitive taxa are present at LC08 such as mayflies and amphipods. Concordance between community impairment and toxicity at site LC03, however, is not strong. While LC03 (severely toxic) is void of caddisflies, taxa diversity

is high (22 taxa), and sensitive taxa such as mayflies and amphipods are present. While contaminants are present and sediments are toxic, it is possible that benthic communities have adapted or developed resistance.

4.2 Effects of PCBs in Sediment on PCBs in Invertebrates

Concentrations of PCBs in amphipods and chironomids are significantly influenced by sediment [PCB] (Table 6, Figure 7). The log-log relationship for $[PCB]_{sed}$ and $[PCB]_{inv}$ across sites is strongest for the amphipods. The amphipods accumulated more PCBs than the other three taxa at 64% of Lyons Creek sites; therefore, it is not surprising that the $[PCB]_{sed}$ - $[PCB]_{inv}$ relationship is strongest for the amphipod. With the addition of pH (positively correlated to total PCB concentration), the amount of variance explained increases by ~4% and ~12% for the amphipods and chironomids, respectively, and $[PCB]_{sed}$ is the most significant predictor. With the addition of pH, total P in the water and %sand, the oligochaete model becomes significant, and the amount of variance explained increases greatly (~60%). There is no significant relationship between $[PCB]_{inv}$ – $[PCB]_{sed}$ for the odonates.

Because concentrations of PCB in the benthic invertebrates were measured without clearing their guts, a fraction of the observed $[PCB]_{inv}$ could include sediment-bound PCB in the gut. This is relevant for assessing uptake of PCBs by predators of invertebrates, which consume whole organisms, but likely contributes to the strength of the $[PCB]_{sed}$ - $[PCB]_{inv}$ relationship.
For the amphipod and chironomid models, the fact that the model that best predicts $[PCB]_{inv}$ includes $[PCB]_{sed}$ as the most significant term, and the magnitude and direction of the regression coefficient is stable across both models suggests a real relationships between $[PCB]_{inv}$ and $[PCB]_{sed}$. Results from this assessment indicate that [PCB] for the amphipods and chironomids is largely determined by $[PCB]_{sed}$. Observing positive relationships between sediment and invertebrate PCB concentrations is evidence that PCB transfers from sediment into the food web.

4.3 Predicted PCB Concentrations in Receptor Species

4.3.1 Integration of prediction outcomes

Models involving a range of biomagnification conditions were used to predict [PCB] in receptors of concern for Lyons Creek. The six receptor species are considered important to the study area and encompass the trophic levels linking sediments to the top predators, where biomagnification

is expected to be greatest. Three levels of dietary exposure and trophic transfer of PCB were assumed: minimum and maximum scenarios to bracket the range of potential outcomes, and an intermediate scenario to characterize "average" conditions. The critical outcome of the evaluation is whether or not the predicted $[PCB]_{rec}$ values for exposed sites exceed the appropriate tissue guideline (IJC objective) and exceed the reference site maximum $[PCB]_{rec}$. For the minimum scenario, 2 of the 11 Lyons Creek sites exceed the IJC tissue objective and maximum reference concentration, and for the intermediate and maximum scenarios, all 11 sites, where tissue was collected, exceed the criteria.

Comparisons of the predicted fish receptor [PCB] with actual [PCB] in fishes collected from Lyons Creek are a means of qualitatively ground-truthing the prediction model. Measured [PCB] in fish receptors (sampled at the same time as the benthos by the MOE) are indications of actual bioaccumulation of PCBs, which is thought to occur primarily through dietary sources at the higher trophic levels. Brown bullhead, Carp, White sucker, Bluegill/Pumpkinseed and Largemouth bass (as well as other fish species not mentioned here) were collected by the MOE just upstream of Highway 140 (≡ LC16) and downstream of the QEW (near site LC38) in 2002 and 2003. Mean [PCB] in sport fish fillets range from 0.140 to 1.164 µg/g ww at Highway 140 and from 0.020 to 0.076 µg/g ww downstream of the QEW (MOE 2003b). Mean [PCB] in fish collected at Highway 140 are all above the IJC objective of 0.1 µg/g with the highest total [PCB] observed for the carp, followed by the White sucker. In some cases, PCBs in carp and white sucker (collected in 2003) are > levels that warrant total restriction on fish consumption. Restrictions on fish consumption for total PCBs begin at levels of 0.153 µg/g with total restriction on consumption for levels > 1.22 µg/g (MOE 2005). There are no consumption restrictions for sport fish downstream of the QEW (fish sampled in 2002 only). Actual PCB levels in sport fish receptors fall between the predicted minimum and intermediate exposure and uptake scenarios.

The IJC tissue objective applies to concentrations of PCBs in fishes, and is for the protection of wildlife consumers of fishes. Data are available for direct evaluation of the predicted tissue PCB levels for mink, specifically effects on reproduction. Mink are found to be very sensitive to PCB contamination through diet, more so than rats, mice, ferrets, and birds (Aulerich and Ringer

1977; Bleavins et al. 1980; CCME 2001). Mink kits are especially susceptible up to weaning, as PCB accumulation through milk is found to be more significant than placental transfer. Wren et al. (1987) found similar levels of liver PCBs in 5 week old kits as adults fed a continuous PCB diet for 8 months. The studies examining effects of PCBs on reproduction in mink involved the feeding of contaminated fish (i.e., carp) in various percentages (dose-response), or the feeding of standard mink diets supplemented with specific PCB mixtures (i.e. Aroclors 1016, 1221, 1248, 1252). Bleavins et al. (1980) investigated the chronic toxicity to mink fed (continuously) diets supplemented with Aroclors 1016 and 1242 for ~8 months. Aroclor 1242 was found to be more toxic than Aroclor 1016, and complete reproductive failure occurred at 5 ppm of the diet. Heaton et al. (1995) found that at a concentration of 2.6 ppm in carp from Saginaw Bay, MI, fed continuously to mink for 85 days resulted in decreased litter size, few live kits at birth and no kits surviving past 24 hours. Aulerich and Ringer (1977) found that mink fed diets supplemented with Aroclor 1254 for 8 months at 2 ppm resulted in complete reproductive failure. Wren et al. (1987a, b) found that PCBs as low as 1 ppm caused reduction in growth and survival of mink kits when exposed to supplemented diets, and that liver PCB concentration between 2 – 3 ppm may adversely affect reproduction. Mason (1989), in his review on distribution of river otters (a similar but more specialized feeder than the mink) in Europe, found that 2 ppm PCBs in the tissues is the level above which otter populations were decreasing or endangered.

Actual PCB concentrations in wild mink are reported in some studies. Foley et al (1988) report mean PCB concentrations (1:1 Aroclor 1254:1260) measured in fat tissue in the range of 1.6 – 9.5 µg/g lipid. These mink were trapped in New York State from 1982 to 1984 and the highest concentrations were seen in mink trapped in surrounding areas of Lake Ontario and the North and South Hudson River. These values reported in Foley et al. (1988) fall within both the minimum (range: 0.06 to 3.5µg/g, median 0.6µg/g) and intermediate (range: 3.0 to 118.6µg/g, median 24.1µg/g) scenarios (Table 7b). The maximum scenario (range: 15.4 to 691.7µg/g, median 134.4µg/g) overestimates actual values. Harding et al. (1999) report hepatic concentrations ranging from < 0.01 to 0.46 µg/g ww in wild mink collected along the Fraser River in B.C., 1994 -1996 (mean 0.07 and 0.08 for lower and upper Fraser River, respectively). A maximum percent lipid of 4.2 was reported for mink liver in this study. Using this lipid value, the adjusted range in PCB concentration is <0.2 to 10.9 µg/g lipid, which falls within the

minimum and lower end of the intermediate scenario for Lyons Creek. Haffner et al. (1998) collected mink from several townships in southern Ontario adjacent to Lake Ontario and Lake Erie in 1988 – 89. Total PCBs (Aroclor 1254:1260) ranged from 0.039 to 1.8 µg/g ww and % lipid values in the liver ranged from 2.2 to 7.7 %. The highest [PCB] were observed in the mink adjacent to western Lake Erie, with a total [PCB] of 24 µg/g lipid, higher than that observed in New York and B.C. studies. This value falls in the predicted intermediate scenario for Lyons Creek.

From the Wren et al. (1987a,b) study, the most conservative PCB concentration in mink liver that may cause reproductive impairment is 2µg/g. Using lipid values for mink liver provided from two studies (Harding et al. 1999, Haffner et al. 1998), a mean lipid value for mink liver of 3.4% was determined. The 2µg/g corresponds to 58.8 µg/g lipid or 1.8 on the log scale in Figure 8c. Under the minimum exposure and uptake scenario, this benchmark is not exceeded at any site. Under the intermediate exposure and uptake scenario, this benchmark is exceeded at site LC12 by ~2×. Under the maximum exposure and uptake scenario, this benchmark is exceeded at LC12 (by ~12×) as well as LC03, LC08, and LC14 to LC19 (Figure 8c, Table 7b). Therefore, under an average scenario, predicted mink receptor concentrations could be at levels associated with adverse effects at site LC12.

Studies examining the toxic effects of PCBs to ducks are less numerous than mink studies. Custer and Heinz (1980) found that levels as high as 55µg/g PCBs in mallard hen carcasses did not impair reproduction, although this level is above the Health and Welfare Canada (1991) guideline for PCBs in poultry (0.5 µg/g lipid). In Figure 8c, 55 ppm = 1.74 on the log scale. Under the intermediate scenario, site LC12 is slightly above this value, and under the maximum uptake and exposure scenario, LC12 is ~ 3× higher than the value. In comparison to other birds, Bush et al. (1974) found that Leghorn hen eggs containing 50µg/g Aroclor 1254 resulted in 50% mortality in chicks when exposed continuously for 1.6 weeks, and at 18.7 weeks, the concentration resulting in 50% mortality dropped to 9µg/g. High levels of PCBs in waterfowl have been reported by several authors. Total PCBs (1254:1260) in migratory diving ducks collected from Hamilton Harbour between 1981 and 1992 were in the range of 8.7 to 44 µg/g ww and 19 to 396 µg/g ww in breast muscle and liver tissue, respectively (Weseloh et al. 1995).

Total PCBs (as Aroclor equivalents) in Lesser Scaup wintering in Indiana Harbor Canal (south end of Lake Michigan) were in the range of 0.04 to 4.9 µg/g ww in carcass (excluding liver and gut contents) (Custer et al. 2000). Swift et al. (1993) report a maximum PCB concentration (combined Aroclors 1016, 1254 and 1260) in the breast muscle of Goldeneye wintering on the upper Niagara River, New York, of 0.3 µg/g ww. Kim et al. (1984) report a range in PCB concentrations in breast and fat muscle of waterfowl (Greater Scaup, Goldeneye and Bufflehead) collected in New York State of 0.05 to 2.2 µg/g and 0.24 to 53 µg/g ww, respectively. The range of values for breast muscle and carcass fall mostly between the predicted minimum and intermediate scenarios for Lyons Creek. Interspecific differences are observed in PCBs levels of the breast muscle and liver of three diving ducks (Bufflehead, Greater and Lesser Scaup) (Weseloh et al. (1995), and this is likely due to different diets, movements, and physiology (Custer and Heinz 1980). Assuming an average lipid value of 3% for breast muscle (from Weseloh et al. 1995), most predicted diving duck concentrations for Lyons Creek sites (and reference sites) are above the Health and Welfare guideline under all scenarios; however, it is not known whether these predicted levels would be associated with adverse effects. The Great Lakes Sport Fish consumption advisory 'do not eat' category of 1.9 µg/g ww (Anderson et al. 1993) is exceeded at 1 Lyons Creek site (LC12) under the minimum scenario and at 9/11 and all Lyons Creek sites under the intermediate and maximum scenarios, respectively. No reference site exceeds this consumption advisory under any scenario. The predicted values under the minimum and intermediate scenarios are within those observed for ducks in the Great Lakes region and therefore the model is not likely overestimating PCB levels.

4.3.2 Uncertainty in the prediction of PCB concentrations in receptors

The prediction of the potential transfer of PCBs from benthic invertebrates to the trophically linked receptor species involves several simplifying assumptions, each of which is associated with some degree of uncertainty in its relevance to conditions in Lyons Creek. While it is beyond the scope of this study to quantify these uncertainties, those considered most important are identified here.

Assumptions regarding the modelling of PCB biomagnification include those dealing with the exposure of the receptors to PCB, and those dealing with the effects of PCB on the receptors.

Regarding the latter category, some of the sources of uncertainty discussed by USEPA (1997c) could apply to the present study:

- Validity of the biomagnification model;
- Variability of the calculated FCMs;
- Selection of the receptors of concern;
- Trophic levels at which receptors feed;
- Limitations of the toxicity database (with respect to the determination of tissue guidelines);
- Effects of environmental cofactors and multiple stressors; and
- Total PCB vs. congener specific toxicity.

Among these sources, the greatest contributor to uncertainty in predicting the trophic transfer of PCBs could be the large ranges in the selected BMF and FCM values. For fish receptors, BMFs are derived from studies that report PCBs based on total Aroclor mixtures or on the sum of PCB congeners and the values are based on non lipid-corrected [PCB] in muscle tissue or whole fish samples (see Appendix A; Table A1). Niimi and Oliver (1989) found up to 5–fold higher total PCB congener concentrations in whole fish samples than in muscle samples taken from Lake Ontario salmonids. Koslowski et al. (1994) found variability among tissues of fish in the levels of PCBs (despite being lipid corrected), with consistently higher levels found in liver than the muscle by ~1 to 1.5×. The BMFs used in the current study range between 1 to 2 orders of magnitude between lowest and highest in some cases, and include all BMFs judged to be potentially applicable to Lyons Creek. Further validation of their relevance would require field studies beyond the scope of this assessment. Owing to limitations of the available data and the desire to minimize assumptions about the distributions of the data, a probabilistic approach was not applied to predict receptor PCB concentrations. Rather, low, medium and high BMFs were used to define the range of possible outcomes and intermediate values that "balance" the minimum and maximum rates of biomagnification. Another problem inherent in the literature-derived BMF data is the difficulty in assigning prey and predator species to discrete trophic levels due to omnivory. When omnivory is integrated with a continuous measurement of trophic position (e.g., using stable isotope methods), estimates of BMFs will generally be higher for each discrete trophic level (Vander Zanden and Rasmussen, 1996). Correct determination of trophic levels is also limited by how well the composition of a predator's diet is quantified. Often the

information necessary to clearly establish this is not available in the published studies. This is particularly important for the BMFs used for the Goldeneye predictions. Drobney et al. (1982) examined gut contents of 169 wintering ducks (Goldeneye, Lesser and Greater Scaup) and found on average their diets consisted of 35.6, 27.9, and 18.6% oligochaetes, and 64.8, 72.1 and 81.6% plants. Their guts contained a variety of invertebrates but on the most part were oligochaetes. Therefore while invertebrates are an important part of the winter diet of these ducks, plants make up larger portion of their diets. (The BMFs used to predict Goldeneye [PCB] in the current study is based on only one study performed in the lower Detroit River.)

It is also known that PCB congeners that act like 2,3,7,8 TCDD are most toxic (CCME 2001). In general, PCBs with a K_{ow} in the midrange tend to accumulate most readily in organisms and individual congeners can vary in their toxicity by up to a factor of 10,000 (Ahlborg et al. 1994). Also, it was found that fish that contain environmentally derived PCBs are more toxic to mink than the commercially derived Aroclor mixtures (Heaton et al. 1995). Tillitt et al. (1996) observed higher chlorinated PCBs in mink livers compared to their diets, and that mink liver BMFs increased with the number of chlorine atoms in general. Predicted receptor values in the current study are based on total PCBs and therefore do not reflect congener specific toxicity. However, PCB congeners were measured in the Lyons Creek benthic invertebrates and the TEQ concentrations determined for the coplanar PCBs.

Another potentially large source of uncertainty in predictions of $[PCB]_{rec}$ relates the exposure of receptors to PCB. These assumptions (listed in Section 3.2.4) are recognized as being conservative and limited in their representation of natural conditions. Spatial (and perhaps temporal) heterogeneity in the distribution of PCBs throughout the study area, and aspects of receptor ecology challenge the maximum exposure scenario. A particularly important source of uncertainty could be the assumption of 100% residency of all consumers in the food chain on each site. The degree to which this assumption is unrealistic is proportional to the size of the foraging areas of the receptor species relative to the area of contaminated sediment. Given that the sampling sites could be on the order of 10 × 10 m to 100 × 100 m (= 0.01 to 1.0 ha), the 100% residency assumption is likely unrealistic. According to data compiled in the Wildlife Exposure Factors Handbook (USEPA 1993), home range size for the Lesser Scaup is reported as

6.5 ha. Home range sizes of mink are reported as 7.8 to1626 ha, and 1.85 to 5.9 km of stream/river. If areas outside of the PCB-contaminated areas of Lyons Creek are not equally PCB-contaminated, the actual $[PCB]_{rec}$ would be lower than those predicted by the models.

4.4 Potential Risk of Adverse Effects of PCBs due to Biomagnification

Concluding that PCB originating from contaminated sediment could concentrate in the food web at levels that can cause adverse effects depends on establishing that:

- PCB in invertebrates from sites exposed in the past to industrial effluents is elevated relative to concentrations in invertebrates from reference sites;
- PCB in invertebrates is related to PCB in sediment; and
- Predicted levels of PCB in receptors at exposed sites that exceed levels in receptors at reference sites also exceed the IJC tissue objective.

The results of this study leads to one of two alternate conclusions: (a) PCBs are unlikely to concentrate in the food web at levels that can cause adverse effects, or (b) PCBs **could** concentrate in the food web at levels that can cause adverse effects. The determination of whether PCB biomagnification and adverse effects to higher trophic level organisms (fish or wildlife) are actually occurring in Lyons Creek is beyond the scope of this study, and would need to be addressed by a more comprehensive assessment such as a detailed risk assessment. The latter conclusion (b) is of **potential** biomagnification, but does not determine **actual** biomagnification. However, resident forage fish and sport fish collected from Lyons Creek by the MOE provides evidence of actual PCB bioaccumulation/biomagnification in higher trophic level organisms.

Results show that at most Lyons Creek sites, [PCB] in invertebrate taxa are significantly higher than concentrations for the reference sites (Figure 5). Measured total PCBs in 2 of the 4 taxa are very strongly related to total PCBs in sediment (Table 6, Figure 7). Regarding the trophic transfer modelling, based on outcomes for the fish receptors under the minimum and intermediate PCB exposure and uptake scenarios, from 2 to 11 Lyons Creek sites (where tissue was collected) could be considered "of concern" because of predicted $[PCB]_{rec}$ exceeding the IJC tissue objective as well as the maximum reference site concentration (Figure 8b). The highest

predicted $[PCB]_{rec}$ are at sites LC03 to LC17, and peaks at LC12. The likelihood of realizing this degree of PCB biomagnification is not clear due to uncertainties associated with predicting receptor [PCB] values and conservative assumptions of the assessment. Reducing uncertainty in the predictions of PCB biomagnification in Lyons Creek would be best achieved by identifying a more narrow range of appropriate BMFs, and by quantifying the actual exposures of receptors to dietary PCB.

Regarding the overall assessment of sediment conditions based on the integrated framework outlined in Section 1.2, the biomagnification line of evidence can differ from the other three lines of evidence. If fish and wildlife receptors are the concern, the appropriate spatial and temporal boundaries for assessing potential biomagnification are not the same as those for assessing sediment contaminant concentrations, sediment toxicity and benthic invertebrate communities. Activities of fishes, birds and mammals are not limited to individual sites to the same degree as contaminants and invertebrates. Whereas incorporating invertebrate contaminant bioaccumulation information into the framework works well on a site-by-site basis, fish and wildlife data require some form of spatial averaging or weighting to reflect realistic contaminant exposure conditions. On a per site basis, fish and wildlife biomagnification predictions remain "theoretical" or overly conservative.

One way of addressing the problem is to assess exposure to contaminants across areas of sediment comparable to the foraging areas of the receptors, as suggested by Freshman and Menzie (1996). Their "average concentration with area curve" exposure model involves determining the average concentration of a contaminant for increasing areas of soil, starting with the most contaminated site up to and beyond the foraging area of the receptor of interest. The average contaminant concentration for a section of soil corresponding to the foraging area is then compared to appropriate benchmark adverse effect levels. Exceedence of the benchmark by the average contaminant concentration is considered a potential impact to the receptor individual. The application of this method requires a grid-type or other statistically suitable array of sampling sites designed to representatively quantify contaminant conditions across the study area.

The application of tissue PCB residue data that are associated with adverse effects in other studies to evaluate potential risks to the receptors in the present study carries some uncertainty. The data come from different tissues, species and environmental conditions. The PCB reference concentration (RC) for avian (2.4 ng TEQ/kg ww) is derived from studies using white leghorn chicks. No uncertainty factor is incorporated (compared to the mammalian RC which incorporates an uncertainly factor of 10), as white leghorn hens may be particularly sensitive (10 to 1000× more sensitive) than wildlife (CCME 2001). Considering these uncertainties and the generally conservative ("worst case") assumption of the trophic transfer model, quantifying the probability that PCB from sediments in Lyons Creek could cause adverse effects to receptors is difficult.

4.5 Sediment Decision-Making Framework

The overall assessment of Lyons Creek sites is achieved by incorporating of the multiple lines of evidence in a sediment decision-making framework for contaminated sediments (Grapentine et al. 2002; Chapman and Anderson 2005). Table 9 depicts results of bulk sediment chemistry, benthic community, toxicity, and biomagnification components, shown in a separate column for each site. A decision is achieved by integration of all lines of evidence. A "●" denotes that adverse effects are likely, a "◐" denotes that adverse effects may or may not occur, and a "○" denotes that adverse effects are unlikely (Chapman and Anderson 2005).

Sediment PCBs

A "●" in the contaminant column indicates an elevation of contaminants above a sediment quality guideline. One or more exceedences of the SEL or PEL (0.277 µg/g) constitute a "●", one or more exceedences of the LEL or TEL constitute a "◐", and contaminant concentrations below the LEL or TEL constitute a "○". The SEL is not exceeded at any site (LC03 is very close); however, the PEL is exceeded at 13 of 15 sites (●). The remaining 2 sites (LC01 and LC38) are below the LEL (0.07 µg/g) (○). (For LC03, the SEL is exceeded for metals as well - As, Cu, Ni and Zn.)

Overall toxicity

A "●" in this column occurs when there is strong evidence of toxicity. Sites LC03, LC08 and LC12 fall into this category. These three sites have multiple endpoints exhibiting major toxicological effects, including survival, growth and reproduction effects and fall in Band 4 from the BEAST analysis. Potential toxicity is noted for LC14, which falls in Band 2 from the BEAST analysis (◐); there is slightly reduced *Chironomus* survival at this site and *Tubifex* reproductive outputs are lower than those observed at most sites. Remaining sites are non-toxic (○).

Benthos alteration

Differences in biological structure between reference creek sites (control) and Lyons Creek sites were determined using pattern analysis (ordination) and ANOVAs. Results indicate that LC12 has an impaired benthic community. Bonferrroni's simultaneous tests detect a significant difference ($p \leq 0.05$) between the control sites and LC12 with respect to abundance of odonates; LC12 is the only site where odonates are absent in the creek (amphipods, mayflies and caddisflies are also absent from LC12). The number of taxa present at site LC12 is also below 2 standard deviations of the reference mean.

Biomagnification potential

A "◐" in this column is determined by (a) a significant positive relationship between [PCB] in the sediment and [PCB] in the biota for the study area (three of the four taxa show significant relationships), (b) using the *minimum and intermediate* uptake and exposure scenarios (actual sport fish concentrations fall in between these two scenarios), predicted receptor PCB values are > IJC tissue objective and > the predicted maximum reference concentration. Under the minimum scenario, all fish receptors exceed the tissue objective and reference maximum at LC12, and 2 of the 4 fish receptors exceed the criteria at LC16. Under the intermediate scenario, all receptors at all Lyons Creek sites are above the IJC tissue objective and above the reference maximum However, actual PCB concentrations in fish collected from Lyons Creek are greater than those predicted under the minimum scenario and 4 to 10× lower that those predicted under the intermediate scenario.

The need to fully assess the risk of biomagnification is needed when there are no site-specific situations, such as sufficient evidence from fish advisories or previous research in the study area (Chapman and Anderson 2005). If there is sufficient evidence, significant biomagnification can be indicated (a "●" will replace "⊙" in the column), and management actions would be required. Currently, there are fish advisories in place at Highway 140 for several fish species (MOE 2005). Highway 140 is in the vicinity of LC16, and therefore this site likely requires management actions. Tissue was not collected at four sites (LC06, LC10, LC22, and LC23); therefore, these sites could not be assessed with respect to biomagnification potential.

5 CONCLUSIONS

Sediment and invertebrate PCBs

Sediment [PCB] are most significantly elevated in the upper reaches of the creek (upstream of Highway 140) and PCBs in sediment at the majority of Lyons Creek sites are elevated above those at reference sites. The highest $[PCB]_{sed}$ is found just downstream of the former Welland Pipe outfall (LC03) (~13 µg/g) and [PCB] decreases overall with distance downstream of the pipe. Sediment [PCB] at the site farthest downstream (downstream of the QEW) (LC38) is similar to that at the upstream site (LC01). The SEL is not exceeded at any site (LC03 is very close to the SEL), while 13 of 15 sites (from LC03 to LC29) exceed the PEL.

Invertebrate [PCB] are also most significantly elevated in the upper reaches of the creek; the highest $[PCB]_{inv}$ is found at LC12 (range 1 to 55 µg/g), which does not coincide with the highest $[PCB]_{sed}$. Total [PCB] are elevated above reference at most Lyons Creek sites for 3 of the 4 taxa. Overall, total $[PCB]_{inv}$ decreases with downstream of LC12; PCB levels farthest downstream are similar (slightly lower) to that upstream of the pipe outfall, but ~ 3 to 4× higher than [PCB] at reference sites for 2 of the 4 taxa. Total [PCB] are above the IJC PCB objective for all 4 taxa at LC12 and for at least 1 taxon at 8 other sites.

Sediment metals and nutrients

Some metals (primarily zinc) are elevated above the PSQG SEL criteria in the upper portion of the creek (upstream of Highway 140). Zinc, copper and nickel all exceed the PSQG SEL criteria at LC03; zinc exceeds the SEL by almost an order of magnitude. Metals in the sediments at

these concentrations may pose potential threat to the health of the resident benthic fauna at LC03. Elevated nitrogen is also observed along the creek.

Toxicity

There is severe toxicity at three sites: LC03, LC08 and LC12. Acute toxicity is evident at these sites for 1 to 3 laboratory organisms, and toxicity is most severe for LC03. Several metals and perhaps PAHs appear jointly related the pattern of toxicity among sampling sites but these metals (with the exception of zinc) are not unusually high in Lyons Creek. Further data and experimental evidence would be needed to test whether the contaminants showing the strongest relationships in these analyses are in fact responsible for the sediment toxicity. The strong hydrocarbon smell and oily residue present in the water observed at the toxic sites needs to be considered with respect to toxicity as well.

Benthos alteration

Generally, Lyons Creek benthic communities are similar to those at reference creeks except for LC12. There is a significantly lower abundance of Coenagrionidae (odonates) at LC12, low taxon diversity, and LC12 is void of key groups of odonates, mayflies, amphipods and caddisflies. Results for LC12 show concordance with toxicity results.

Biomagnification potential

From 2 to 11 sites (where tissue was collected) are predicted to have $[PCB]_{rec}$ higher than the IJC tissue objective and the maximum reference site $[PCB]_{rec}$. Thus, PCBs could bioaccumulate in receptors to levels that are not protective of adverse effects at 2 to 11 sites (under minimum and intermediate PCB-exposure and uptake scenarios). Sites LC12 and LC16 are most severe.

MOE PCB data at Highway 140 clearly show that PCBs accumulate in higher trophic organisms above the IJC guideline and above sport fish consumption advisories for several fish species. Comparison of predicted $[PCB]_{rec}$ to actual $[PCB]_{rec}$ reveals that the minimum uptake and exposure scenario underestimates $[PCB]_{rec}$ and the intermediate uptake and exposure scenario overestimates the $[PCB]_{rec}$.

Decision-making framework for sediment contamination

Using the rule-based, weight-of-evidence approach described in Chapman and Anderson (2005), management actions are required for LC12 due to elevated sediment PCBs, toxicity, benthos alteration and potential for biomagnification. Management actions are also likely required for sites in the vicinity of Highway 140 (due to elevated PCBs observed in fish collected in this area). The reasons for sediment toxicity need to be determined for LC03, LC08 and perhaps LC14. Under the intermediate scenario, the risk of biomagnification needs to be fully assessed at remaining sites where tissue was collected; however, under the minimum scenario this would not be required. (Actual concentrations fall between the minimum and intermediate uptake and exposure scenarios.)

The area from Ridge Road (LC03) to Highway 140 (LC16) is the most critical area of the creek. The highest sediment, invertebrate, (and fish) PCB concentrations occur in this area. Toxicity, altered benthic communities and potentially adverse effects due to biomagnification are also observed in this area.

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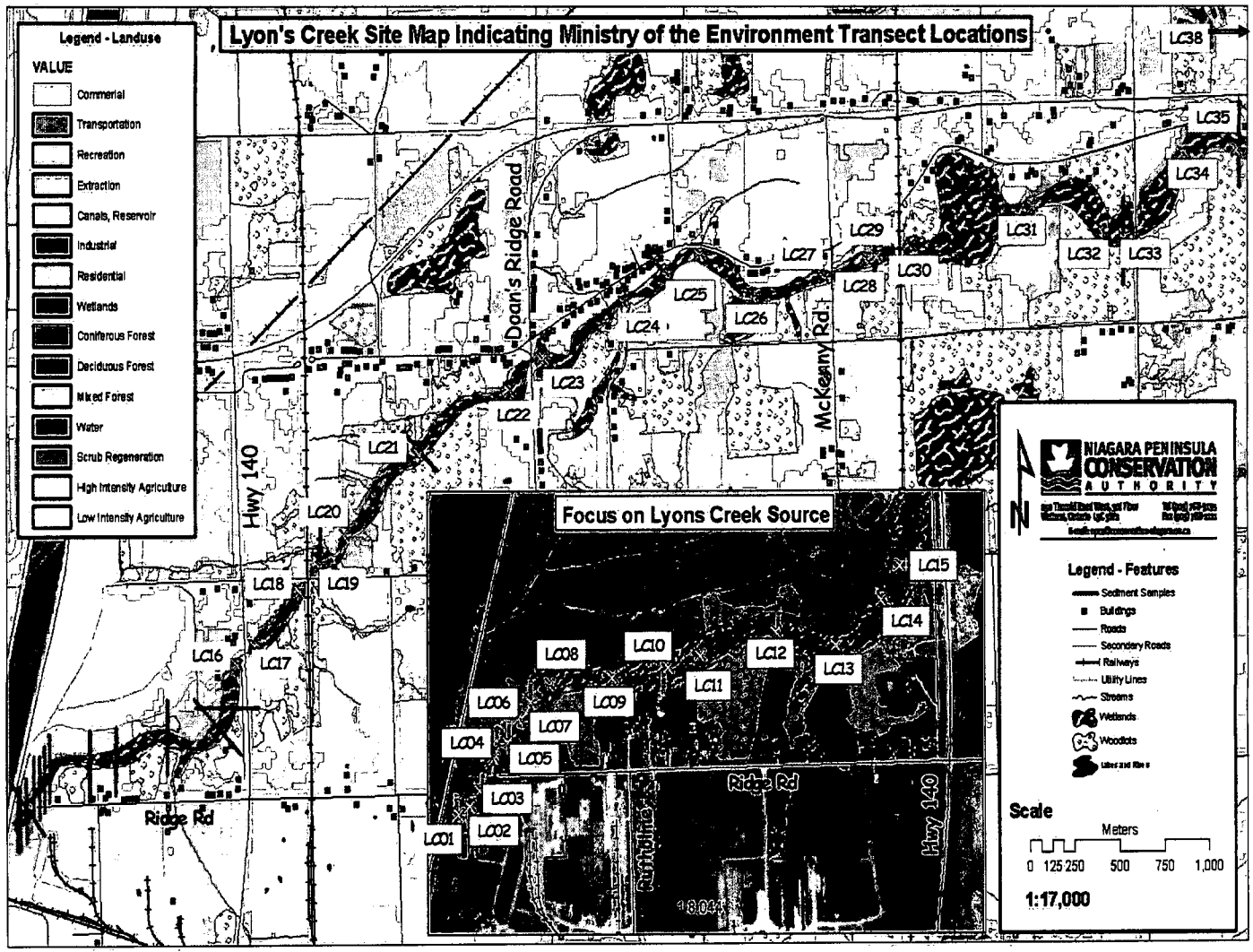


Figure 1. Environment Canada sampling locations (2002 and 2003) (highlighted yellow). Site LC38, indicated with an arrow, is located downstream of the QEW and approximately 3 km downstream of site LC35. (Other sites are MOE sampling locations from 2002 chemical screening survey.)

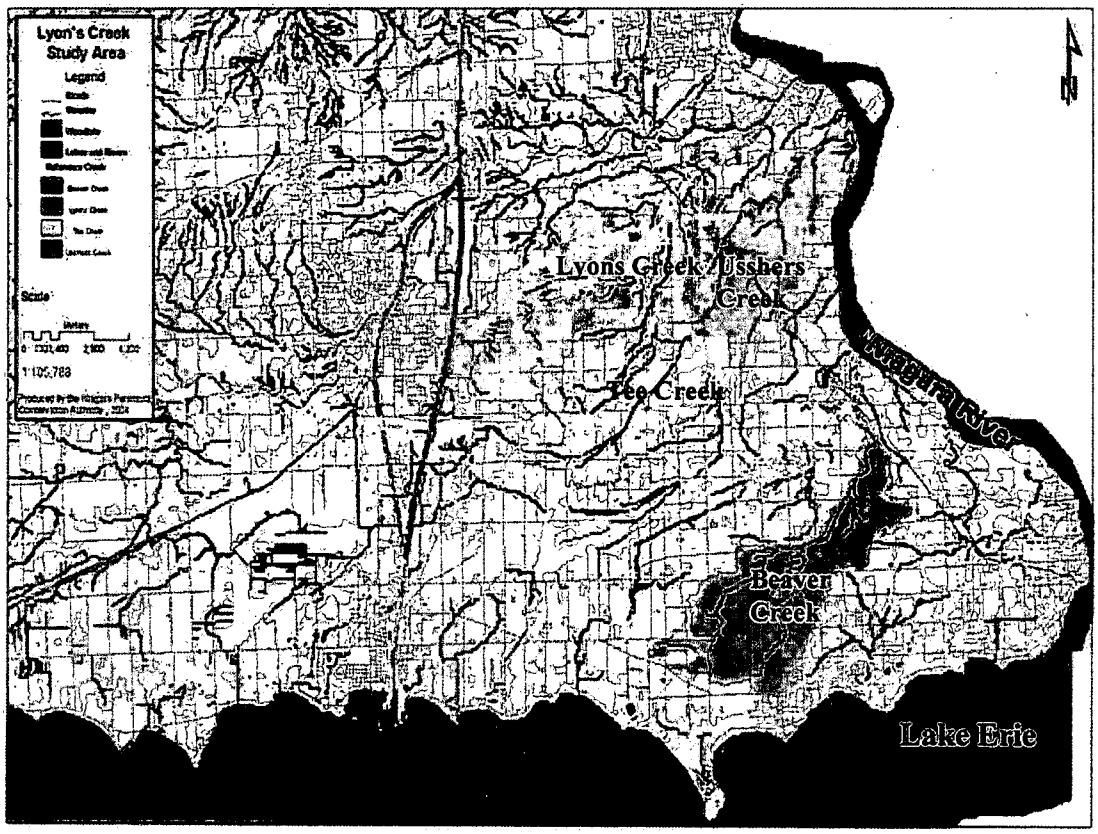


Figure 2. Location of reference creeks (Black Creek is not shown).

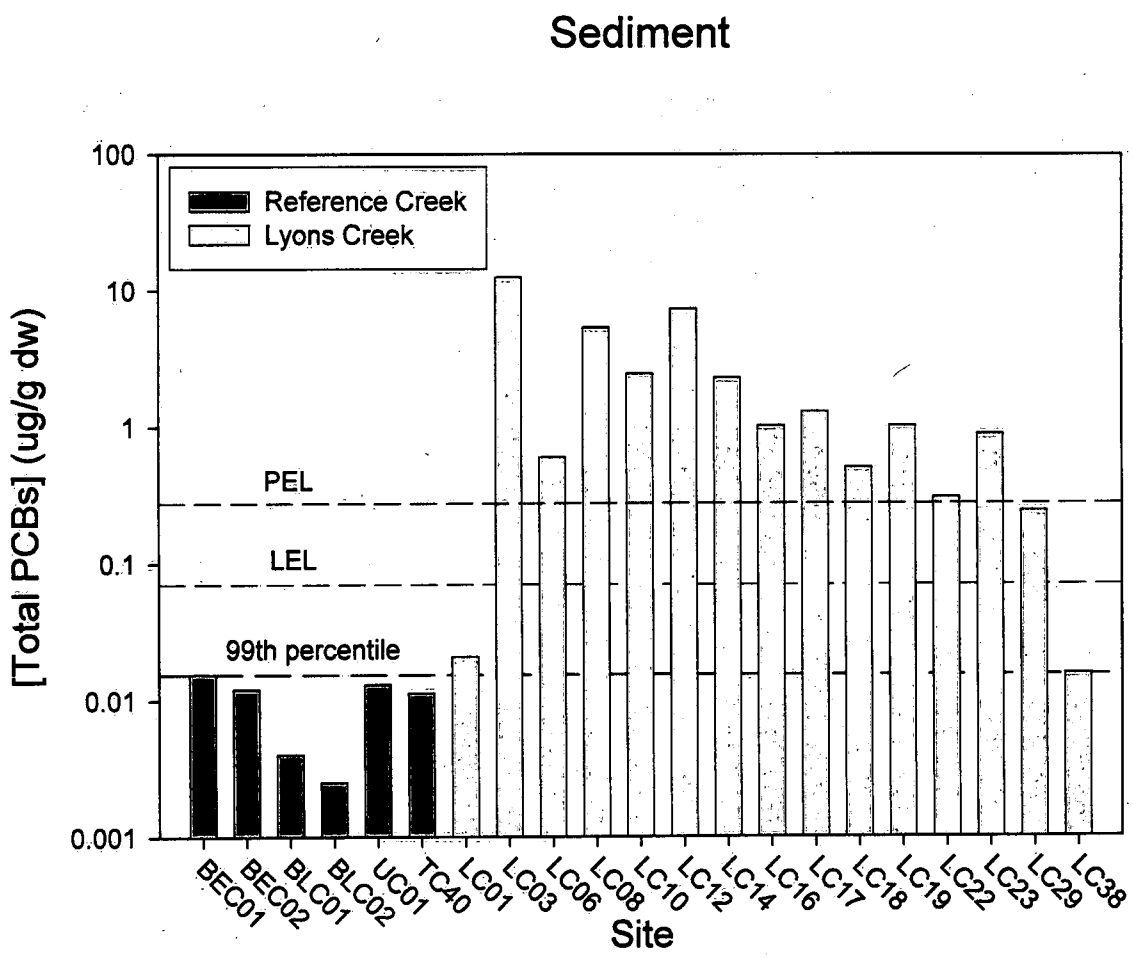


Figure 3. Total PCBs in Lyons Creek (grey) and reference creeks (green). The dotted line indicates the 99th percentile for reference sites. The lowest effect level (LEL) and probable effect level (PEL) for PCBs are indicated.

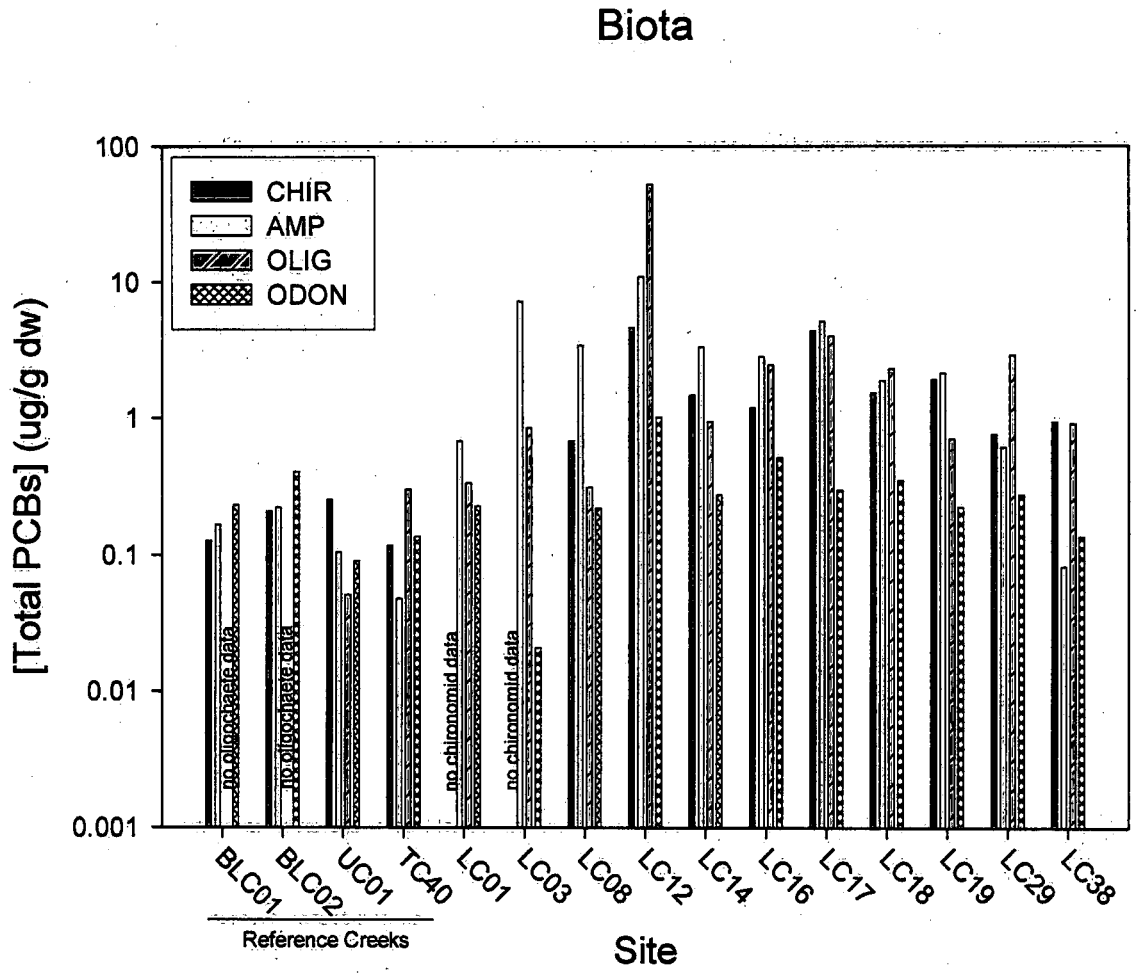


Figure 4. Total PCBs in benthic invertebrates (µg/g dry weight) collected from Lyons Creek (LC) and reference creeks.

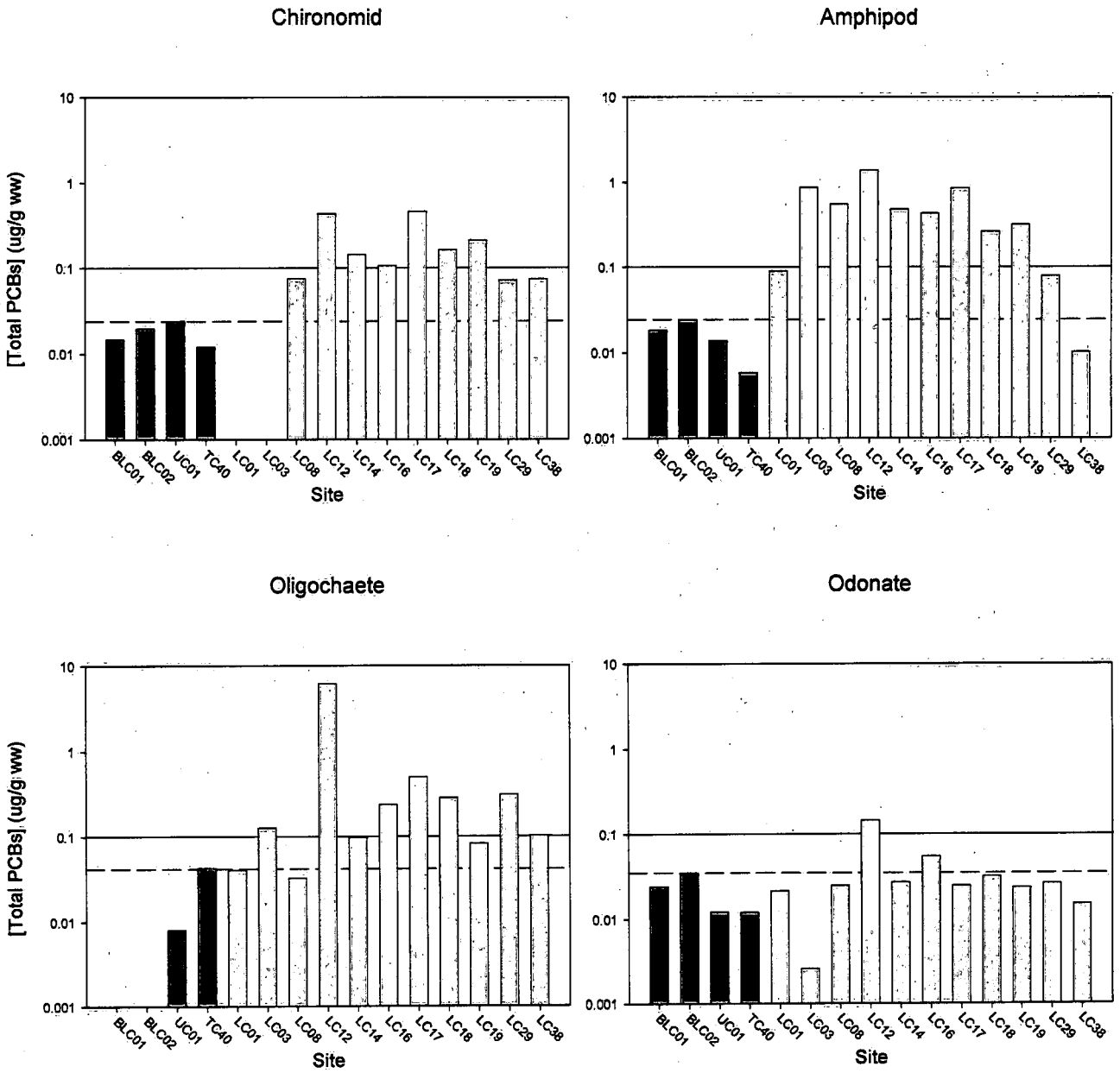


Figure 5. Total PCBs in biota (µg/g wet weight) collected from Lyons Creek (grey) and reference creeks (green). The dotted green line indicates the 99th percentile for the reference sites. The solid red line indicates the IJC guideline for the protection of wildlife consumers of aquatic species (0.1 µg/g ww).

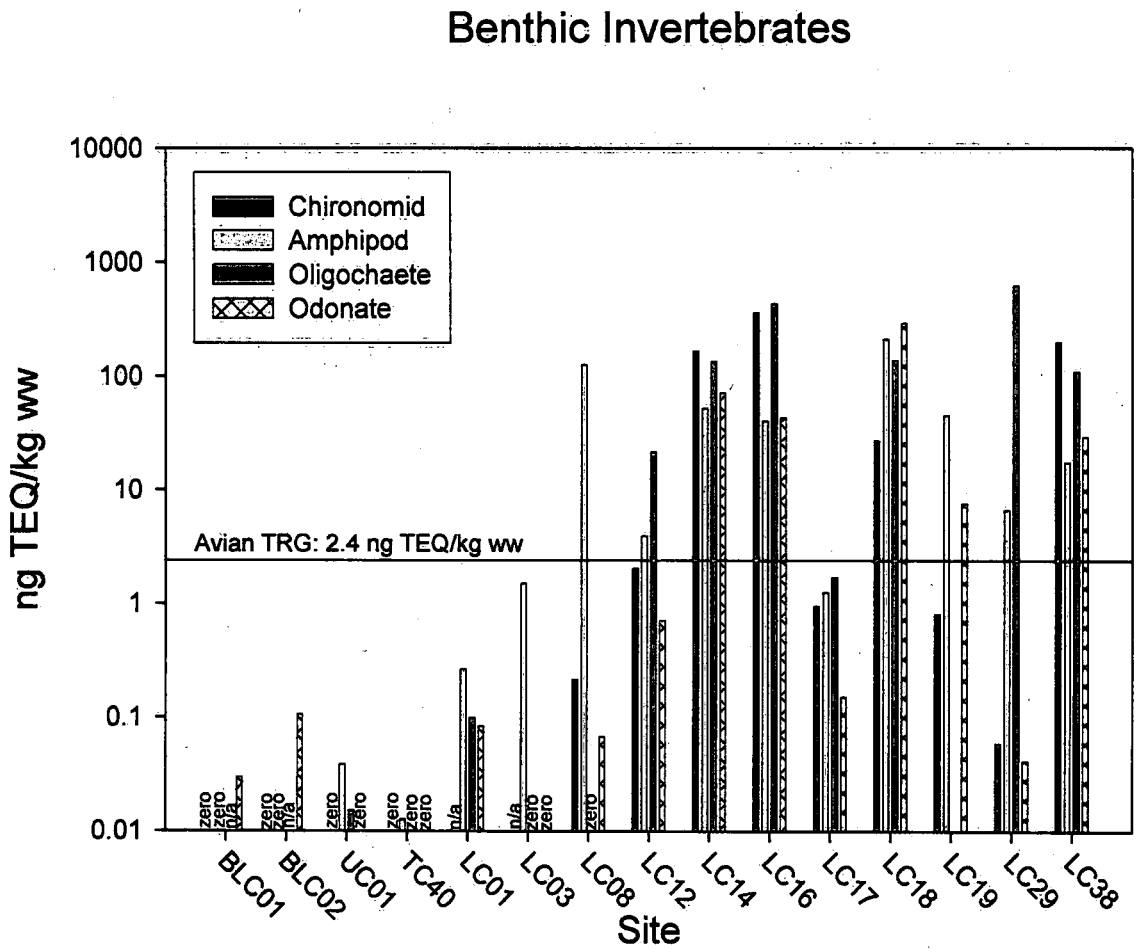


Figure 6. PCB concentrations expressed in toxic equivalent quantities for coplanar PCBs. The red lines indicate the Canadian tissue residue guideline (TRG) for the protection of avian consumers of aquatic biota (CCME 2001).

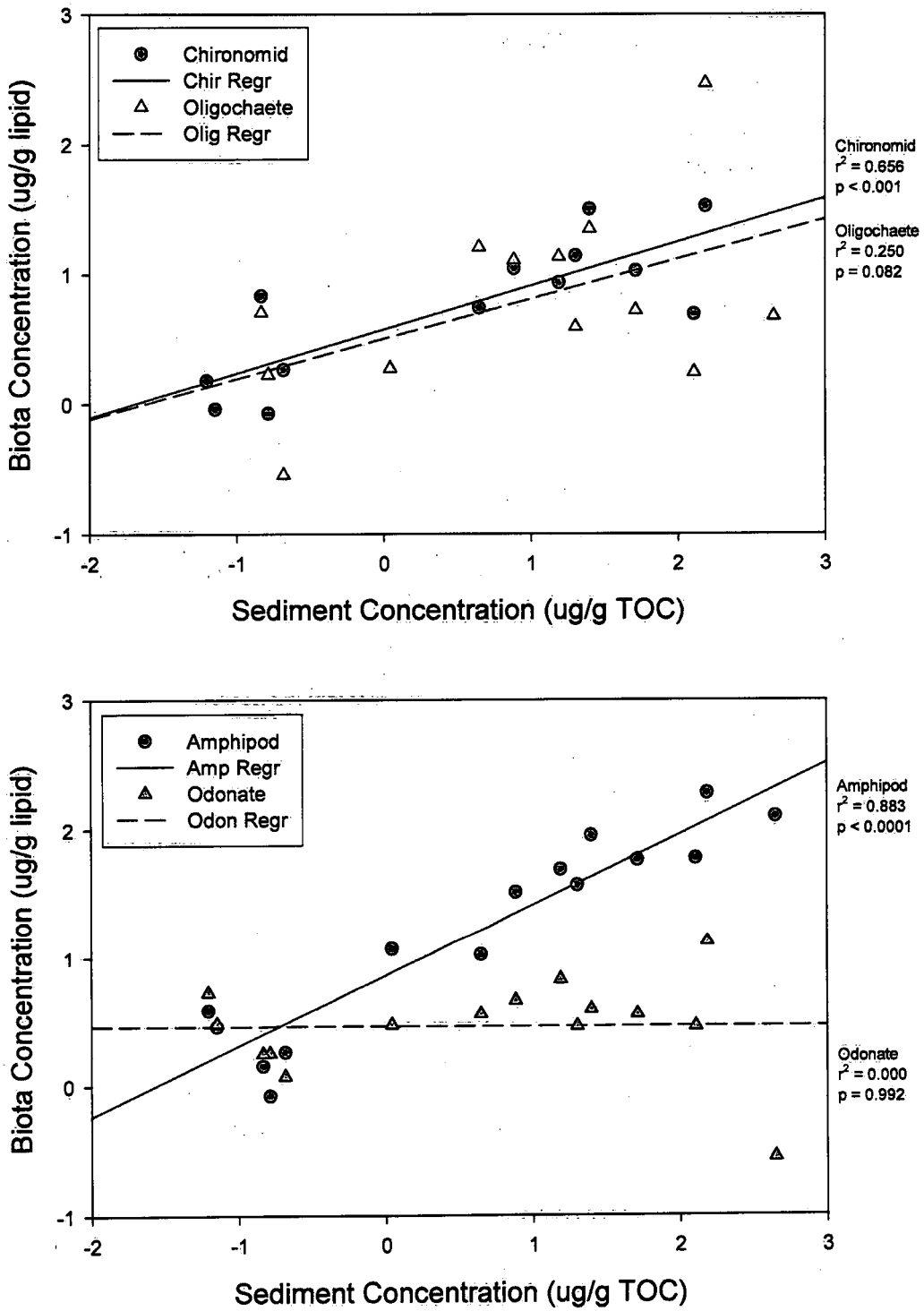


Figure 7. Relationships between total PCBs in biota (normalized to % lipid) and total PCBs in sediment (normalized to % total organic carbon). Separate regression lines are shown for each taxon.

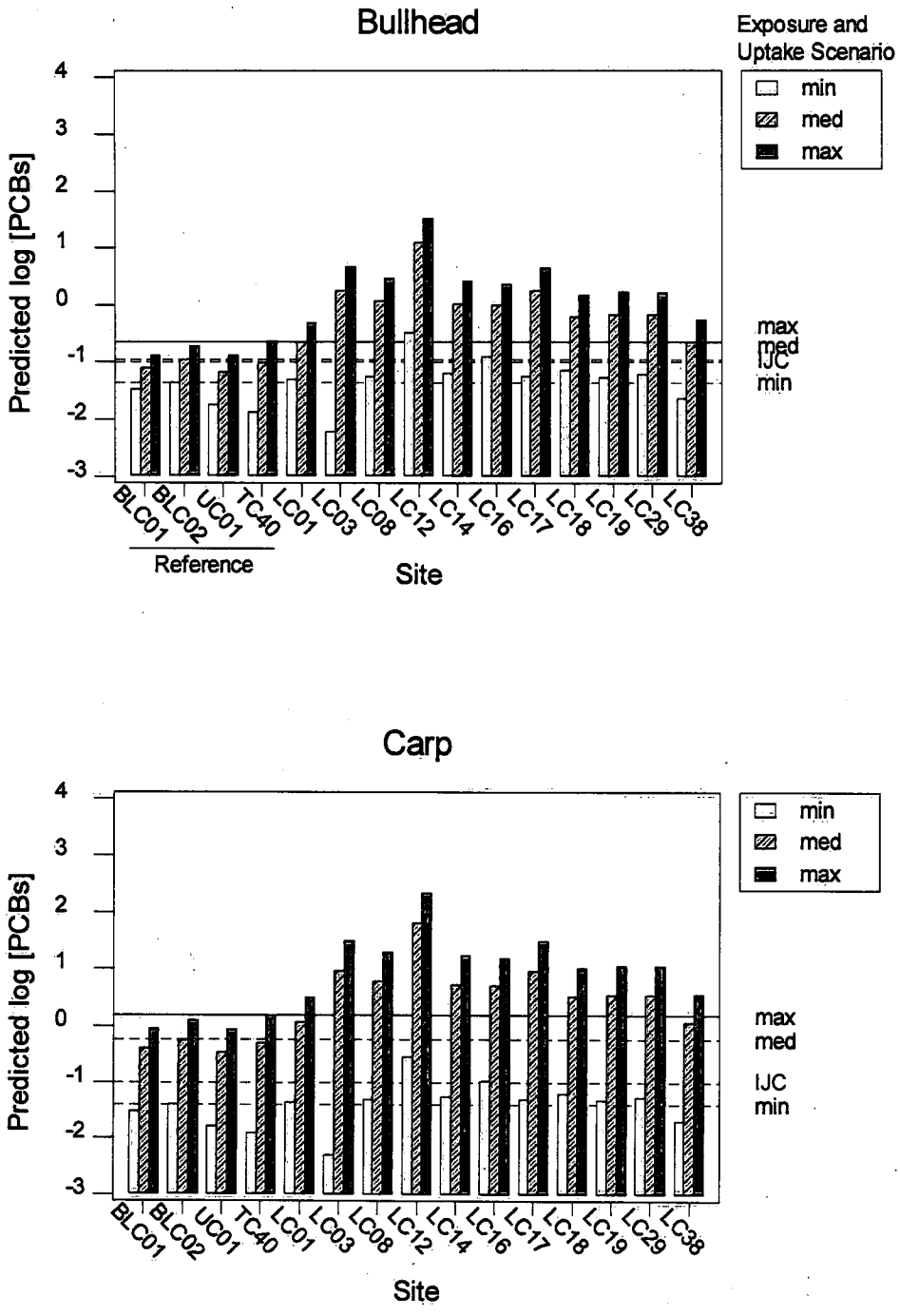


Figure 8a. Predictions (minimum, intermediate, and maximum) of total PCBs (μg/g ww) in benthivorous fish receptor species. Charts compare predicted [PCBs] among receptors and between reference and test sites. Highest predicted [PCBs] for reference sites for each scenario is indicated on the chart (min, med, max). The tissue objective (0.1 μg/g ww, IJC), where applicable, is indicated by the red dotted line.

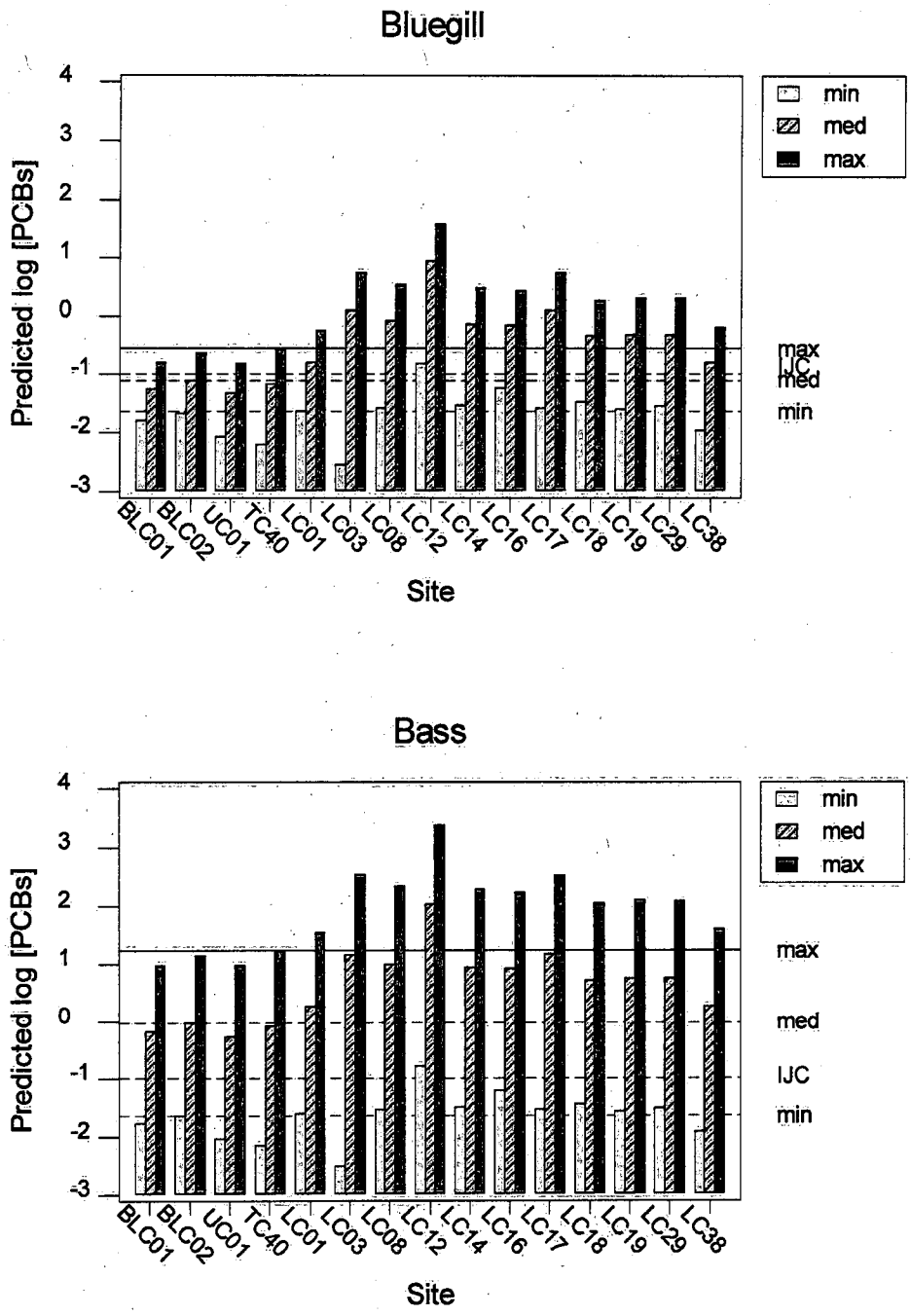
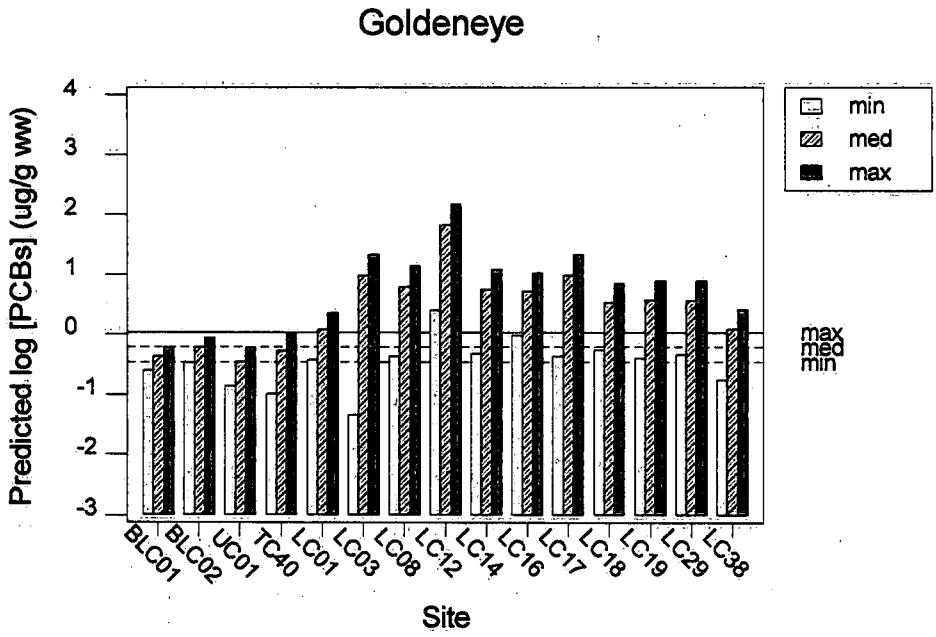


Figure 8b. Predictions (minimum, intermediate, and maximum) of total PCBs (µg/g ww) in fish receptor species. Charts compare predicted [PCBs] among receptors and between reference and test sites. Highest predicted [PCBs] for reference sites for each scenario is indicated on the chart (min, med, max). The tissue objective (0.1 µg/g ww, IJC), where applicable, is indicated by the red dotted line.



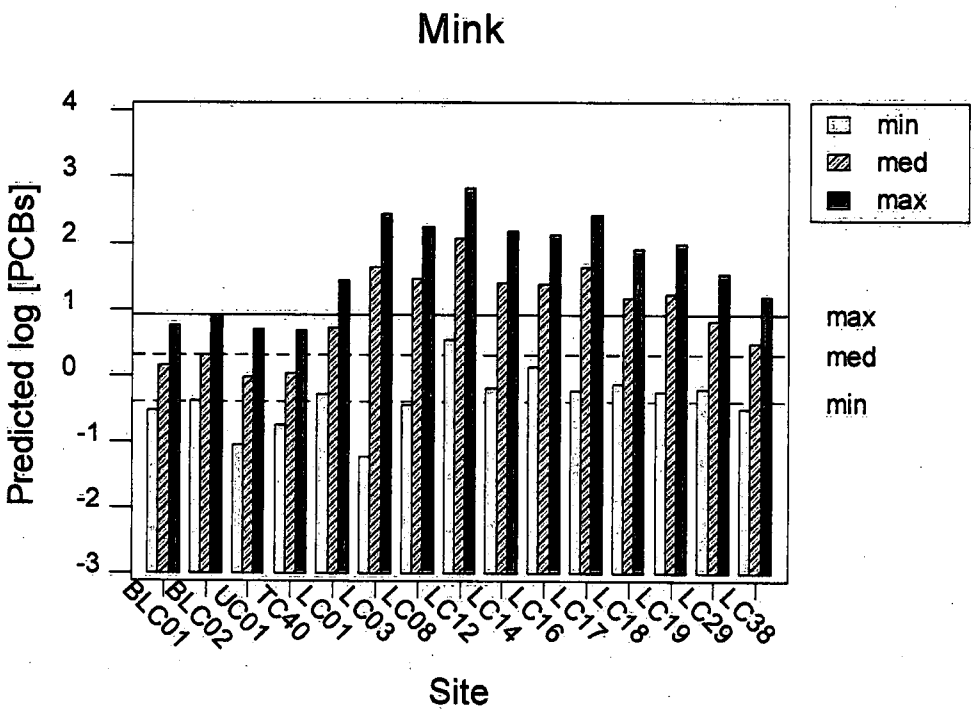


Figure 8c. Predictions (minimum, intermediate, and maximum) of total PCBs in waterfowl (μg/g) and mammal (μg/g lipid) receptor species. Charts compare predicted [PCBs] among receptors and between reference and test sites. Highest predicted [PCBs] for reference sites for each scenario is indicated on the chart (min, med, max).

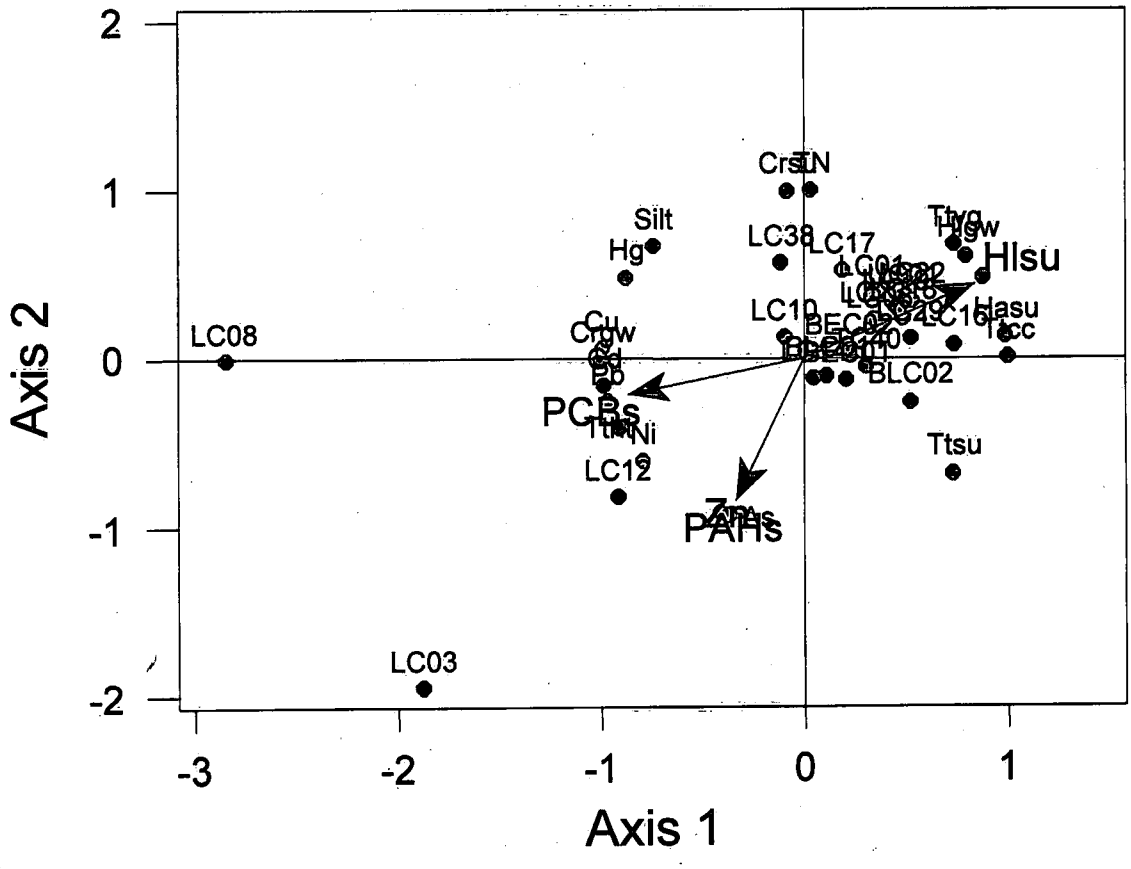
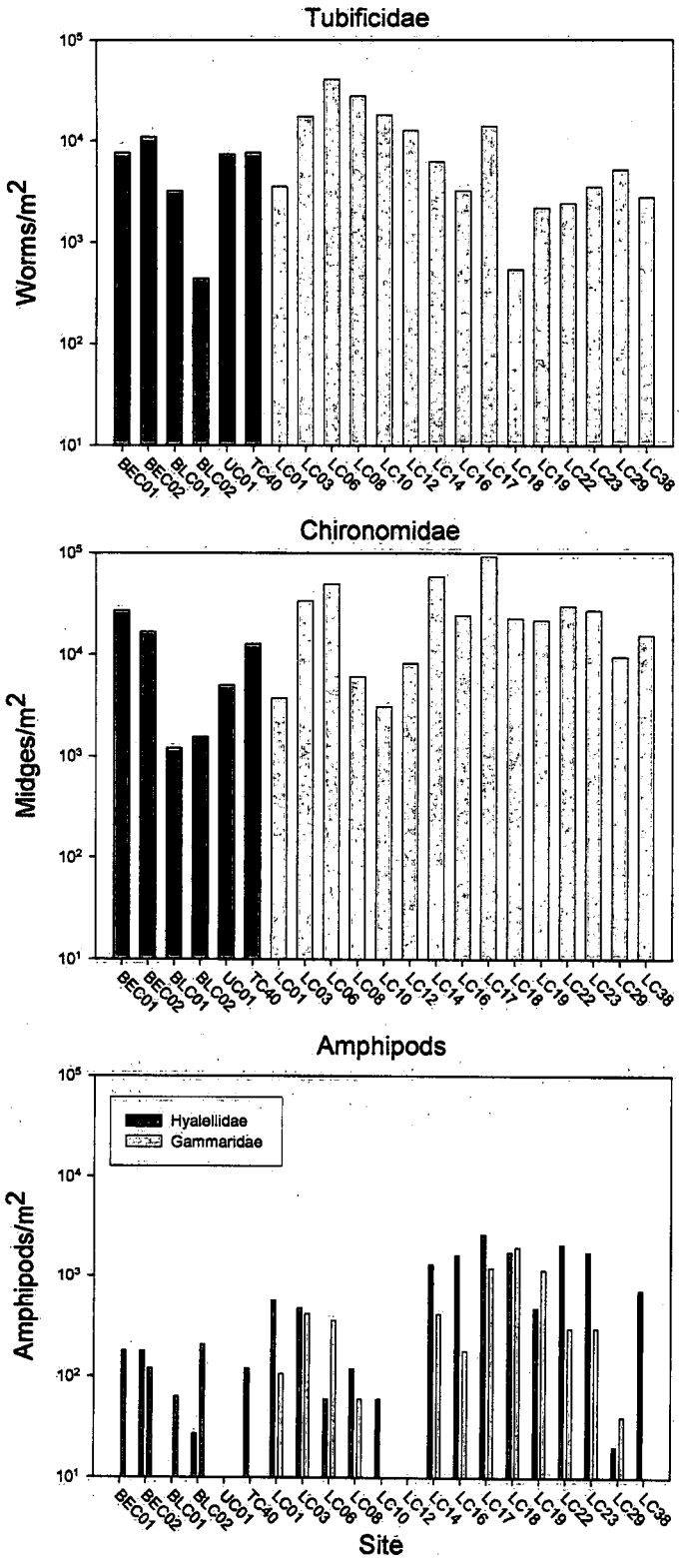


Figure 9. Toxicological response of Lyons Creek and reference sites represented by 2-dimensional hybrid multidimensional scaling (HMDS) (stress = 0.07). The directions of maximum correlations of toxicity endpoints and environmental variables with sites are shown as vectors.



Site	No. Taxa (macroinvertebrates)
BEC01	18
BEC02	25
BLC01	24
BLC02	20
UC01	17
TC40	18
LC01	27
LC03	22
LC06	17
LC08	14
LC10	14
LC12	11
LC14	23
LC16	15
LC17	28
LC18	23
LC19	19
LC22	22
LC23	19
LC29	16
LC38	23

Figure 10. Abundance (per m^2) of dominant macroinvertebrate taxa and total number of taxa present at reference sites (green) and Lyons Creek sites (grey).

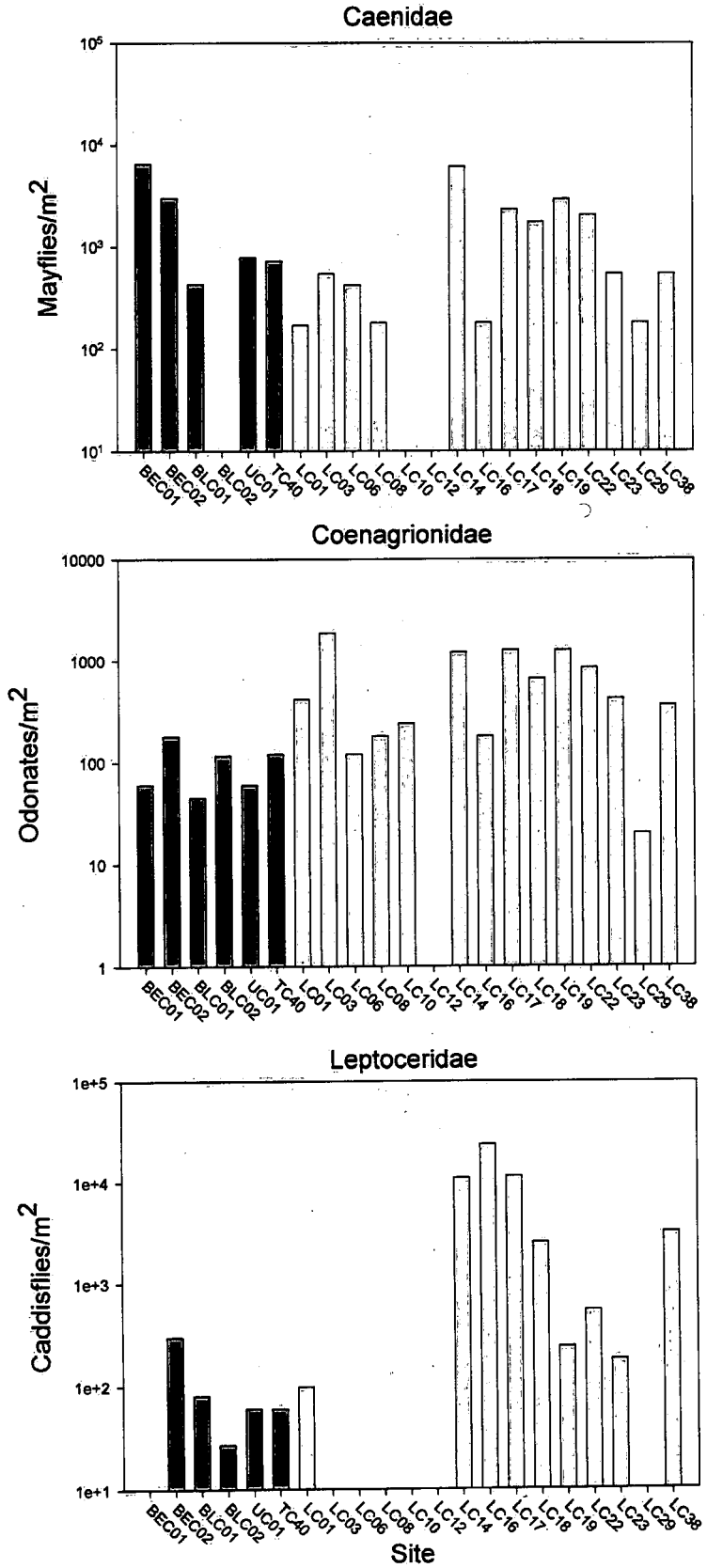


Figure 10. Continued.

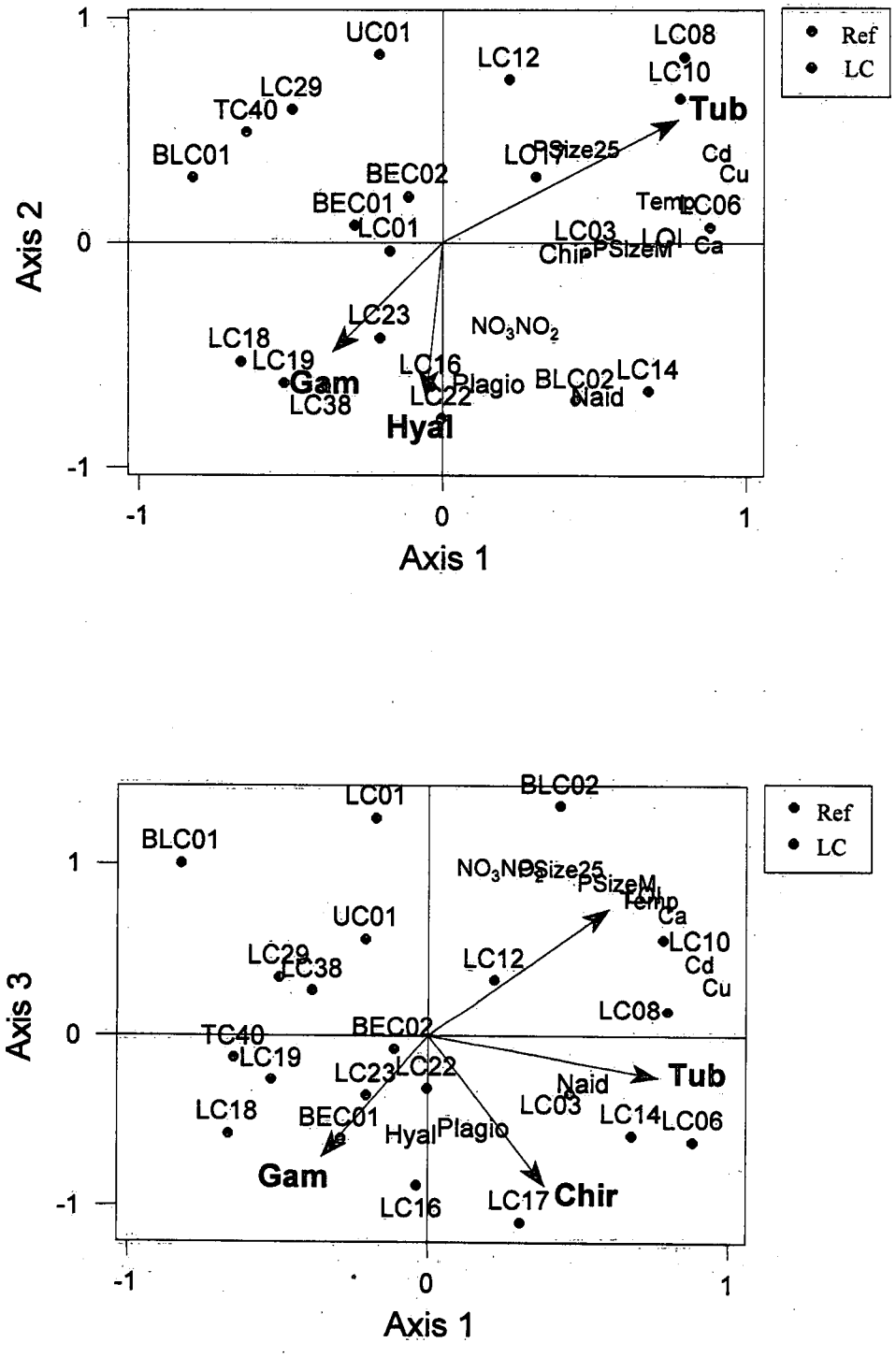


Figure 11. Ordination of Lyons Creek and reference community structure data represented by 3-dimensional hybrid multidimensional scaling (HMDS) (stress = 0.130). The directions of maximum correlations of community endpoints with sites are shown as vectors. [Tub = Tubificidae, Chir = Chironomidae, Naid = Naididae, Hyal = Hyalellidae, Gam = Gammaridae, Plagio = Plagiostomidae]

Table 1. Lyons Creek and reference creek site co-ordinates and depth (m).

Site	Year	Location	Site Depth	Easting	Northing	Comments
Reference						
BEC01	2002	Beaver Creek	0.47	662816.8	4757775.5	No tissue
BEC02	2002	Beaver Creek	0.39	661751.9	4767807.7	No tissue
BLC01	2002	Black Creek	0.60	660280.1	4757785.9	
BLC02	2002	Black Creek	0.66	660139.3	4757674.1	
UC01	2003	Ushers Creek	0.25	661074	4768269	
TC40	2003	Tee Creek	0.33	654026	4765367	
Lyons Creek						
LC01	2002	Upstream Welland Pipe	2.00	645128.9	4759516.0	
LC03	2002	Downstream Welland Pipe	0.91	645093.7	4759466.8	
LC06	2003	Between Welland Pipe and Hwy 140	0.41	645233	4759641	No tissue
LC08	2003	Between Welland Pipe and Hwy 140	0.36	645404	4759768	
LC10	2003	Between Welland Pipe and Hwy 140	0.25	645617	4759964	No tissue
LC12	2002	MOE Transect 6	0.90	645933.5	4759889.6	
LC14	2003	Between Welland Pipe and Hwy 140	0.41	646252	4759917	
LC16	2002	Upstream Hwy 140	0.46	646312.8	4760194.9	
LC17	2002	Downstream Hwy 140	0.40	646441.2	4760309.2	
LC18	2003	Upstream Railway	0.35	646681	4760589	
LC19	2003	Downstream Railway	0.30	646759	4760662	
LC22	2003	Upstream Doan's Ridge Road	0.46	647894	4761488	No tissue
LC23	2003	Downstream Doan's Ridge Road	0.46	648039	4761691	No tissue
LC29	2002	Downstream McKenny Rd	0.56	649666.4	4762066.8	
LC38	2002	Downstream QEW	0.47	655075.7	4766330.0	

Table 2. List of environmental variables measured at each site.

Field	Water	Sediment	Biota
Northing	Alkalinity	Major Oxides	PCBs
Easting	Conductivity	Trace Elements	PAHs
Site Depth	Dissolved Oxygen	Percent Clay, Silt, Sand, & Gravel	Lipids (2003 only)
	pH	Total Phosphorus	
	Temperature	Total Nitrogen	
	Total Kjeldahl Nitrogen	Total Organic Carbon	
	Total Phosphorus	Loss on Ignition	
	NH_3, NO_3/NO_2	PCBs, PAHs, OCs	

Table 3. Literature derived biomagnification factors (BMFs) for the receptors of concern. For each receptor, the number of trophic levels removed from benthic invertebrates (Level 1) is indicated. To calculate the food chain multipliers (FCM) for the bass the lowest, medium and highest BMFs were estimated from the combined data of all trophic 2 receptors (i.e., bullhead, carp and bluegill). For the mink, BMFs are based on lipid normalized PCB concentrations in invertebrate and fish tissues.

Total PCBs

Receptor	Predator Type	Trophic levels of transfer	BMFs (low | med | high)	FCM (low | med | high)
Brown Bullhead	Benthivorous fish	1 – 2	2.247 | 3.996 | 5.342	2.247 | 3.996 | 5.342
Carp		1 – 2	1.992 | 20.916 | 36.364	1.992 | 20.916 | 36.364
Goldeneye	Benthivorous bird	1 – 2	17.27 | 21.67 | 25.00	17.27 | 21.67 | 25.00
Bluegill	Benthivorous/	1 – 2	1.053 | 2.851 | 6.438	1.053 | 2.851 | 6.438
	Small Piscivorous fish	1 – 2 – 3	N/D	-
Largemouth Bass	Large piscivorous fish	1 – 2 – 3	1.053 | 7.502 | 36.364 × (1.097 | 4.583 | 12.650)	1.155 | 34.382 | 460.017
		1 – 2 – 3 – 4	N/D	-
Mink	Piscivorous mammal	1 – 2 – 3	1.12 | 2.38 | 5.31 × (1.65 | 2.45 | 3.36)	1.85 | 5.83 | 17.84
(lipid normalized)		1 – 2 – 3 – 4	N/D	-

N/D = not determined

Table 4. Concentration of PCBs in top 10 cm of sediment (µg/g dry weight). Severe Effect Levels (SELs) are included.

Area	Site	Total PCBs	SEL PCBs
Ref. Creeks	**BEC01**	**0.016**	**17.0**
	BEC02	**0.012**	**56.2**
	BLC01	**0.004**	**29.7**
	BLC02	**0.003**	**21.2**
	UC01	**0.013**	**33.4**
	TC40	**0.011**	**36.6**
Lyons Creek	LC01	0.021	10.1
	LC03	12.548	14.8
	LC06	0.600	25.4
	LC08	4.681 (6.102)[a]	22.3
	LC10	2.484	33.4
	LC12	7.396	25.4
	LC14	2.319	23.9
	LC16	1.513 (1.024)[b]	35.0
	LC17	1.303	27.6
	LC18	0.467 (0.545)[a]	35.0
	LC19	1.025	27.0
	LC22	0.318 (0.293)[a]	26.5
	LC23	0.889	33.4
	LC29	0.440 (0.424)[a] (0.411)[a] (0.244)[b]	29.2
	LC38	0.020 (0.016)[b]	56.7

[a] field replicate; [b] yearly replicate

Table 5. Concentration of PCBs in benthic invertebrates (µg/g dry weight).

Area	Site	Total PCBs			
		Chironomid	Amphipod	Oligochaete	Odonate
Ref. Creeks	**BEC01**	-[a]	-[a]	-[a]	-[a]
	BEC02	-[a]	-[a]	-[a]	-[a]
	BLC01	**0.126**	**0.166**	-[b]	**0.231**
	BLC02	**0.208**	**0.222**	-[b]	**0.404**
	UC01	**0.253**	**0.105**	**0.051**	**0.090**
	TC40	**0.117**	**0.048**	**0.301**	**0.136**
Lyons Creek	LC01	-[b]	0.675	0.336	0.227
	LC03	-[b]	7.232	0.843	0.021
	LC06	-[a]	-[a]	-[a]	-[a]
	LC08	0.677	3.429	0.312	0.220
	LC10	-[a]	-[a]	-[a]	-[a]
	LC12	4.622	10.926	52.577	1.009
	LC14	1.466	3.332	0.939	0.274
	LC16	1.185	2.817	2.439	0.514
	LC17	4.384	5.171	4.002	0.299
	LC18	1.524	1.872	2.290	0.350
	LC19	1.911	2.126	0.699	0.221
	LC22	-[a]	-[a]	-[a]	-[a]
	LC23	-[a]	-[a]	-[a]	-[a]
	LC29	0.758	0.609	2.884	0.275
	LC38	0.936	0.082	0.907	0.135

[a] benthos not collected [b] taxa not analyzed

Table 6. Prediction of whole body concentrations of total PCBs in biota based on sediment PCB concentration alone ("A" models), and sediment PCB concentration + other sediment physico-chemical variables ("B" models). The groups of multiple predictors listed are from the models that best predicted $[PCB]_{inv}$ using sediment and water variables. $[PCB]_{sed}$ was retained in all models.

Response ($[PCB]_{inv}$)	Model	Predictor ([X])	Coefficient	P (predictor)	r^2	Adj. r^2	P (regression)
Total PCBs	A	Total PCBs	0.3382	0.001	0.656	0.625	0.001
Chironomid	B	Total PCBs	0.3294	<0.001	0.791	0.749	<0.001
		pH	0.2726	0.030			
Total PCBs	A	Total PCBs	0.5506	<0.001	0.883	0.874	<0.001
Amphipod	B	Total PCBs	0.5508	<0.001	0.929	0.918	<0.001
		pH	4.4520	0.016			
Total PCBs	A	Total PCBs	0.3081	0.082	0.250	0.182	0.082
Oligochaete	B	Total PCBs	0.4531	0.004	0.855	0.783	0.002
		pH	0.5320	0.004			
		Total P (water)	1.5223	0.007			
		Sand	-2.0258	0.032			
Total PCBs	A	Total PCBs	0.0009	0.991	0.000	0.000	0.991
Odonate	B	-	-	-	-	-	-

Table 7a. Predicted PCB concentrations (µg/g wet weight) in Lyons Creek fish receptors. Highlighted values exceed the IJC tissue objective (0.1 µg/g ww) applicable for fishes.

Receptor		Brown Bullhead			Carp			Bluegill			Largemouth Bass		
Mean PCBs at Hwy 140[a]		0.140			1.164			0.188			0.278		
Mean PCBs Downstream QEW[a]		0.068			0.076			0.024			0.044		
Area	**Site**	**min**	**med**	**max**	**min**	**med**	**max**	**min**	**med**	**max**	**min**	**med**	**max**
Reference	**BLC01**	**0.03**	**0.08**	**0.13**	**0.03**	**0.41**	**0.89**	**0.02**	**0.06**	**0.16**	**0.02**	**0.68**	**9.80**
	BLC02	**0.04**	**0.11**	**0.19**	**0.04**	**0.58**	**1.30**	**0.02**	**0.08**	**0.23**	**0.02**	**0.95**	**14.28**
	UC01	**0.02**	**0.06**	**0.13**	**0.02**	**0.34**	**0.88**	**0.01**	**0.05**	**0.16**	**0.01**	**0.56**	**9.71**
	TC40	**0.01**	**0.10**	**0.23**	**0.01**	**0.51**	**1.57**	**0.01**	**0.07**	**0.28**	**0.01**	**0.84**	**17.26**
Lyons Creek	LC01	0.05	0.22	0.48	0.04	1.16	3.26	0.02	0.16	0.58	0.03	1.91	35.84
	LC03	0.01	1.75	4.67	0.01	9.17	31.79	0.00	1.25	5.63	0.00	15.07	349.72
	LC08	0.06	1.16	2.97	0.05	6.08	20.22	0.03	0.83	3.58	0.03	9.99	222.45
	LC12	0.33	12.57	32.85	0.29	65.82	223.59	0.15	8.97	39.59	0.17	108.19	2459.52
	LC14	0.06	1.02	2.59	0.05	5.36	17.66	0.03	0.73	3.13	0.03	8.82	194.21
	LC16	0.12	0.97	2.31	0.11	5.10	15.73	0.06	0.70	2.78	0.06	8.39	172.98
	LC17	0.06	1.76	4.58	0.05	9.22	31.16	0.03	1.26	5.52	0.03	15.16	342.72
	LC18	0.07	0.63	1.52	0.06	3.32	10.36	0.03	0.45	1.83	0.04	5.45	113.95
	LC19	0.05	0.68	1.69	0.05	3.56	11.51	0.03	0.49	2.04	0.03	5.85	126.60
	LC29	0.06	0.67	1.66	0.05	3.52	11.28	0.03	0.48	2.00	0.03	5.79	124.06
	LC38	0.02	0.22	0.55	0.02	1.17	3.71	0.01	0.16	0.66	0.01	1.93	40.82

[a] MOE 2003b

Table 7b. Predicted PCB concentrations in Lyons Creek wildlife receptors.

Area	Site	Goldeneye (µg/g ww)			Mink (µg/g lipid)		
		min	med	max	min	med	max
Reference	**BLC01**	**0.26**	**0.43**	**0.61**	**0.32**	**1.44**	**5.77**
	BLC02	**0.34**	**0.60**	**0.89**	**0.42**	**2.04**	**8.41**
	UC01	**0.14**	**0.35**	**0.61**	**0.09**	**0.97**	**5.04**
	TC40	**0.10**	**0.53**	**1.08**	**0.19**	**1.09**	**4.85**
Lyons Creek	LC01	0.37	1.21	2.24	0.53	5.38	27.85
	LC03	0.04	9.50	21.86	0.06	44.50	271.74
	LC08	0.43	6.30	13.90	0.38	28.85	172.84
	LC12	2.50	68.19	153.72	3.53	118.58	691.65
	LC14	0.47	5.56	12.14	0.67	25.71	150.90
	LC16	0.96	5.29	10.81	1.35	24.09	134.41
	LC17	0.43	9.55	21.42	0.61	44.47	266.30
	LC18	0.56	3.44	7.12	0.78	14.60	81.78
	LC19	0.41	3.69	7.91	0.58	16.99	98.37
	LC29	0.46	3.65	7.75	0.65	6.72	34.89
	LC38	0.18	1.22	2.55	0.33	3.04	15.42

Table 8. Percent survival and growth (mg) in sediment toxicity tests and BEAST difference-from-reference band. Toxicity, based on numeric criteria is highlighted yellow; potential toxicity is italicized.

Site	*C. riparius* growth	*C. riparius* %survival	*H. azteca* growth	*H. azteca* %survival	*Hexagenia* growth	*Hexagenia* %survival	*T. tubifex* No. cocoons/ adult	*T. tubifex* %cocoons hatched	*T. tubifex* %survival	*T. tubifex* No. young/ adult	BEAST BAND
GL Ref. Mean	**0.35**	**87.1**	**0.50**	**85.6**	**3.03**	**96.2**	**9.9**	**0.57**	**97.8**	**29.0**	1
BEC01	**0.21**	**77.3**	**0.65**	**94.7**	**1.97**	**98.0**	**8.7**	**0.55**	**100.0**	**11.8**	1
BEC02	***0.20***	**80.0**	**0.38**	**90.7**	**1.06**	**100.0**	**10.3**	**0.59**	**100.0**	**14.5**	1
BLC01	**0.21**	**73.3**	**0.37**	**90.7**	**1.62**	**100.0**	**8.4**	**0.93**	**100.0**	**13.3**	1
BLC02	**0.23**	**96.0**	**0.51**	**85.3**	**0.99**	**94.0**	**5.7**	**0.87**	**100.0**	***5.2***	2
UC01	**0.47**	**91.7**	**0.52**	**94.7**	**6.55**	**100.0**	**10.5**	**0.57**	**100.0**	**20.4**	1
TC40	**0.56**	**78.7**	**0.44**	**93.3**	**5.58**	**100.0**	**9.9**	**0.52**	**95.0**	**17.5**	1
LC01	0.30	96.0	0.64	90.7	3.22	94.0	9.8	0.65	100.0	14.4	1
LC03	**0.06**	**38.7**	0.27	**40.0**	**-0.09**	**2.0**	**4.1**	0.87	90.0	***2.3***	**4**
LC06	0.47	88.0	0.45	90.0	4.90	100.0	10.8	0.64	100.0	19.2	1
LC08	0.24	78.3	***0.15***	**34.7**	**-0.02**	**4.0**	**0.2**	1.00	**35.0**	**0.0**	**4**
LC10	0.38	84.0	0.25	88.0	*0.60*	*84.0*	8.6	0.50	100.0	12.5	1
LC12	***0.19***	***64.0***	0.37	75.0	**-0.01**	**46.0**	9.1	0.62	100.0	11.2	**4**
LC14	0.31	68.0	0.25	83.3	3.07	94.0	7.7	0.64	95.0	11.5	2
LC16	***0.20***	93.3	0.32	75.0	3.35	96.0	10.9	0.56	100.0	27.2	1
LC17	0.23	89.3	0.47	76.0	3.72	98.0	10.0	0.51	100.0	25.1	1
LC18	0.41	90.7	0.71	94.7	5.09	100.0	9.7	0.66	95.0	18.9	1
LC19	0.40	93.3	0.74	92.0	5.45	100.0	9.2	0.62	100.0	17.8	1
LC22	0.36	94.7	0.53	92.0	5.75	100.0	9.0	0.47	95.0	19.1	1
LC23	0.37	89.3	0.47	88.0	4.64	100.0	8.0	0.59	100.0	17.7	1
LC29	***0.19***	88.3	0.31	83.0	2.90	100.0	10.2	0.57	100.0	24.5	1
LC38	0.21	78.7	0.37	68.0	3.16	100.0	11.0	0.54	100.0	21.4	1
Non-toxic[a]	0.49 – 0.21	67.7	0.75 – 0.23	67.0	5.00 – 0.90	85.5	12.4 – 7.2	0.78 – 0.38	88.9	46.3 – 9.9	-
Potentially toxic	*0.20 – 0.14*	*67.6 – 58.8*	*0.22 – 0.10*	*66.9–57.1*	*0.80 – 0*	*85.4 – 80.3*	*7.1 – 5.9*	*0.38 – 0.28*	*88.8– 84.2*	*9.8 – 0.8*	-
Toxic	**< 0.14**	**<58.8**	**< 0.10**	**< 57.1**	**neg**	**< 80.3**	**< 5.9**	**< 0.28**	**< 84.2**	**<0.8**	-

[a] Upper limit for non-toxic category is set using 2 × SD of the mean and indicates excessive growth or reproduction.

Table 9. Decision matrix for weight-of-evidence categorization of Lyons Creek sites based on three or four lines of evidence. For the sediment chemistry column, sites with exceedences of the Probable Effect Level for PCBs are indicated by "●", and sites with exceedences of the Lowest Effect Level for PCBs by "⊙". For the toxicity column, sites determined from BEAST analyses as toxic/severely toxic are indicated by "●"; sites determined as potentially toxic by "⊙". For the benthos alteration column, sites determined from ANOVAs as significantly difference from reference creek sites are indicated by "●". Sites with no SQG exceedences, no sediment toxicity, or benthic communities that equivalent to reference conditions are indicated by "O". For the biomagnification column, both the minimum (Min.) and intermediate (Inter.) exposure and uptake scenarios are provided.

Site	Response for individual decision elements					Assessment
	Sediment PCB Chemistry	Toxicity	Benthos Alteration	Biomagnification Potential (minimum)	Biomagnification Potential (intermediate)	
LC01	O	O	O	O	⊙	Min. - No further actions needed. Inter. - Fully assess risk of biomagnification.
LC03	●	●	O	O	⊙	Min. - Determine reasons for sediment toxicity. Inter. - Above plus fully assess risk of biomagnification.
LC06	●	O	O	N/A	N/A	No further actions needed based on 3 lines of evidence - potential for biomagnification not assessed.
LC08	●	●	O	O	⊙	Min. - Determine reasons for sediment toxicity. Inter. - Above plus fully assess risk of biomagnification.
LC10	●	O	O	N/A	N/A	No further actions needed based on 3 lines of evidence - potential for biomagnification not assessed.
LC12	●	●	●	⊙	⊙	Both - Management actions required.
LC14	●	⊙	O	O	⊙	Min. - Determine reasons for sediment toxicity. Inter. – Above plus fully assess risk of biomagnification.
LC16[1]	●	O	O	⊙	⊙	Fully assess risk of biomagnification.
LC17	●	O	O	O	⊙	Min. - No further actions needed. Inter. - Fully assess risk of biomagnification.
LC18	●	O	O	O	⊙	Min. - No further actions needed. Inter. - Fully assess risk of biomagnification.
LC19	●	O	O	O	⊙	Min. - No further actions needed. Inter. - Fully assess risk of biomagnification.
LC22	●	O	O	N/A	N/A	No further actions needed based on 3 lines of evidence - potential for biomagnification not assessed.
LC23	●	O	O	N/A	N/A	No further actions needed based on 3 lines of evidence - potential for biomagnification not assessed.
LC29	●	O	O	O	⊙	Min. - No further actions needed. Inter. - Fully assess risk of biomagnification.
LC38	O	O	O	O	⊙	Min. - No further actions needed. Inter. - Fully assess risk of biomagnification.

N/A = not applicable (tissue not collected)

[1] PCBs in fish collected at Highway 140 are at levels that warrant fish consumption advisories.

APPENDIX A. Literature Review of PCB Biomagnification Factors

1.0 Introduction

This literature review was carried out to provide supporting information for the assessment of risk of biomagnification of total PCBs from contaminated sediments in Lyons Creek, Welland, Ontario (Niagara River Area of Concern). Biomagnification factors (BMFs), predator-prey factors (PPFs), and trophic transfer coefficients (TTCs) were obtained or derived from the literature for the calculation of total PCB concentrations in different trophic levels of a simple benthic freshwater food chain model (Figure A1).

1.1 Terminology

Biomagnification is the process at by which the chemical concentration in an organism exceeds that in the organism's diet, due to dietary absorption (Gobas and Morrison, 2000). The biomagnification factor (BMF) is an empirically-derived measure of the rate of contaminant transfer between the organism's diet and the organism, and is expressed as the ratio of chemical concentration in the organism to the concentration in its diet (Gobas and Morrison, 2000). The synonymous terms predator-prey factor (PPF) and trophic transfer coefficient (TTC) are also found in the literature (USEPA, 1997a; Suedel et al., 1994). A food chain multiplier (FCM) is used to quantify the increase in contaminant body burden through uptake from the food chain, but is defined as the factor by which a substance at higher trophic levels exceeds the bioconcentration factor (BCF) at trophic level 1 (NCASI, 1999; USEPA, 1997a). Therefore, it does not necessarily apply to a specific trophic transfer, and may be a multiple of more than one BMF. BMFs, TTCs, and PPFs are unitless, and the concentrations used to derive them are usually expressed in units of mass of chemical per kg of the organism, and mass of chemical per kg of food, respectively (Gobas and Morrison, 2000). These concentrations can be expressed on a wet weight or dry weight basis (Gobas and Morrison, 2000). BMFs, TTCs, and PPFs can be applied to specific trophic levels, as well as individual species in a food chain (USEPA, 1997b). The term BMF will be used in this document in reference to biomagnification factors, predator-prey factors, and trophic transfer coefficients acquired from the literature.

2.0 Methods

2.1 Literature Search

Obtaining the information required to estimate PCB concentrations in receptors involved reviewing published literature, unpublished reports, databases, web pages and any other sources of data on BMFs relevant to the benthic invertebrate taxa and receptors; assessing the quality of the BMF data; and tabulating BMFs and estimates of their variability, together with information on the BMF's determination (e.g., location of study, organisms involved, proportion of receptor's diet that is invertebrates, effects of cofactors (if any), assumed ingestion rates and home ranges). The following criteria were applied to screen literature to obtain either BMFs or candidate datasets for calculating BMFs, after Suedel et al. (1994) and Gobas and Morrison (2000):

- If organisms that were presented were not from a logical food chain, or no evidence was presented that the feeding relationship between predator and prey was a functional feeding relationship, the data were not used.
- Mean concentrations of total PCB needed to be presented for both predator and prey, and in comparable units.
- BMFs involving PCB concentrations in feathers or fur of predators were excluded.
- Unless evidence of comparability could be found, studies from non-freshwater systems or with non-comparable species were not used. More information is presented below on the assessment of comparability of different systems and species.

There were few studies that quoted BMF estimates specifically for most of the receptor species and feeding relationships defined in Figure A1. Of the small number of studies that calculated BMFs that were directly comparable in part to the food chain model, all were from freshwater pelagic food webs. It was necessary to use the most relevant studies to obtain BMFs and document the relative comparability of different species and ecosystems to those presented in the study design for this assessment. Information to support substitutions of receptor with comparable species from the literature (in applying BMF estimates) is presented in Tables A3 - A12. Species were

considered the most qualitatively similar when they occupied similar habitats, had similar feeding habits and dietary composition, similar range, similar feeding substrate, and similar food ingestion:body weight ratio. Sources for this information were CCME (1999), CWS (2002), Sample and Suter (1999), Scott and Crossman (1973), and USEPA (1997c). A breakdown of the number of BMFs obtained/calculated per feeding relationship, and the range of corresponding BMF values is presented in Table A1.

The literature search was done using typical methods of electronic database and chain-of-citation searches as well as consultation with leading researchers in the field of PCB ecotoxicology and risk assessment. The following electronic databases were used to search primary literature, secondary literature, grey literature, and internet resources:

- ISI Current Contents Connect
- US Environmental Protection Agency (USEPA)- various databases of government publications
- US Army Corp. of Engineers (USACE)- various databases of government publications
- Integrated Risk Information System (IRIS)
- Oak Ridge National Laboratory (ORNL) publications
- GLIER DRCCC

Trophic Level

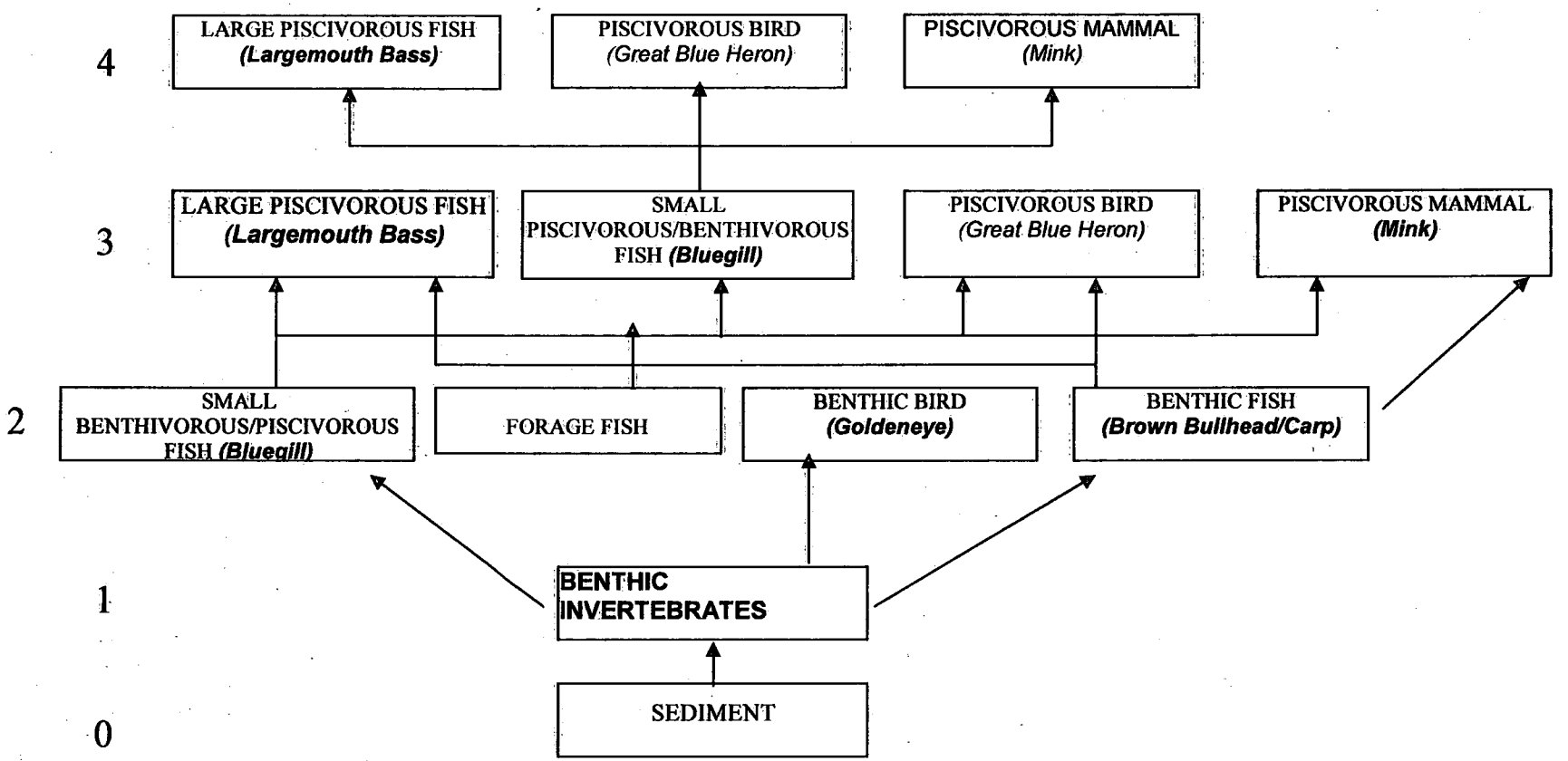


Figure A1. Generalized foodweb model for the assignment of trophic level to biomagnification factor estimates. Receptors used in Lyons Creek modelling are highlighted.

2.2 Assigning Trophic Levels to Receptor Species

Discrete trophic levels were applied using the food chain model (Figure A1). This was done to allow comparison of BMFs from different systems/foodwebs, as well as to conceptualize the transfer and magnification of total PCBs in the Lyons Creek scenario. However, the use of discrete trophic levels may lead to lower estimates of BMFs. An excellent discussion about the effects of omnivory on trophic position is found in Vander Zanden and Rasmussen (1996). In short, omnivory is common in aquatic communities (for example, up to 50% in pelagic food webs), and the use of discrete variables to represent trophic position will not adequately account for omnivory. When omnivory is integrated with the use of a continuous measurement of trophic position (i.e., using stable isotope methods), estimates of BMFs will generally be higher for each discrete trophic level (Vander Zanden and Rasmussen 1996). Unfortunately, this literature survey did not yield any stable isotope studies on benthic freshwater food webs, and therefore system-specific BMFs based on continuous trophic position could not be obtained for lower trophic levels. It was also suggested that much of the uncrtainty around applying BMFs from different systems may be due to an oversimplification of predator-prey relationships by using discrete trophic levels (Vander Zanden and Rasmussen, 1996).

2.3 Selecting Biomagnification Factor Estimates or Candidate Datasets from the Literature

The following criteria were applied to screen literature to obtain either BMFs or candidate datasets for calculating BMFs, after Suedel et al. (1994) and Gobas and Morrison (2000):

- If organisms that were presented were not from a logical food chain, or no evidence was presented that the feeding relationship between predator and prey was a functional feeding relationship, the paper was not used. One exception to this rule was made in selecting studies of mink fed diets of different proportions of contaminated and uncontaminated fish (Platanow and Karstad 1973, Hornshaw et al. 1983, Wren et al. 1987, Tillitt et al 1996, Halbrook et al. 1999), since there was a reasonable likelihood that these fish species would have been part of their diet.
- Mean concentrations of total PCBs needed to be presented for both predator and prey, and in comparable units.
- Unless evidence of comparability could be found, studies from non-freshwater systems or with non-comparable species were not used. More information is presented below on the assessment of comparability of different systems and species.

2.4 Calculation of Biomagnification Factors from Candidate Datasets

Biomagnification factors were calculated from mean concentrations of total PCBs from the literature using the equation (Gobas and Morrison, 2000):

$$BMF = C_B/C_D$$

where:
C_B= mean contaminant concentration in the consumer (receptor) species
C_D= mean contaminant concentration in the diet of the organism
In all cases where BMFs were calculated from mean concentrations, the calculation was for the mean concentrations from two trophic levels with a functional feeding relationship which was defined and demonstrated in the study. Where results were presented for a number of different locations (ie- several different lakes), BMFs were calculated for each location and then averaged, as opposed to averaging the mean concentrations from all locations to calculate a BMF.

2.5 Comparability of Species, Systems and PCBs

Some studies which quoted BMF estimates for different receptor species and feeding relationships as defined in Figure A1. All the studies which calculated BMFs that were directly comparable in part to the food chain model were from freshwater pelagic foodwebs. It was important to document the relative comparability of different species to those presented in the study design for this assessment. Information to support substitutions of receptor species for comparable species from the literature (in applying BMF estimates) is provided in a similar studies examining Hg biomagnification (Grapentine et al. 2003a,b). Species were considered the most qualitatively similar when they

occupied similar habitats, had similar feeding habits and dietary composition, similar range, similar feeding substrate, and similar food ingestion:body weight ratio. Sources for this information were CCME (1999), CWS (2002), Sample and Suter (1999), Scott and Crossman (1973), and USEPA (1997c).

Applying BMFs calculated from one system to another is controversial, since rates of trophic transfer of total PCBs are thought to vary due to abiotic and biotic factors (USEPA, 2001). Additionally, congener specific PCB vs. total PCB analysis based on Aroclor standards may reveal different patterns (Rasmussen et al. 1990). Bioaccumulation will vary greatly depending on the degree and pattern of chlorine substitution with PCBs containing 4 or less chlorine atoms being more rapidly metabolized or eliminated than PCBs with 5-7 chlorine atoms (Niimi and Oliver 1983). Factors affecting the bioaccumulation of PCBs include the productivity of an ecosystem (total phosphorus concentrations, chlorophyll *a*, and transparency) suspended solid concentrations and organic carbon content of the sediment (Roe et al. 2000) as well as exposure route, lipid content of organism, food chain length, horizontal food web structure, feeding mechanisms of organisms at lower trophic levels, and the age/size/weight or metabolic rates of individuals (Kucklick and Baker 1998, Roe et al. 2000, Environment Canada, 2001; Power et al., 2002; USEPA, 2000).

Studies from marine, arctic marine, and tropic freshwater were not used to select or derive BMFs in this study.

3.0 Results

A total of 172 references were examined in detail to yield BMFs, datasets to calculate BMFs, or to provide supporting information in applying BMFs. Of those 172, only 17 yielded appropriate BMFs or datasets, following guidelines set out in Section 2 above. Along with BMF estimates, the following supporting information was gathered where available:

- Range, standard deviation, or standard error of BMF estimates
- Trophic level of predator/receptor
- Type of study (field, laboratory, modelling, review)
- Prey species
- Predator species
- PCB parameter (total PCBs, sum of PCB congeners or Aroclors)
- Lipid normalization or not
- Scope of study (ie- number of lakes sampled)
- Location of study
- Biological medium sampled
- Relative age/size of organisms sampled
- Reference from which BMF or dataset came from
- Comments

These results are reported in Table A1.A breakdown of the number of BMFs obtained/calculated per feeding relationship, and the range of corresponding BMF values is presented in Table A2.

Table A1. Breakdown of results of literature review for each hypothetical feeding relationship for each trophic level.

TROPHIC LEVEL 2 – BENTHIVOROUS FISH

Value (lipid norm)	ww/ dw	Prey or Species	Predator	Section of Predator	PCB	Lake/ River	Location	Year	Reference	Comments
3.982	dw	*P. hoyi*	Deepwater sculpin	whole fish	homologues	Michigan	Southeastern	1982	Evans et al. 1991	
2.247	dw	*P. affinis*	Slimy sculpin	whole fish	total congeners	Ontario	Eastern	1977-79	Borgmann and Whittle 1983	
2.789	ww	Chironomid/mayfly/ oligochaete	Brown bullhead	Dorsal muscle	total PCBs	Detroit R.	Peche Is.	2000-01	GLIER 2001	
3.299	ww	*T. tubifex* *L. hoffmeisteri* *P. affinis*	Slimy sculpin	Composite of 5 fish – section?	Sum congeners (27)	Niagara R./ Ontario	mouth/ Grimsby	1985-86	Oliver and Niimi 1988	
5.342 (5.31)	ww	*D. hoyi*	Slimy sculpin	homogenates of whole fish	total congeners	Ontario	Cobourg	1992	Metcalfe and Metcalfe 1997	Lipid normalized values given
(1.46)	ww	*D. hoyi*	Sucker	homogenates of whole fish	total congeners	Ontario	Cobourg	1992	Metcalfe and Metcalfe 1997	Lipid normalized values given
5.281	ww	*D. hoyi*	Slimy sculpin	whole composites	1:1:1 mixture Aroclors 1242:1254: 1260:	Ontario	Grimsby/Port Credit/Cobourg	1992	Kiriluk et al. 1995	
4.131 (2.43)	ww	D. hoyi	Sculpin	Whole fish	Total PCBs	Ontario	Grimsby	1992	Niimi 1996	Lipid values provided
4.00 (3.66)	ww	D. hoyi	Slimy Sculpin	Whole fish	Total PCBs	Superior	Apostle Islands	1998	Wong et al. 2004	Lipid values provided
4.80 (2.33)	ww	D. hoyi	Deepwater Sculpin	Whole fish	Sum 103 congeners	Michigan	Grand Traverse Bay	1997	Stapleton and Baker 2003	Lipid values provided
4.09 (1.12)	ww	D. hoyi	Slimy Sculpin	Whole fish	Total PCBs	Superior	Keweenaw Peninsula	1994	Kucklick and Baker 1998	Lipid values provided
36.364	ww	Oligochaete	Carp	Homogenized whole fish	Sum congeners (72)	Detroit R		1981	Smith et al. 1985	
1.992	ww	Chironomid	Carp	Dorsal muscle	total PCBs	Detroit R	Celeron Is.	2000-01	GLIER 2001	
11.087	ww	Chironomid/mayfly/ oligochaete	Carp	Dorsal muscle	total PCBs	Detroit R	Turkey Is.	2000-01	GLIER 2001	
34.221	ww	Chironomid/mayfly/ oligochaete	Carp	Dorsal muscle	total PCBs	Detroit R	Peche Is.	2000-01	GLIER 2001	

Table A1. Continued.

TROPHIC LEVEL 2 – DIVING DUCK

Value	ww/ dw	Prey	Predator	Section of Predator	PCB	Lake/ River	Location	Year	Reference	Comments
22.73	ww	Oligochaete	Lesser Scaup	homogenized aliquots	total congeners (68)	Detroit R	Mud Is/Lower River	1981	Smith et al. 1985	no feathers, heads, feet, or stomach included in calculation
25.00	ww	Oligochaete	Greater Scaup	homogenized aliquots	total congeners (68)	Detroit R	Mud Is/Lower River	1981	Smith et al. 1985	"
17.27	ww	Oligochaete	Goldeneye	homogenized aliquots	total congeners (68)	Detroit R.	Mud Is/Lower River	1981	Smith et al. 1985	"

TROPHIC LEVEL 2 – BENTHIVOROUS/SMALL PISCIVOROUS FISH

Value	ww/ dw	Prey	Predator	Section of Predator	PCB	Lake/ River	Location	Year	Reference	Comments
6.438	ww	Invertebrates	Yellow perch	Dorsal muscle	total PCBs	Detroit R.	Peche Is.	2000-01	GLIER 2001	
2.231	ww	Invertebrates	Bluegill	Dorsal muscle	total PCBs	Detroit R.	Turkey Is.	2000-01	GLIER 2001	
1.053	ww	Crayfish	Bluegill	Dorsal muscle	total PCBs	Detroit R.	Goyers marina	2000-01	GLIER 2001	
1.680	dw	Chironomid/Snail	European Perch	homogenized	1:1:1 mixture of Aroclors 1242:1254:1260	River Seine	Paris		Teil et al. 1996	

Table A1. Continued.

TROPHIC LEVEL 3 – LARGE PISCIVOROUS FISH

Value	ww/ dw	Prey/Species	Predator	Section of Predator	PCB	Lake/ River	Location	Year	Reference	Comments
3.691	dw	Slimy Sculpin	Lake Trout	Whole fish	total PCBs	Ontario	Eastern	1977-79	Borgmann and Whittle 1983	Logarithmic means used in calculations
2.754	ww	Bluegill	Muskie	Dorsal muscle	total PCBs	Detroit R.	Turkey Is.	2000-01	GLIER 2001	
3.170	ww	White sucker	Walleye	Dorsal muscle	total PCBs	Detroit R.	LSC	2000-01	GLIER 2001	
3.643	ww	Yellow perch	Walleye	Dorsal muscle	total PCBs	Detroit R.	LSC	2000-01	GLIER 2001	
2.470	ww	Rock bass	Walleye	Dorsal muscle	total PCBs	Detroit R.	LSC	2000-01	GLIER 2001	
2.688	ww	Slimy Sculpin	Salmonid mixture	?	total congeners	Ontario/ Niagara R.	Grimsby Mouth	1985-86	Oliver and Niimi 1988	Mixture of Lake, Brown and Rainbow trout, Coho Salmon
12.650	ww	White Sucker	Lake Trout	Dorsal muscle	total congeners	Ontario	Cobourg	1992	Metcalfe and Metcalfe 1997	Lipid normalized values also given
1.097	ww	Slimy Sculpin	Lake Trout	Dorsal muscle	total congeners	Ontario	Cobourg	1992	Metcalfe and Metcalfe 1997	Lipid normalized values also given
2.845	ww	Slimy Sculpin	Lake Trout	Whole fish	1:1:1 mixture of Aroclors 1242:1254:1260	Ontario	Grimsby/ Port Credit/ Cobourg	1992	Kiriluk et al. 1995	
3.028	ww	Bluegill/Shiner	Largemouth Bass/Spotted Gar	Whole fish	Aroclor 1254	Lake Providence (Mississippi R.)	Northeastern Louisiana	1980	Niethammer et al. 1984	Geometric means used in calculations
4.333	ww	Bluegill	Largemouth Bass/Spotted Gar	Whole fish	Aroclor 1254	Lake Bruin (Mississippi R.)	Northeastern Louisiana	1980	Niethammer et al. 1984	Geometric means used in calculations
6.354	ww	Slimy Sculpin	Lake Trout	Whole fish	Total PCBs	Ontario	Grimsby	1992	Niimi 1996	Lipid values provided
5.440	ww	Slimy Sculpin	Lake Trout	Whole fish	Total PCBs	Superior	Apostle Islands	1998	Wong et al. 2004	Lipid normalized values
10.00	ww	Deepwater Sculpin	Lake Trout	Whole fish	Total congeners (103)	Michigan	Grand Traverse Bay	1997	Stapleton and Baker 2003	Lipid values provided

[a] mink used in all experiments were ranch bred

Table A1. Continued.

TROPHIC LEVEL 3 – MAMMAL

Value (range)	ww/ dw	Prey/Species	Predator	Section of Predator	PCB	Lake/ River	Location	Year	Reference	Comments
2.49 (1.99- 3.04)	ww	10%, 20%, 40% whole carp diets (Saginaw River)	Mink[a]	Liver – normalized to % lipid	Total PCBs	-	Michigan	1988	Tillitt et al. 1996	Normalized to feeding consumption also provided
(2.08-3.36)	?	?	Mink[a]	Liver	?	-	?	?	Platanow and Karstad 1973	Cited in Tillitt et al. 1996
(1.65-2.85)	ww	Prepared carp diets (Saginaw Bay)	Mink[a]	Adipose tissue – normalized to %lipid	Aroclor 1254	-	Michigan	1979	Hornshaw et al. 1983	Cited in Tillitt et al. 1996
5.33	ww	75% fish (mainly carp) 25% rand bred chow	Mink[a]	Liver	Aroclor 1260	-	Tennessee	1993-94	Halbrook et al. 1999	Not lipid normalized

[a] mink used in all experiments were ranch bred

Table A2. Summary of results of literature review for each hypothetical feeding relationship.

Feeding Relationship	Trophic levels of transfer	# of Estimates	PCB BMFs		
			Low	Medium	High
Benthic invertebrates to benthivorous fish[1]	1 – 2	10	2.25	3.99	5.34
Benthic invertebrates to benthivorous fish[2]	1 – 2	4	1.99	20.92	36.36
Benthic invertebrates to benthivorous waterfowl	1 – 2	3	17.27	21.67	25.00
Benthic invertebrates to benthivorous/small piscivorous fish	1 – 2	4	1.05	2.85	6.44
Benthivorous or forage fish to large piscivorous fish	2 – 3	14	1.09	4.58	12.65
Benthic invertebrates to benthivorous fish[3]	1 – 2	6	1.12	2.38	5.31
Benthivorous or forage fish to piscivorous mammal[3]	2 – 3	3	1.65	2.45	3.36

[1] bullhead
[2] carp
[3] lipid normalized values

APPENDIX B. Total PCBs (Wet Weight) in Resident Benthic Invertebrates

Table B1. Total PCBs in biota (converted to µg/g wet weight).

Area	Site	Total PCBs			
		Chironomid	Amphipod	Oligochaete	Odonate
Reference	**BEC01**	-[a]	-[a]	-[a]	-[a]
	BEC02	-[a]	-[a]	-[a]	-[a]
	BLC01	**0.0148**	**0.0183**	-[b]	**0.0245**
	BLC02	**0.0197**	**0.0247**	-[b]	**0.0357**
	UC01	**0.02427**	**0.01394**	**0.00804**	**0.01219**
	TC40	**0.01213**	**0.00584**	**0.04315**	**0.01210**
Lyons Creek	LC01	-[b]	0.0896	0.0406	0.0217
	LC03	-[b]	0.8743	0.1270	0.0026
	LC06	-[a]	-[a]	-[a]	-[a]
	LC08	0.07562	0.55612	0.03284	0.02514
	LC12	0.4348	1.3855	6.1488	0.1447
	LC14	0.14316	0.48553	0.09728	0.02744
	LC16	0.10718	0.43245	0.23566	0.05533
	LC17	0.4651	0.8568	0.4989	0.0250
	LC18	0.16339	0.26314	0.28488	0.03215
	LC19	0.21190	0.31651	0.08306	0.02376
	LC22	-[a]	-[a]	-[a]	-[a]
	LC23	-[a]	-[a]	-[a]	-[a]
	LC29	0.07232	0.07889	0.31014	0.02655
	LC38	0.07427	0.01019	0.10206	0.01519

[a] Resident benthos not collected at this site for tissue analysis
[b] Insufficient tissue

APPENDIX C. Sediment and Overlying Water Physico-Chemical Properties

Table C1. PCB congeners in Lyons Creek (LC) and reference creek (BEC, BLC, UC, TC) sediment.

CHEMICAL	BEC01	BEC02	BLC01	BLC02	UC01	UC01*	TC40	TC40*	LC01	LC03	LC06*
2,2',3,5'-tetrachlorobiphenyl	0.1	0.8	0.1	0.1	0.1	0.1	0.1	0.1	2.7	1000	15
2,2',4,5'-tetrachlorobiphenyl	0.8	0.6	0.1	0.1	0.1	0.1	0.1	0.1	1.3	830	53
2,2',5,5'-tetrachlorobiphenyl	0.7	1.4	0.1	0.1	0.1	0.1	0.1	2	2.6	1200	73
2,2',5-trichlorobiphenyl	2	2	2	2	2	2	2	2	2	730	5.6
2,2',6,6'-tetrachlorobiphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	140	0.2
2,2',6-trichlorobiphenyl	2	2	2	2	2	2	2	2	2	73	2
2,2'3,4,5'-pentachlorobiphenyl	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
2,2'3,44'5'-hexachlorobiphenyl	0.2	0.2	0.2	0.2	0.2	1	0.2	0.2	0.2	140	17
2,2'3,4'5'6-hexachlorobiphenyl	0.2	0.2	0.2	0.2	0.2	2.8	0.2	2	0.2	110	12
2,2'3,5,5'6-hexachlorobiphenyl	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	72	0.1
2,2'3,5',6-pentachlorobiphenyl	0.6	1	0.53	0.1	1.7	1.7	1.4	1.8	4.7	1400	120
2,2'4,4',5-pentachlorobiphenyl	0.5	1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	240	25
2,2'4,5,5'-pentachlorobiphenyl	0.7	0.5	0.1	0.1	1	1	0.1	0.1	0.96	490	43
2,2'4,6,6'-pentachlorobiphenyl	1.7	1.4	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
2,3,3'4,4'5-hexachlorobiphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	25	0.7
2,3,3'4,4'6-hexachlorobiphenyl	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	19	0.8
2,3,3'4,4'-pentachlorobiphenyl	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	230	0.1
2,3,3'4',6-pentachlorobiphenyl	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.77	720	4.1
2,3,3'44'5'-hexachlorobiphenyl	0.3	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
2,3,4,4',5-pentachlorobiphenyl	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	18	0.97
2,3',4',5-tetrachlorobiphenyl	1.3	0.7	0.37	0.1	0.9	0.9	0.9	1.2	2.5	1100	67
2',3,4-trichlorobiphenyl	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	1.3	510	4.8
2,3,4'-trichlorobiphenyl	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	210	0.5
2,3'4,4',5-pentachlorobiphenyl	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	320	33
2,3'4,4',6-pentachlorobiphenyl	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	6.4	2.3
2,4,4',5-tetrachlorobiphenyl	1	0.3	0.1	0.1	0.1	0.1	0.1	0.1	0.1	480	29
2,4,4'-trichlorobiphenyl	1	1	3.1	2.5	0.6	0.6	0.5	0.8	4.2	700	32
22',33',44'-hexachlorobiphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	33	0.2
22',44',55'-hexachlorobiphenyl	1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	110	12
22',44',66'-hexachlorobiphenyl	1.2	1.3	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	3
22'33'44'55'6-nona(Cl)biphenyl	0.6	0.4	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	1.6
22'33'44'55'-octa(Cl)biphenyl	0.2	0.2	0.2	0.2	0.2	1.4	0.2	0.2	0.2	19	0.5
22'33'44'5-heptachlorobiphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.8	0.2	37	2.9
22'33'44'6-heptachlorobiphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	15	1.3
22'33'455'66'-nona(Cl)biphenyl	0.8	0.4	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
22'33'455'6'-octa(Cl)biphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	14	3.2
22'33'45'66'-octa(Cl)biphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	2.3
22'33'4'56-heptachlorobiphenyl	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	11	2.7
22'33'55'66'-octa(Cl)biphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.8
22'33'55'6-heptachlorobiphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
22'344'55'-heptachlorobiphenyl	0.2	0.5	0.2	0.2	0.2	1.5	0.2	0.7	0.2	61	6.9
22'344'5'6-heptachlorobiphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	14	0.2
22'34'55'6-heptachlorobiphenyl	0.4	0.8	0.1	0.1	0.1	2.2	0.1	2	0.1	28	1.9
22'34'566'-heptachlorobiphenyl	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	13	0.1
2'3,4,4',5-pentachlorobiphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	100	8.2
23',44',55'-hexachlorobiphenyl	.2w	.2w	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
23',44',5'6-hexachlorobiphenyl	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	230	7.9
233'44'55'6-octachlorobiphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
233'44'55'-heptachlorobiphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
233'44'5'6-heptachlorobiphenyl	1	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
3,3',4,4'-tetrachlorobiphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
3,3'4,4',5-pentachlorobiphenyl	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
3,3'4,4'55'-hexachlorobiphenyl	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
3,4,4',5-tetrachlorobiphenyl	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5
3,4,4'-trichlorobiphenyl	1	1	1	1	1	1	1	1	1	1100	6.4
PCB congeners; total	16	12	4	2.5	5.5	13	2.6	11	25.8	13000	600

Table C1. Continued.

CHEMICAL	LC08-1	LC08-2	LC08-3	LC10*	LC12	LC12	LC14	LC14*	LC16*	LC16	LC17	LC17
2,2',3,5'-tetrachlorobiphenyl	410	62	200	190	5	660	4.2	45	26	120	38	100
2,2',4,5'-tetrachlorobiphenyl	360	520	550	170	86	540	56	220	93	120	34	92
2,2',5,5'-tetrachlorobiphenyl	490	720	710	250	110	750	75	310	130	160	47	130
2,2',5-trichlorobiphenyl	430	80	250	82	3	560	2	63	27	62	13	42
2,2',6,6'-tetrachlorobiphenyl	3.2	1.9	110	16	4.7	77	3	38	11	11	4.8	6.3
2,2',6-trichlorobiphenyl	2	2	2	2	2	2	2	2	2	2	2	2
2,2'3,4,5'-pentachlorobiphenyl	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
2,2'3,44'5'-hexachlorobiphenyl	53	70	120	47	21	88	10	30	15	28	7.9	26
2,2'3,4'5'6-hexachlorobiphenyl	3.7	8.1	8.8	37	18	74	11	2	11	21	5.3	18
2,2'3,5,5'6-hexachlorobiphenyl	11	16	26	8	2.2	18	1.6	5.7	0.3	5.2	0.3	4.9
2,2'3,5',6-pentachlorobiphenyl	730	980	1100	390	190	1000	110	420	190	250	8.6	200
2,2'4,4',5-pentachlorobiphenyl	90	160	150	77	38	150	21	59	28	48	12	39
2,2'4,5,5'-pentachlorobiphenyl	160	220	12	120	62	220	37	100	49	74	24	62
2,2'4,6,6'-pentachlorobiphenyl	0.1	0.1	0.1	0.1	1.8	0.1	2.2	2.5	2.1	0.1	2.6	0.1
2,3,3'4,4'5-hexachlorobiphenyl	4.4	7.4	8.8	3.4	1.3	8.4	0.2	2.6	0.6	2.5	1.8	2.4
2,3,3'4,4'6-hexachlorobiphenyl	4.2	6.8	14	3.7	1.3	10	0.4	2.3	0.6	3.6	1.6	3.1
2,3,3'4,4'-pentachlorobiphenyl	18	2	21	9.9	7.6	74	4.9	14	5.8	5.7	4.4	28
2,3,3'4',6-pentachlorobiphenyl	150	7.9	84	110	20	260	12	37	25	60	25	73
2,3,3'44'5'-hexachlorobiphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
2,3,4,4',5-pentachlorobiphenyl	4.4	7.7	7.7	2.7	1.4	9.7	0.63	2.2	1.2	2.5	0.51	2.2
2,3',4',5-tetrachlorobiphenyl	470	410	750	230	110	810	67	300	130	160	47	130
2',3,4-trichlorobiphenyl	260	110	120	57	9.9	350	6.5	36	18	49	13	35
2,3,4'-trichlorobiphenyl	1.1	2.4	2	7.6	2	37	1.7	8.7	0.5	0.5	0.5	9.5
2,3'4,4',5-pentachlorobiphenyl	37	51	210	86	48	210	27	80	37	62	19	53
2,3'4,4',6-pentachlorobiphenyl	1.9	6.5	1.4	0.8	0.1	6.6	0.1	0.55	2.6	0.1	3.1	0.1
2,4,4',5-tetrachlorobiphenyl	220	330	360	98	49	350	32	140	60	80	21	60
2,4,4'-trichlorobiphenyl	500	590	690	130	59	690	42	270	92	110	27	76
22',33',44'-hexachlorobiphenyl	1.9	0.2	0.2	1.3	0.2	12	0.2	0.2	0.2	1.4	0.2	4.4
22',44',55'-hexachlorobiphenyl	39	70	92	32	14	53	8.6	25	11	17	5.3	16
22',44',66'-hexachlorobiphenyl	12	19	20	10	4.5	0.1	8	9.5	4.2	0.1	2	0.1
22'33'44'55'6-nona(Cl)biphenyl	0.5	1.9	0.65	0.2	2.4	1.8	1.7	3.9	1.5	0.2	1.4	0.2
22'33'44'55'-octa(Cl)biphenyl	4.9	6.2	9.9	3.5	0.89	7.6	0.5	1.7	2.7	1.7	2.2	2
22'33'44'5-heptachlorobiphenyl	10	14	24	8.2	3	19	0.72	6.3	1.2	5.3	0.8	5.6
22'33'44'6-heptachlorobiphenyl	1.9	3.4	3.9	3.9	2	3.7	1.2	0.23	1.1	0.2	0.9	0.2
22'33'455'66'-nona(Cl)biphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
22'33'455'6'-octa(Cl)biphenyl	4.1	5.8	9.1	2.9	0.52	7.1	0.2	1.1	2.5	0.93	1.9	1.1
22'33'45'66'-octa(Cl)biphenyl	14	20	22	9.9	4	4.6	1.1	7.3	1.1	0.2	3.7	2
22'33'4'56-heptachlorobiphenyl	5.1	7.2	8.8	3.4	0.4	8.8	0.5	1.3	2.3	1.5	1.8	1.5
22'33'55'66'-octa(Cl)biphenyl	1.5	1.4	1.8	1	0.2	0.2	0.2	1.1	0.2	0.2	0.2	0.2
22'33'55'6-heptachlorobiphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
22'344'55'-heptachlorobiphenyl	26	34	62	20	9.4	36	3.7	13	4.5	10	1.9	11
22'344'5'6-heptachlorobiphenyl	4.8	6.7	15	3.3	2.4	9.7	0.35	1.8	0.8	2.9	2.7	2.4
22'34'55'6-heptachlorobiphenyl	13	17	20	8.6	3.8	18	0.78	6.2	1.2	4.8	3.7	5
22'34'566'-heptachlorobiphenyl	5.3	8.2	8.8	4.1	1.1	11	0.3	2.3	3.1	2.9	2.5	2.4
2'3,4,4',5-pentachlorobiphenyl	13	22	25	8.3	2.4	24	4.9	7.3	7.2	5	1.8	4.3
23',44',55'-hexachlorobiphenyl	0.2	4	0.6	3.6	0.4	2.3	0.2	0.6	1	0.2	1.2	0.2
23',44',5'6-hexachlorobiphenyl	37	54	59	30	14	53	6.5	23	7.1	16	2.5	16
233'44'55'6-octachlorobiphenyl	0.2	0.2	1	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
233'44'55'-heptachlorobiphenyl	0.2	1.2	1	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
233'44'5'6-heptachlorobiphenyl	0.6	0.5	1.1	0.6	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
3,3',4,4'-tetrachlorobiphenyl	2.7	0.2	0.5	3	1.4	1.7	4.7	2.6	6.4	0.2	2	0.2
3,3'4,4',5-pentachlorobiphenyl	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
3,3'4,4'55'-hexachlorobiphenyl	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
3,4,4',5-tetrachlorobiphenyl	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5
3,4,4'-trichlorobiphenyl	470	15	210	200	4.7	170	4.5	16	10	9.1	2.1	37
PCB congeners; total	5200	4700	6100	2500	910	7500	570	2300	1000	1500	380	1300

Table C1. Continued.

CHEMICAL	LC18-1	LC18-2	LC19	LC19*	LC22-1	LC22-2	LC23*	LC29*	LC29-1	LC29-2	LC29-3	LC38	LC38*	LC38
2,2',3,5'-tetrachlorobiphenyl	7.7	33	30	41	8.1	1.4	2	11	25	3.7	21	1.5	0.1	1.3
2,2',4,5'-tetrachlorobiphenyl	41	41	39	53	19	19	77	15	26	35	29	2.3	0.1	0.11
2,2',5,5'-tetrachlorobiphenyl	59	58	54	75	26	27	120	19	34	49	38	2.4	2.4	0.74
2,2',5-trichlorobiphenyl	2.9	14	14	34	3.9	3	2	3.3	15	8.8	15	2	2	6.4
2,2',6,6'-tetrachlorobiphenyl	1.1	1.3	0.7	4	5.2	4	6.2	3.2	2.7	1.1	2.9	0.2	0.2	0.2
2,2',6-trichlorobiphenyl	2	2	2	2	2	2	2	2	2	2	2	2	2	2
2,2'3,4,5'-pentachlorobiphenyl	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
2,2'3,44'5'-hexachlorobiphenyl	8.3	9.2	56	53	11	9	22	6.3	13	14	15	1.3	0.9	0.2
2,2'3,4'5'6-hexachlorobiphenyl	7.4	7.8	63	56	9.7	10	28	6.1	10	14	12	1.4	1.8	0.2
2,2'3,5,5'6-hexachlorobiphenyl	0.1	0.1	0.1	17	0.36	0.52	6.3	0.2	2.9	4.3	3.4	0.1	0.1	0.1
2,2'3,5',6-pentachlorobiphenyl	80	98	99	120	48	37	140	37	49	65	55	1.9	2.2	0.84
2,2'4,4',5-pentachlorobiphenyl	16	16	16	19	6.9	7.1	27	5	10	14	11	0.1	0.1	0.1
2,2'4,5,5'-pentachlorobiphenyl	29	28	40	43	13	15	51	11	20	24	20	0.4	0.2	0.1
2,2'4,6,6'-pentachlorobiphenyl	3	2.1	2.9	2.6	4.2	3.2	1.9	3.3	0.1	0.1	0.1	0.1	0.1	0.1
2,3,3'4,4'5-hexachlorobiphenyl	1.9	0.2	2.9	2.5	1.5	1.5	0.6	1.4	1.2	0.77	1.3	0.2	0.2	0.2
2,3,3'4,4'6-hexachlorobiphenyl	1.7	0.3	2.9	3	1.8	1.6	1.1	0.8	1.6	2.1	1.6	0.1	0.1	0.1
2,3,3'4,4'-pentachlorobiphenyl	5	0.45	9.4	7.5	5.8	7.6	1.5	4.7	11	2.4	4.3	0.1	0.1	0.1
2,3,3'4',6-pentachlorobiphenyl	13	16	19	18	22	21	4	16	29	0.1	20	0.8	0.1	0.1
2,3,3'44'5'-hexachlorobiphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
2,3,4,4',5-pentachlorobiphenyl	0.32	0.49	0.9	0.7	0.1	0.1	0.62	0.6	1.6	0.1	1.7	0.1	0.1	0.1
2,3',4',5-tetrachlorobiphenyl	52	54	48	67	24	23	99	18	28	38	30	0.8	1.3	0.51
2',3,4-trichlorobiphenyl	6.2	14	13	21	4.1	0.5	9.4	3.7	5	6.9	0.69	4	1.6	4.1
2,3,4'-trichlorobiphenyl	0.5	1	1	0.5	0.5	2	1.4	1.8	5.3	0.5	0.5	0.5	0.5	0.5
2,3'4,4',5-pentachlorobiphenyl	20	21	22	24	10	9.5	33	6.4	13	17	15	1.4	1.2	0.1
2,3'4,4',6-pentachlorobiphenyl	2.1	1.9	2.4	0.4	0.8	0.1	1.6	1.4	0.18	0.1	0.1	0.1	0.1	0.1
2,4,4',5-tetrachlorobiphenyl	25	26	23	31	11	11	52	8.1	14	20	16	0.1	0.1	0.1
2,4,4'-trichlorobiphenyl	30	38	31	49	18	13	55	12	23	33	26	0.65	0.5	6.2
22',33',44'-hexachlorobiphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	2.5	0.2	0.5	0.2	0.2	0.2
22',44',55'-hexachlorobiphenyl	7.2	6.7	61	55	7.7	11	26	6.1	14	13	16	0.9	1.1	0.1
22',44',66'-hexachlorobiphenyl	3.2	2.2	2.2	2.9	1.2	1.7	3.6	1.3	0.1	0.1	0.1	0.1	0.1	0.1
22'33'44'55'6-nona(Cl)biphenyl	2.2	1.3	0.7	0.3	1.7	0.2	2.5	2	0.2	0.2	0.21	1.1	0.9	0.2
22'33'44'55'-octa(Cl)biphenyl	3.7	0.3	6.4	8	0.2	0.5	2.4	4.5	2.2	1.7	2.7	0.2	0.2	0.2
22'33'44'5-heptachlorobiphenyl	0.46	0.69	20	21	2.7	3.3	7.9	1	5	5.5	5.3	0.8	0.5	0.2
22'33'44'6-heptachlorobiphenyl	1.4	0.8	6.8	5.8	2.1	2.9	0.9	1.9	0.46	0.2	0.73	0.2	0.2	0.2
22'33'455'66'-nona(Cl)biphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
22'33'455'6'-octa(Cl)biphenyl	3.3	2.1	6.6	7.3	3.7	0.2	1.5	1.9	1.3	0.81	1.9	0.2	0.2	0.2
22'33'45'66'-octa(Cl)biphenyl	0.4	0.2	0.3	0.2	3.1	3.7	10	1.5	1.3	2.8	1.5	1.2	0.8	0.2
22'33'4'56-heptachlorobiphenyl	2.1	1.9	16	14	0.29	4.2	2	3.8	1.6	1.7	1.9	0.8	0.1	0.1
22'33'55'66'-octa(Cl)biphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
22'33'55'6-heptachlorobiphenyl	0.2	0.2	0.2	0.2	0.2	2.6	2.7	2.5	0.2	0.2	0.2	0.2	0.2	0.2
22'344'55'-heptachlorobiphenyl	3.1	2.6	47	49	8.2	7.7	18	3.6	10	12	11	0.8	0.8	0.2
22'344'5'6-heptachlorobiphenyl	3.6	2.7	11	11	0.92	0.94	3.7	0.2	2.2	2.8	2.5	0.2	0.2	0.2
22'34'55'6-heptachlorobiphenyl	0.59	0.3	26	27	3.1	3.1	9	0.8	4.8	6	5.5	0.1	0.1	0.1
22'34'566'-heptachlorobiphenyl	2.9	0.5	7.3	6.6	3.3	3.6	2.4	2	0.1	0.1	0.1	0.1	0.1	0.1
2'3,4,4',5-pentachlorobiphenyl	3.3	2.9	4.3	5.7	5.1	5.3	20	1.5	0.2	0.2	0.2	0.2	0.2	0.2
23',44',55'-hexachlorobiphenyl	1	0.7	1.3	1.2	1.1	0.9	1.3	0.7	0.2	0.2	0.2	0.2	0.2	0.2
23',44',5'6-hexachlorobiphenyl	4.7	4.1	9.3	7.6	7.7	8.4	24	3.9	11	2.4	4.3	0.1	0.1	0.1
233'44'55'6-octachlorobiphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
233'44'55'-heptachlorobiphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
233'44'5'6-heptachlorobiphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.4	0.2	0.2	0.2	0.2	0.2	0.2	0.2
3,3',4,4'-tetrachlorobiphenyl	6	3.7	23	25	5.5	7	9.6	4.9	0.2	0.2	0.2	0.2	0.2	0.2
3,3'4,4',5-pentachlorobiphenyl	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
3,3'4,4'55'-hexachlorobiphenyl	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
3,4,4',5-tetrachlorobiphenyl	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5
3,4,4'-trichlorobiphenyl	5.7	30	20	36	5.8	1	1	4.7	43	9.6	31	1	1	1
PCB congeners; total	450	540	850	1000	290	270	870	230	450	410	430	24	16	21

Table C2. Polycyclic aromatic hydrocarbons and organo-chlorine pesticides in Lyons Creek (LC) and reference creek (BEC, BLC, UC, TC) sediment.

CHEMICAL	BEC01	BEC02	BLC01	BLC02	UC01	UC01*	TC40	TC40*	LC01	LC03	LC06*
a-BHC (hexachlorocyclohexane)	1	1	1	1	1	1	1	1	1	1	1
Acenaphthene	20	20	20	20	20	20	20	20	20	400	20
Acenaphthylene	20	20	20	20	20	20	20	20	20	400	20
a-Chlordane	2	2	2	2	2	2	2	2	2	2	2
Aldrin	1	1	1	1	1	1	1	1	1	1	1
Anthracene	20	20	20	20	20	20	20	20	20	400	20
b-BHC (hexachlorocyclohexane)	1	1	1	1	1	1	1	1	1	1	1
Benzo(a)anthracene	20	20	80	20	20	20	20	20	40	5000	120
Benzo(a)pyrene	40	40	40	40	40	40	40	40	40	2900	120
Benzo(b)fluoranthene	20	20	120	40	20	20	20	20	80	3800	200
Benzo(g,h,i)perylene	40	40	40	40	40	40	40	40	40	3100	120
Benzo(k)fluoranthene	20	20	40	20	20	20	20	20	20	1300	60
Chrysene	20	20	80	40	20	20	20	20	60	9900	180
d10-phenanthrene	140	140	140	120	91	86	90	100	100	140	86
d12-chrysene	75	69	98	78	65	62	68	67	47	140	47
d8-naphthalene	76	81	100	120	50	52	59	58	63	67	53
DDT & Metabolites	2	2	2	2	2	2	2	2	2	340	32
Dibenzo(a,h)anthracene	40	40	40	40	40	40	40	40	40	840	40
Dieldrin	2	2	2	2	2	2	2	2	2	2	2
Endosulphan I	2	2	2	2	2	2	2	2	2	2	2
Endosulphan II	4	4	4	4	4	4	4	4	4	12	4
Endosulphan sulphate	4	4	4	4	4	4	4	4	4	16	4
Endrin	4	4	4	4	4	4	4	4	4	24	4
Fluoranthene	20	20	220	80	40	40	20	20	120	13000	240
Fluorene	20	20	20	20	20	20	20	20	20	1100	20
g-BHC (hexachlorocyclohexane)	1	1	1	1	1	1	1	1	1	3	1
g-Chlordane	2	2	2	2	2	2	2	2	2	2	2
Heptachlor	1	1	1	1	1	1	1	1	1	1	1
Heptachlor epoxide	1	1	1	1	1	1	1	1	1	2	1
Indeno(1,2,3-c,d)pyrene	40	40	40	40	40	40	40	40	40	1200	160
Methoxychlor	5	5	5	5	5	5	5	5	5	5	5
Mirex	5	5	5	5	5	5	5	5	5	5	5
Naphthalene	20	20	40	40	20	20	20	20	20	400	20
op-DDT	5	5	5	5	5	5	5	5	5	5	5
Oxychlordane	2	2	2	2	2	2	2	2	2	2	2
Phenanthrene	20	20	100	40	20	20	20	20	80	1200	120
pp-DDD	5	5	5	5	5	5	5	5	5	5	5
pp-DDE	1	1	1	1	1	2	1	2	2	340	32
pp-DDT	5	5	5	5	5	5	5	5	5	5	5
Pyrene	20	20	160	60	20	20	20	20	100	18000	260
Toxaphene	50	50	50	50	50	50	50	50	50	50	50

Table C2. Continued.

CHEMICAL	LC08-1	LC08-2	LC08-3	LC10*	LC12	LC12	LC14	LC14*	LC16*	LC16	LC17	LC17
a-BHC (hexachlorocyclohexane)	1	1	1	1	1	1	1	1	1	1	1	1
Acenaphthene	60	60	60	20	20	40	20	20	20	20	20	20
Acenaphthylene	20	20	20	20	20	20	20	20	20	20	20	20
a-Chlordane	2	2	2	2	2	2	2	2	2	2	2	2
Aldrin	1	1	1	1	1	1	1	1	1	1	1	1
Anthracene	60	160	140	40	20	60	20	20	20	20	20	20
b-BHC (hexachlorocyclohexane)	1	1	1	1	1	1	1	1	1	1	1	1
Benzo(a)anthracene	300	420	460	240	80	220	40	100	80	60	60	60
Benzo(a)pyrene	240	320	360	280	120	200	80	80	80	80	80	80
Benzo(b)fluoranthene	460	560	600	420	220	380	140	200	120	120	100	120
Benzo(g,h,i)perylene	200	280	320	320	120	240	80	80	80	80	80	80
Benzo(k)fluoranthene	160	180	200	120	60	100	20	60	40	40	40	40
Chrysene	640	780	860	480	180	400	120	280	180	120	100	100
d10-phenanthrene	96	86	97	89	85	78	89	91	93	72	85	93
d12-chrysene	67	57	61	53	62	37	57	61	62	36	58	53
d8-naphthalene	48	46	42	42	42	33	82	85	83	23	95	49
DDT & Metabolites	94	110	120	74	40	140	22	58	32	50	18	44
Dibenzo(a,h)anthracene	40	40	40	80	40	40	40	40	40	40	40	40
Dieldrin	2	2	2	2	2	32	2	2	2	8	2	4
Endosulphan I	2	2	2	2	2	2	2	2	2	2	2	2
Endosulphan II	4	4	4	4	4	4	4	4	4	4	4	4
Endosulphan sulphate	4	4	4	4	4	4	4	4	4	4	4	4
Endrin	4	4	4	4	4	4	4	4	4	4	4	4
Fluoranthene	860	1300	1400	480	200	680	140	340	160	120	100	120
Fluorene	80	120	140	20	20	60	20	40	20	20	20	20
g-BHC (hexachlorocyclohexane)	1	1	1	1	1	1	1	1	1	1	1	1
g-Chlordane	2	2	2	2	2	2	2	2	2	2	2	2
Heptachlor	1	1	1	1	1	1	1	1	1	1	1	1
Heptachlor epoxide	1	1	1	1	1	1	1	1	1	1	1	1
Indeno(1,2,3-c,d)pyrene	320	400	440	360	200	240	120	120	120	80	80	80
Methoxychlor	5	5	5	5	5	5	5	5	5	5	5	5
Mirex	5	5	5	5	5	5	5	5	5	5	5	5
Naphthalene	20	20	20	20	20	20	20	20	20	20	20	20
op-DDT	5	5	5	5	5	5	5	5	5	5	5	5
Oxychlordane	2	2	2	2	2	2	2	2	2	2	2	2
Phenanthrene	360	660	740	180	80	260	60	60	60	40	60	60
pp-DDD	5	5	5	5	5	5	5	5	5	5	5	5
pp-DDE	94	110	120	74	40	140	22	58	32	49	18	43
pp-DDT	5	5	5	5	5	5	5	5	5	5	5	5
Pyrene	760	1100	1200	600	200	680	120	340	180	120	100	120
Toxaphene	50	50	50	50	50	50	50	50	50	50	50	50

Table C2. Continued.

CHEMICAL	LC18-1	LC18-2	LC19	LC19*	LC22-1	LC22-2	LC23*	LC29*	LC29-1	LC29-2	LC29-3	LC38	LC38*	LC38
a-BHC (hexachlorocyclohexane)	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Acenaphthene	20	20	20	20	20	20	20	20	20	20	20	20	20	20
Acenaphthylene	20	20	20	20	20	20	20	20	20	20	20	20	20	20
a-Chlordane	2	2	2	2	2	2	2	2	2	2	2	2	2	2
Aldrin	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Anthracene	20	20	20	20	20	20	20	20	20	20	20	20	20	20
b-BHC (hexachlorocyclohexane)	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Benzo(a)anthracene	100	60	60	80	40	40	120	20	40	20	40	20	20	20
Benzo(a)pyrene	80	80	80	80	40	40	160	40	40	40	40	40	40	40
Benzo(b)fluoranthene	140	120	120	120	80	80	180	60	120	80	120	40	40	20
Benzo(g,h,i)perylene	120	80	80	120	40	40	160	40	80	80	80	40	40	40
Benzo(k)fluoranthene	40	40	40	20	20	20	20	20	40	20	40	20	20	40
Chrysene	180	120	120	160	80	80	260	60	100	60	100	20	40	20
d10-phenanthrene	87	91	87	88	89	85	97	89	110	110	120	84	92	95
d12-chrysene	61	65	72	62	62	67	64	56	84	58	85	67	67	54
d8-naphthalene	85	44	47	39	48	49	40	48	90	58	91	53	48	55
DDT & Metabolites	16	20	18	26	10	8	20	8	2	14	2	2	2	4
Dibenzo(a,h)anthracene	40	40	40	40	40	40	40	40	40	40	40	40	40	40
Dieldrin	2	2	2	2	2	2	2	2	2	4	2	2	2	2
Endosulphan I	2	2	2	2	2	2	2	2	2	2	2	2	2	2
Endosulphan II	4	4	4	4	4	4	4	4	4	4	4	4	4	4
Endosulphan sulphate	4	4	4	4	4	4	4	4	4	4	4	4	4	4
Endrin	4	4	4	4	4	4	4	4	4	4	4	4	4	4
Fluoranthene	160	120	100	120	60	60	100	60	80	60	80	40	40	40
Fluorene	20	20	20	20	20	20	20	20	20	20	20	20	20	20
g-BHC (hexachlorocyclohexane)	1	1	1	1	1	1	1	1	1	1	1	1	1	1
g-Chlordane	2	2	2	2	2	2	2	2	2	2	2	2	2	2
Heptachlor	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Heptachlor epoxide	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Indeno(1,2,3-c,d)pyrene	120	120	120	80	80	80	80	40	80	40	80	40	40	40
Methoxychlor	5	5	5	5	5	5	5	5	5	5	5	5	5	5
Mirex	5	5	5	5	5	5	5	5	5	5	5	5	5	5
Naphthalene	20	20	20	20	20	20	20	20	40	20	40	20	20	20
op-DDT	5	5	5	5	5	5	5	5	5	5	5	5	5	5
Oxychlordane	2	2	2	2	2	2	2	2	2	2	2	2	2	2
Phenanthrene	60	40	40	40	20	20	40	20	40	20	40	20	20	20
pp-DDD	5	5	5	5	5	5	5	5	5	5	5	5	5	5
pp-DDE	16	20	18	26	10	8	20	8	17	13	14	1	1	3
pp-DDT	5	5	5	5	5	5	5	5	5	5	5	5	5	5
Pyrene	160	120	120	140	60	60	160	60	80	60	80	20	40	40
Toxaphene	50	50	50	50	50	50	50	50	50	50	50	50	50	50

Table C3. Measured environmental variables in overlying water. Values in mg/L unless otherwise noted.

Site	Alkalinity	Conductivity (µS/cm)	Dissolved Oxygen	NH_3	NO_3/NO_2	pH	Total Nitrogen	Total Phosphorus	Temp. (°C)
BEC01	136	670	7.1	0.02	0.03	8.3	0.80	0.08	10.9
BEC02	136	630	9.2	0.03	0.01	8.3	0.82	0.08	11.6
BLC01	107	710	8.3	0.06	2.00	8.3	0.80	0.19	10.9
BLC02	109	680	8.3	0.06	1.96	8.3	0.86	0.12	12.7
UC01	95	300	9.3	0.04	0.30	7.5	0.40	0.02	11.9
TC40	120	450	5.5	0.10	0.19	7.0	1.51	0.18	10.8
LC01	92	310	9.7	0.05	0.28	8.8	0.34	0.02	16.8
LC03	91	270	11.3	0.05	0.27	8.0	0.37	0.01	17.2
LC06	93	240	13.1	0.03	0.18	7.5	0.23	0.01	15.9
LC08	97	250	10.2	0.02	0.19	7.5	0.25	0.01	14.2
LC10	92	240	9.5	0.03	0.21	7.2	0.23	0.01	16.0
LC12	91	280	11.4	0.05	0.26	8.7	0.31	0.02	18.2
LC14	96	260	11.6	0.04	0.12	7.2	0.30	0.02	9.7
LC16	92	290	10.4	0.04	0.11	8.5	0.27	0.02	12.9
LC17	93	300	10.4	0.05	0.09	8.7	0.40	0.02	12.7
LC18	109	350	12.0	0.09	0.04	9.4	0.38	0.02	11.9
LC19	94	250	13.3	0.05	0.01	7.5	0.30	0.02	11.2
LC22	97	280	13.0	0.03	0.01	9.0	0.29	0.02	9.6
LC23	97	290	11.2	0.05	0.03	7.0	0.33	0.02	8.5
LC29[a]	93	310	10.0	0.03	0.01	8.5	0.29	0.03	8.3
LC38	96	360	10.6	0.05	0.02	8.6	0.55	0.04	9.3

[a]QA/QC site (value represents the mean of three replicates).

Table C4. Physical characteristics of Lyons Creek (LC) and reference creek (BEC, BLC, UC, TC) sediment (top 10 cm).

Site	% Sand	% Silt	% Clay	% Gravel
BEC01	5.0	23.9	71.1	0.0
BEC02	15.9	32.6	51.5	0.0
BLC01	16.6	36.9	46.5	0.0
BLC02	11.3	20.3	61.5	6.8
UC01	18.0	43.6	38.4	0.0
TC40	14.7	37.4	47.0	0.8
LC01	5.4	52.5	42.1	0.0
LC03	17.9	54.0	28.0	0.1
LC06	2.6	65.6	31.9	0.0
LC08	0.8	55.8	43.4	0.0
LC10	2.5	58.7	38.9	0.0
LC12	0.8	59.1	40.2	0.0
LC14	0.3	83.1	16.6	0.0
LC16	0.8	54.0	45.2	0.0
LC17	1.0	51.8	47.2	0.0
LC18	7.0	46.8	46.3	0.0
LC19	1.7	43.7	54.6	0.0
LC22	0.4	44.3	55.3	0.0
LC23	2.9	33.5	63.6	0.0
LC29[a]	1.7	47.1	51.3	0.0
LC38	4.4	47.3	48.3	0.0

[a]QA/QC site (value represents the mean of three replicates)

Table C5. Nutrients and trace metal concentrations in Lyons Creek (LC) and reference creek (BEC, BLC, UC, TC) sediment (top 10 cm). Values in µg/g dry weight unless otherwise noted. Values exceeding the SEL, where available, are highlighted.

Site	Total N	Total Organic C (%)	Total P	LOI (%)	Al_2O_3 (%)	As	Ca	Cd	Co	Cr	Cu	Fe (%)
BEC01	3490	3.2	428	12.8	16.9	5.1	0.84	<1	8.6	27.0	29.6	1.7
BEC02	**8420**	**10.6**	763	27.2	12.5	7.9	1.42	0.7	14.7	33.3	30.0	2.7
BLC01	4050	5.6	630	14.6	12.5	<5	0.79	<1	14.1	30.7	24.6	2.4
BLC02	1970	4.0	875	16.6	12.8	<5	5.42	<1	14.8	38.2	23.5	2.8
UC01	4460	6.3	1040	16.6	12.4	4.0	1.78	0.8	16.0	28.0	22.0	2.7
TC40	**5230**	6.9	794	19.6	11.3	3.0	3.58	1.2	16.0	27.0	28.0	2.5
LC01	2910	1.9	1040	17.9	10.1	<5	9.09	<1	11.4	24.2	39.5	2.1
LC03	**6580**	2.8	**2200**	28.7	10.4	**71.3**	4.81	2.2	18.4	56.1	**131.1**	4.0
LC06	3630	4.8	922	19.3	9.7	6.0	7.05	0.8	13.0	33.0	49.0	2.7
LC08	2480	4.2	1470	17	11.8	8.0	6.36	1.0	15.0	39.0	65.0	3.3
LC10	4310	6.3	1460	26.1	9.8	8.0	6.80	1.1	13.0	34.0	58.0	3.0
LC12	3680	4.8	1460	18.4	11.2	<5	6.53	1.0	12.3	52.4	59.1	3.0
LC14	**5030**	4.5	3070	17.7	12.9	10.0	5.27	1.2	16.0	47.0	65.0	**4.6**
LC16	**8390**	6.6	1020	20.6	13.2	8.7	1.72	<1	12.6	39.8	55.6	3.1
LC17	**5690**	5.2	1100	17.9	12.4	<5	3.14	<1	12.6	45.9	48.0	3.1
LC18	**5480**	6.6	892	32.54	10.4	5.0	2.80	0.7	12.0	36.0	37.0	2.9
LC19	4710	5.1	1710	14.81	13.7	7.0	2.05	0.7	15.0	35.0	44.0	3.7
LC22	**6110**	5	1300	14.5	14.6	7.0	0.69	0.9	16.0	34.0	41.0	3.6
LC23	**4830**	6.3	1460	16.73	13.5	8.0	1.07	0.9	17.0	36.0	42.0	3.8
LC29[a]	**5317**	5.5	1517	16.0	13.2	9.8	0.79	<1	13.1	34.2	35.9	3.2
LC38	**8180**	**10.7**	940	26.9	12.9	8.3	0.79	<1	14.3	30.5	27.3	2.7
LEL	550	1	600	-	-	6.0		0.6	-	26	16	2%
SEL	4800	10	2000	-	-	33.0		10	-	110	110	4%

[a]QA/QC site (value represents the mean of three replicates)

Table C5. Continued.

Site	Hg	K_2O	Mg (%)	Mn	Na_2O (%)	Ni	P_2O_5 (%)	Pb	SiO_2 (%)	Ti	V	Zn
BEC01	0.06	3.1	0.02	196.5	0.72	25.3	0.22	19.1	59.0	137	22.5	81
BEC02	0.04	2.5	0.04	402.1	0.52	35.3	0.24	19.6	45.8	170	17.9	107
BLC01	0.06	2.6	0.02	249.7	1.08	37.1	0.20	25.5	60.4	217	18.9	109
BLC02	0.03	2.8	0.06	623.5	0.73	34.6	0.25	49.2	49.3	255	18.6	81
UC01	0.10	2.5	0.97	444.0	1.05	36.0	0.17	22.0	55.9	173	33.0	112
TC40	0.08	2.3	1.22	352.0	1.00	32.0	0.17	18.0	51.6	196	30.0	166
LC01	0.06	2.3	0.06	562.8	0.85	26.8	0.22	23.7	49.3	238	14.4	126
LC03	0.15	2.1	0.03	349.4	0.49	**147.1**	0.44	117.2	39.6	190	18.7	**7969**
LC06	0.08	2.1	1.57	494.0	1.03	36.0	0.20	30.0	46.5	236	22.0	444
LC08	0.11	2.6	1.50	493.0	0.96	51.0	0.34	68.0	49.3	254	29.0	**1080**
LC10	0.09	2.3	1.22	396.0	0.90	43.0	0.35	45.0	43.2	201	22.0	**841**
LC12	0.10	2.5	0.04	414.2	0.73	50.0	0.37	64.2	48.7	227	15.8	**926**
LC14	0.14	2.8	1.24	460.0	0.71	59.0	0.58	70.0	44.6	271	36.0	**2440**
LC16	0.07	2.8	0.03	310.9	0.66	50.2	0.30	47.2	50.4	220	20.1	645
LC17	0.09	2.6	0.05	492.2	0.77	43.9	0.36	37.8	51.6	200	17.7	590
LC18	0.06	2.2	1.53	489.0	0.70	46.0	0.20	40.0	42.4	161	28.0	407
LC19	0.09	2.9	1.15	525.0	1.01	46.0	0.38	32.0	55.2	262	38.0	709
LC22	0.10	3.1	0.93	532.0	0.90	58.0	0.26	29.0	55.4	201	37.0	522
LC23	0.11	2.7	1.07	585.0	0.87	54.0	0.31	35.0	53.3	201	39.0	783
LC29[a]	0.06	2.6	0.05	473.6	0.85	49.6	0.41	46.5	57.3	194	19.1	657
LC38	0.05	2.5	0.04	438.6	0.61	46.1	0.31	33.2	48.4	174.4	23.0	172
LEL	0.2	-	-	460	-	16	-	31	-	-	-	120
SEL	2.0	-	-	1100	-	75	-	250	-	-	-	820

[a]QA/QC site (value represents the mean of three replicates)

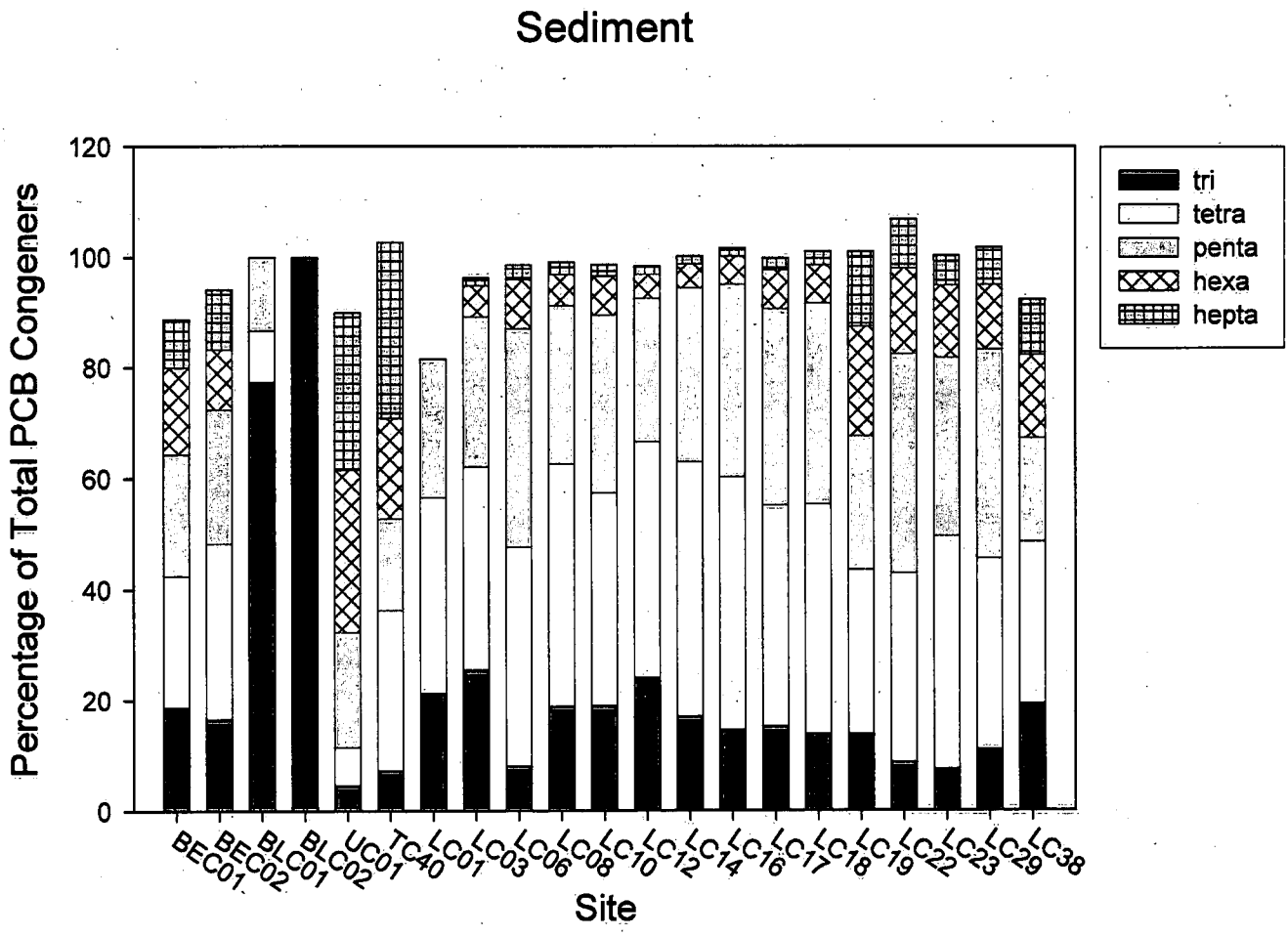


Figure C1. Isomeric composition (%) of Lyons Creek (LC) and reference creek (BEC, BLC, UC, TC) sediment.

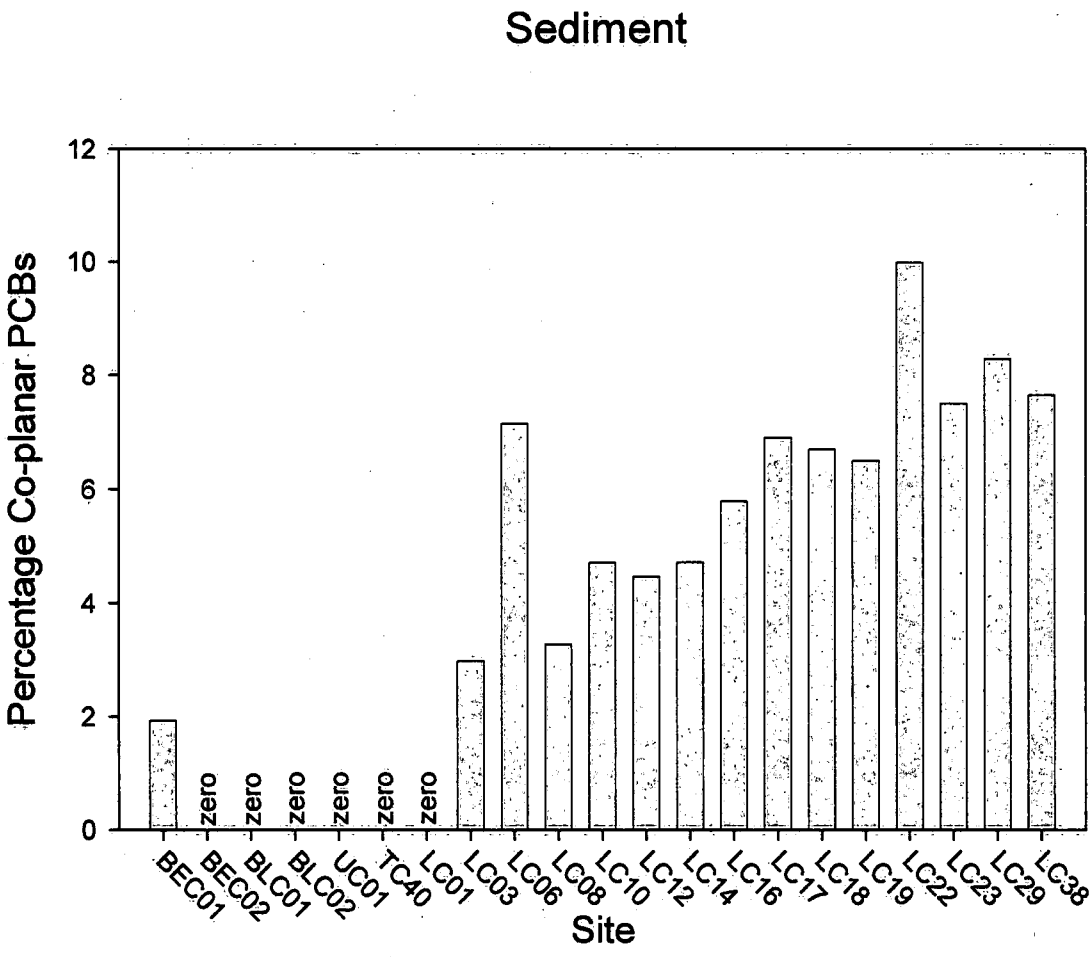


Figure C2. Percentage of co-planar PCBs in Lyons Creek (LC) and reference creek (BEC, BLC, UC, TC) sediment.

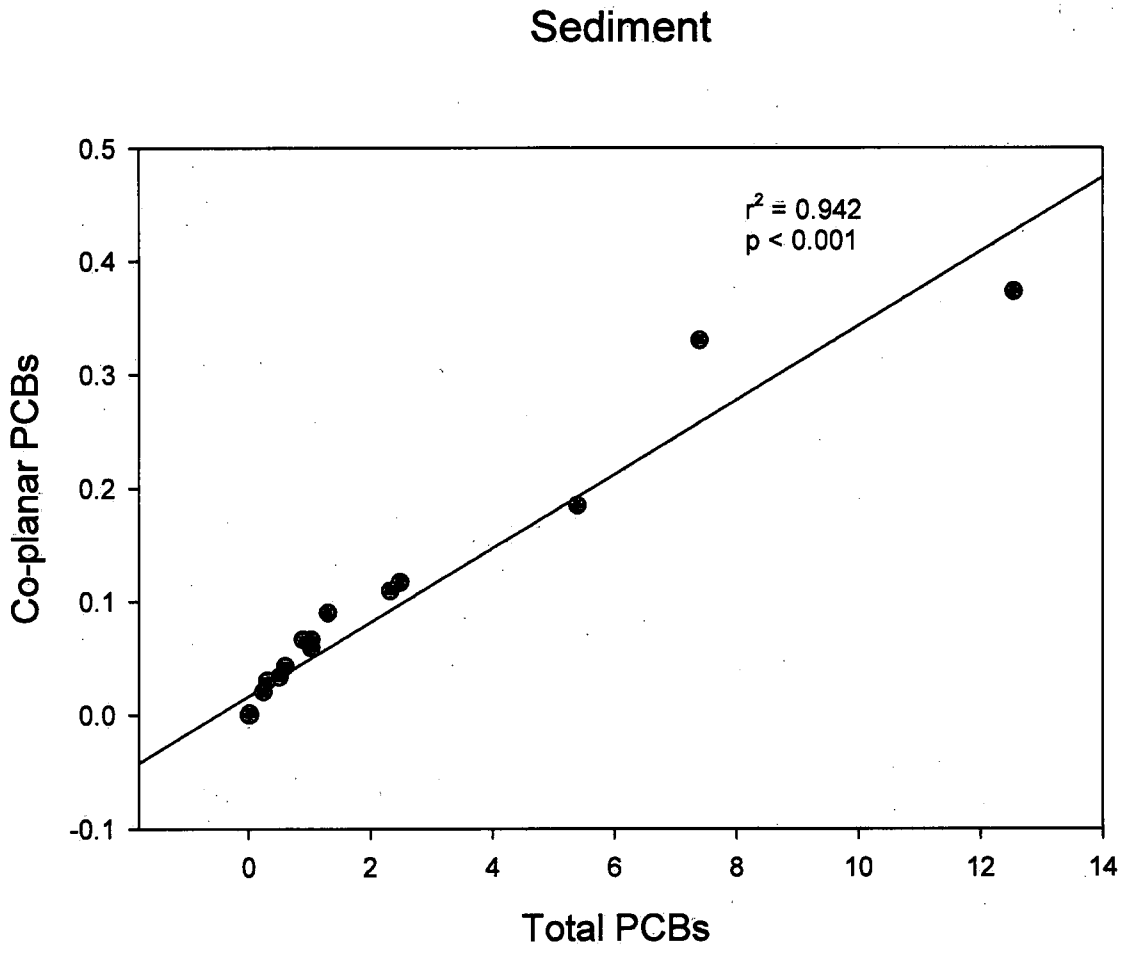


Figure C3. Coplanar PCBs versus total PCBs in sediment.

APPENDIX D. Biota Contaminant Concentrations/Biota-Sediment Accumulation Factors

Table D1. PCB congeners in Lyons Creek (LC) and reference creek (BLC, UC, TC) biota.

Chemical	Units	BLC01 CHIR	BLC01 AMP	BLC01 DAM	BLC02 CHIR	BLC02 AMP	BLC02 DAM	LC01 AMP	LC01 OLIG	LC01 DAM	LC03 AMP	LC03 OLIG	LC03 DAM	LC12 CHIR	LC12 AMP	LC12 OLIG	LC12 DAM	LC17 CHIR	LC17 AMP	LC17 OLIG	LC17 DAM
2,2',3,5'-tetrachlorobiphenyl	ng/g	13	15	21	23	19	38	64	28	26	780	79	1.8	290	980	3800	55	260	390	300	28
2,2',4,5'-tetrachlorobiphenyl	ng/g	0.1	16	12	1.8	10	6.4	50	24	0.1	570	0.1	0.1	190	750	2700	12	260	370	240	14
2,2',5,5'-tetrachlorobiphenyl	ng/g	1	27	60	13	26	12	83	38	6.4	840	4.5	0.1	340	1100	4600	28	450	560	390	26
2,2',5-trichlorobiphenyl	ng/g	42	4	60	98	100	120	200	48	66	1200	2	6	290	620	2300	140	170	250	130	68
2,2',6,6'-tetrachlorobiphenyl	ng/g	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	1.8	24	0.2	0.2
2,2',6-trichlorobiphenyl	ng/g	2	2	2	2	2	2	6	2	2	2	2	2	2	2	2	2	2	2	2	2
2,2'3,4,5'-pentachlorobiphenyl	ng/g	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
2,2'3,44'5'-hexachlorobiphenyl	ng/g	0.2	0.2	0.2	0.2	0.2	0.2	0.2	3.2	0.2	17	0.2	4.2	91	120	1200	0.2	110	63	110	1.2
2,2'3,4'5'6-hexachlorobiphenyl	ng/g	0.2	0.2	0.2	0.2	0.2	0.2	0.2	6.2	0.2	0.2	0.2	0.2	82	150	1300	0.2	31	52	130	0.2
2,2'3,5,5'6-hexachlorobiphenyl	ng/g	0.1	0.1	0.1	0.1	0.1	0.1	0.1	5	0.1	17	0.1	0.1	19	35	320	0.1	17	14	45	0.1
2,2'3,5',6-pentachlorobiphenyl	ng/g	0.1	17	0.1	8.6	13	16	69	49	8.7	1200	40	2.2	730	1600	8100	150	830	770	700	47
2,2'4,4',5-pentachlorobiphenyl	ng/g	0.1	0.9	0.1	0.1	0.1	0.1	0.1	5.8	0.1	74	0.1	0.1	130	300	1800	28	250	180	160	22
2,2'4,5,5'-pentachlorobiphenyl	ng/g	0.1	11	0.1	0.1	1.2	0.1	3.8	20	0.1	230	0.1	0.1	230	440	2700	21	130	190	220	0.4
2,2'4,6,6'-pentachlorobiphenyl	ng/g	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
2,3,3'4,4'5-hexachlorobiphenyl	ng/g	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	8	12	77	6.4	9.6	6.2	8.6	2.8
2,3,3'4,4'6-hexachlorobiphenyl	ng/g	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	17	0.1	0.1	63	38	180	0.1	20	11	15	0.1
2,3,3'4,4'-pentachlorobiphenyl	ng/g	0.1	0.1	0.1	0.1	0.1	12	20	7.7	8.7	100	0.1	0.1	170	240	1400	34	41	43	100	14
2,3,3'4',6-pentachlorobiphenyl	ng/g	0.1	2.6	0.1	0.1	1.7	0.1	7.3	23	0.1	390	0.1	1.4	300	570	3100	3	98	200	260	1.1
2,3,3'44'5'-hexachlorobiphenyl	ng/g	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
2,3,4,4',5-pentachlorobiphenyl	ng/g	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	12	19	93	0.1	11	7.7	7.5	0.1
2,3',4',5-tetrachlorobiphenyl	ng/g	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	25	0.4	540	1200	5200	91	350	450	400	0.1
2',3,4-trichlorobiphenyl	ng/g	21	18	15	22	19	52	58	23	29	590	200	1	130	480	1500	42	51	110	89	15
2,3,4'-trichlorobiphenyl	ng/g	0.5	0.5	0.5	2.5	0.5	0.5	14	0.5	4.5	0.5	0.5	0.5	51	200	320	0.5	0.5	1.5	4.5	0.5
2,3'4,4',5-pentachlorobiphenyl	ng/g	0.1	0.1	28	0.1	0.1	0.1	0.1	4.8	0.1	240	0.1	0.1	260	390	2800	90	270	170	200	15
2,3'4,4',6-pentachlorobiphenyl	ng/g	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	5.6	0.1	0.1	4	4.4	0.1
2,4,4',5-tetrachlorobiphenyl	ng/g	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	33	0.1	300	540	3100	160	340	260	0.1	0.1
2,4,4'-trichlorobiphenyl	ng/g	49	29	35	43	32	84	100	48	53	950	410	4.5	200	630	2300	110	250	330	160	45
22',33',44'-hexachlorobiphenyl	ng/g	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	8.8	1.6	130	6	3.4	2	11	0.2
22',44',55'-hexachlorobiphenyl	ng/g	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	4	0.8	510	0.1	68	35	42	0.1
22',44',66'-hexachlorobiphenyl	ng/g	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
22'33'44'55'6-nona(Cl)biphenyl	ng/g	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
22'33'44'55'-octa(Cl)biphenyl	ng/g	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	9.8	0.2	0.2	0.2	0.6	0.2
22'33'44'5-heptachlorobiphenyl	ng/g	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	6.2	0.2	180	0.2	15	5.8	14	0.2
22'33'44'6-heptachlorobiphenyl	ng/g	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
22'33'455'66'-nona(Cl)biphenyl	ng/g	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
22'33'455'6'-octa(Cl)biphenyl	ng/g	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	10	0.2
22'33'45'66'-octa(Cl)biphenyl	ng/g	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	85	0.2	0.2	0.2	8.8	0.2
22'33'4'56-heptachlorobiphenyl	ng/g	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	1.5	0.5	0.1
22'33'55'66'-octa(Cl)biphenyl	ng/g	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
22'33'55'6-heptachlorobiphenyl	ng/g	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
22'344'55'-heptachlorobiphenyl	ng/g	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	9	0.2	0.2	33	27	440	0.2	34	12	18	0.2
22'344'5'6-heptachlorobiphenyl	ng/g	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	87	0.2	8.8	6.2	4	0.2
22'34'55'6-heptachlorobiphenyl	ng/g	0.1	0.1	0.1	0.1	0.1	0.1	0.1	2.4	0.1	0.1	0.1	0.1	0.1	0.1	530	0.1	10	9.3	93	0.1
22'34'566'-heptachlorobiphenyl	ng/g	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	12	8.9	2.8	0.1
2'3,4,4',5-pentachlorobiphenyl	ng/g	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	13	19	2.8	0.2
23',44',55'-hexachlorobiphenyl	ng/g	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
23',44',5'6-hexachlorobiphenyl	ng/g	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	4	0.8	510	0.1	72	35	42	0.1
233'44'55'6-octachlorobiphenyl	ng/g	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
233'44'55'-heptachlorobiphenyl	ng/g	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
233'44'5'6-heptachlorobiphenyl	ng/g	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
3,3',4,4'-tetrachlorobiphenyl	ng/g	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
3,3'4,4',5-pentachlorobiphenyl	ng/g	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
3,3'4,4'55'-hexachlorobiphenyl	ng/g	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
3,4,4',5-tetrachlorobiphenyl	ng/g	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5
3,4,4'-trichlorobiphenyl	ng/g	1	35	1	1	1	64	1	1	25	8	51	1	140	520	1200	33	210	580	78	1
PCB congeners: total	ng/g	130	160	230	210	220	400	720	340	230	7200	840	21	4600	11000	53000	1000	4400	5200	4000	300

Table D1. Continued.

Chemical	UC01 ODON	UC01 OLIG	UC01 CHIR	UC01 AMP	TC40 AMP	TC40 ODON	TC40 CHIR	TC40 OLIG	LC08 ODON	LC08 AMP	LC08 OIG	LC08 CHIR	LC14 CHIR	LC14 OLIG	LC14 AMP	LC14 ODON	LC16 CHIR	LC16 OLIG
percent lipid	11	8.7	25	4.5	4.5	3.4	75	23	7.1	5.5	17	9.6	7.8	20	7.2	11	15	23
2,2',5-tricholorobiphenyl	12	2	12	11	2	14	44	77	6	60	6	33	38	34	100	16	120	660
2,2',6-trichlorobiphenyl	2	2	2	2	8	2	2	2	8	36	2	2	2	2	2	2	2	2
2,3,4'-trichlorobiphenyl	0.5	13	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	15	0.5	65	0.5	0.5	18
2,4,4'-trichlorobiphenyl	5.5	2	35	12	4	9.5	5	14	9	120	24	65	60	76	170	6	130	270
2',3,4-tricholobiphenyl	5.5	2.5	11	4	3.5	15	26	39	3	34	0.5	1	14	12	36	6.5	30	110
3,4,4'-trichlorobiphenyl	1	1	38	1	1	1	1	1	1	77	1	1	35	22	62	2	33	93
2,2',3,5-tetrachlorobiphenyl	5.5	2.4	28	1	1.8	12	12	0.1	0.1	190	0.1	0.1	4.9	2.8	310	1.7	25	88
2,2',4,5-tetrachlorobiphenyl	8.1	0.4	22	3	1.4	18	23	32	13	240	54	33	46	13	180	0.6	23	14
2,2',5,5'-tetrachlorobiphenyl	13	2.9	20	3	3.4	11	0.1	0.1	7.9	390	44	29	38	21	390	10	25	54
2,2',6,6'-tetrachlorobiphenyl	4.6	1.4	22	5	1.8	6.6	0.2	46	2.4	9.4	31	34	2.8	27	15	5.8	14	99
2,3',4',5-tetrachlorobiphenyl	5.9	3	23	11	3.6	10	5.6	44	9.3	420	66	0.1	130	40	390	2.8	68	200
2,4,4',5-tetrachlorobiphenyl	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
3,3',4,4'-tetrachlorobiphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	5.2	0.2	0.2	0.2	0.2	2.8	0.2	0.2	0.2
3,4,4',5-tetrachlorobiphenyl	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5
2,2'3,4,5'-pentachlorobiphenyl	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
2,2'3,5',6-pentachlorobiphenyl	10	1.7	35	10	3.1	18	0.1	41	42	630	87	150	230	120	550	18	180	130
2,2'4,4',5-pentachlorobiphenyl	0.1	0.1	0.1	3	1	0.1	0.1	0.1	13	160	0.1	55	42	38	130	16	56	45
2,2'4,5,5'-pentachlorobiphenyl	0.1	0.4	0.1	3	0.9	0.1	0.1	0.1	10	240	0.1	11	68	48	170	6.8	45	71
2,2'4,6,6'-pentachlorobiphenyl	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
2,3,3'4,4'-pentachlorobiphenyl	0.1	0.7	0.1	2	0.8	0.1	0.1	0.1	0.7	36	0.1	15	59	7	39	0.9	31	30
2,3,3'4',6-pentachlorobiphenyl	1.7	1.6	0.1	2	1.3	0.1	0.1	0.1	0.1	100	0.1	0.1	57	19	200	23	97	110
2,3,4,4',5-pentachlorobiphenyl	0.1	1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	6.5	0.1	0.1	0.1	0.1	4.5	0.1	0.1	0.1
2,3'4,4',5-pentachlorobiphenyl	0.1	1	1	4	1	0.1	0.1	0.1	13	190	0.1	21	120	23	140	1.5	43	72
2,3'4,4',6-pentachlorobiphenyl	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.4	0.1	0.1	0.1	0.1	1.8	0.1	0.1	0.1
2'3,4,4',5-pentachlorobiphenyl	0.2	1.6	0.2	5	1.4	0.2	0.2	0.2	7	74	0.2	22	42	30	47	9.2	0.2	0.2
3,3'4,4',5-pentachlorobiphenyl	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	5.1	0.1	0.1	17	13	2.1	7.1	40	45
22',33',44'-hexachlorobiphenyl	0.2	0.6	0.2	3	0.6	0.2	0.2	0.2	0.2	12	0.2	0.2	16	8.2	5	2.4	0.2	10
2,2'3,44'5'-hexachlorobiphenyl	0.2	0.2	0.2	5	1.8	0.2	0.2	0.2	4.2	96	0.2	35	22	35	52	19	50	42
2,2'3,4'5'6-hexachlorobiphenyl	2.6	5	0.2	5	1.8	0.2	0.2	0.2	2.6	74	0.2	24	50	48	35	12	35	56
2,2'3,5,5'6-hexachlorobiphenyl	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	2.5	15	0.1	7	9.9	10	9.7	5.8	0.1	0.1
22',44',55'-hexachlorobiphenyl	0.1	2.3	0.1	3	1.1	0.1	0.1	0.1	12	74	0.1	44	23	27	42	15	40	42
22',44',66'-hexachlorobiphenyl	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	1.1	5.9	0.1	0.1	37	36	0.1	10	52	29
2,3,3'4,4'5-hexachlorobiphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	3.2	6.4	0.2	0.2	934	0.2	4.2	2.2	0.2	0.2
2,3,3'44'5'-hexachlorobiphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
2,3,3'4,4'6-hexachlorobiphenyl	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	6.4	0.1	0.1	0.1	0.1	3.3	0.1	0.1	0.1
23',44',55'-hexachlorobiphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	4.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
23',44',5'6-hexachlorobiphenyl	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	13	18	0.1	0.1	33	32	50	15	0.1	0.1
3,3'4,4'55'-hexachlorobiphenyl	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
22'33'44'5-heptachlorobiphenyl	0.2	0.2	0.2	0.2	0.4	0.2	0.2	0.2	2.8	9.4	0.2	0.2	12	9.8	3.6	2.8	0.2	0.2
22'33'44'6-heptachlorobiphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.6	5	0.2	0.2	0.2	0.2	1.8	0.2	0.2	0.2
22'33'4'56-heptachlorobiphenyl	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.6	3.5	0.1	0.1	11	0.1	1.5	0.1	0.1	0.1
22'33'55'6-heptachlorobiphenyl	2.2	2.8	0.2	0.2	2.4	0.2	0.2	0.2	11	0.2	0.2	58	52	59	75	2.2	0.2	74
22'344'55'-heptachlorobiphenyl	0.2	1.2	0.2	0.2	0.2	0.2	0.2	0.2	2.6	30	0.2	16	30	16	13	6.8	26	29
22'344'5'6-heptachlorobiphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.8	5.2	0.2	0.2	13	0.2	2	3	0.2	0.2
22'34'55'6-heptachlorobiphenyl	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	7.7	16	0.1	0.1	14	20	8.7	11	0.1	31
22'34'566'-heptachlorobiphenyl	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	5.5	7.9	0.1	0.1	0.1	0.1	4.7	4.2	0.1	0.1
233'44'55'-heptachlorobiphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
233'44'5'6-heptachlorobiphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	9.6	5.6	0.2	3.2	0.2	0.2
22'33'44'55'-octa(Cl)biphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.4	4.8	0.2	0.2	7.6	0.2	1.4	2.8	0.2	0.2
22'33'455'6'-octa(Cl)biphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	1.4	0.2	0.2	0.2	8.2	0.2	0.2	3.4	0.2	0.2
22'33'45'66'-octa(Cl)biphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	11	13	8.2	3	0.2	0.2
22'33'55'66'-octa(Cl)biphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	1.6	0.2	0.2	7.6	0.2	1.8	0.2	0.2	0.2
233'44'55'6-octachlorobiphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
22'33'44'55'6-nona(Cl)biphenyl	6.6	2.2	3	5	1.6	11	1	4	0.2	5.2	0.2	12	37	37	1.6	8.4	12	6
22'33'455'66'-nona(Cl)biphenyl	6.6	2.2	4	5	1.6	11	0.8	4	3.6	5.4	0.2	10	38	37	2.6	8.6	10	9
PCB congeners; total	91	51	250	56	47	140	120	300	310	3400	310	680	1400	930	3300	270	1200	2400

Table D1. Continued.

Chemical	LC16 AMP	LC16 ODON	LC18 ODON	LC18 AMP	LC16 CHIR	LC18 OLIG	LC19 ODON	LC19 AMP	LC19 CHIR	LC19 OLIG	LC29 ODON	LC29 CHIR	LC29 AMP	LC29 OLIG	LC38 AMP	LC38 ODON	LC38 CHIR	LC38 OLIG
percent lipid	5.3	10	7.7	7.4	11	2.7	2.2	5.2	7.8	26	5.8	11	3.4	31	8.7	10	23	9.3
2,2',5-tricholorobiphenyl	100	41	50	80	30	100	20	68	56	110	40	84	26	440	16	32	41	280
2,2',6-trichlorobiphenyl	2	2	2	2	2	2	16	16	2	2	2	2	2	2	2	2	2	2
2,3,4'-trichlorobiphenyl	18	50	23	190	420	340	0.5	0.5	0.5	0.5	5.5	140	9	120	5	0.5	0.5	50
2,4,4'-trichlorobiphenyl	61	40	16	77	66	83	11	72	49	130	15	19	30	150	8	11	130	110
2',3,4-tricholobiphenyl	55	15	6	53	17	42	5	33	1	61	6	3.5	25	1	4.5	8.5	81	97
3,4,4'-trichlorobiphenyl	130	17	7	21	17	18	1	1	7	1	1	10	18	29	1	1	1	1
2,2',3,5-tetrachlorobiphenyl	230	11	1.7	130	38	150	2	130	39	0.1	9.4	6.2	48	88	0.3	6.8	2	28
2,2',4,5-tetrachlorobiphenyl	120	24	5.9	140	58	99	6.4	160	52	43	20	42	35	240	0.4	11	11	55
2,2',5,5'-tetrachlorobiphenyl	260	22	8.7	190	55	200	4.5	190	6.2	0.1	12	36	69	25	8.4	9.5	140	45
2,2',6,6'-tetrachlorobiphenyl	11	5	13	2	7.4	9.4	3.2	0.8	26	42	13	28	4.4	120	0.4	4.4	130	15
2,3',4',5-tetrachlorobiphenyl	280	42	5.8	130	97	160	6.1	140	41	0.1	21	17	42	250	0.1	10	57	50
2,4,4',5-tetrachlorobiphenyl	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
3,3',4,4'-tetrachlorobiphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
3,4,4',5-tetrachlorobiphenyl	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5
2,2'3,4,5'-pentachlorobiphenyl	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
2,2'3,5',6-pentachlorobiphenyl	460	83	39	330	240	320	26	350	170	110	15	63	90	200	5.8	15	6	63
2,2'4,4',5-pentachlorobiphenyl	110	8	14	46	40	52	0.9	41	44	0.1	1.4	0.9	4.6	78	1.5	9.5	43	28
2,2'4,5,5'-pentachlorobiphenyl	130	4	11	87	17	91	5.1	93	18	0.1	11	33	20	55	5	3.5	63	17
2,2'4,6,6'-pentachlorobiphenyl	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
2,3,3'4,4'-pentachlorobiphenyl	80	4	5.3	24	16	7.1	2.4	11	40	0.1	3.7	4.8	9.6	45	1.1	0.1	15	0.1
2,3,3'4',6-pentachlorobiphenyl	180	13	15	100	37	130	0.1	110	2.6	0.1	8.6	35	31	170	0.3	2.2	59	11
2,3,4,4',5-pentachlorobiphenyl	4.3	0.1	0.1	1.2	3.1	1	0.1	1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
2,3'4,4',5-pentachlorobiphenyl	130	11	6.7	56	94	76	5.5	60	95	0.1	6	14	19	98	2.7	0.1	32	0.1
2,3'4,4',6-pentachlorobiphenyl	0.7	0.1	3.8	2	0.1	0.1	0.1	2.3	0.1	0.1	0.1	6.2	0.9	0.1	0.1	0.1	0.1	0.1
2',3,4,4',5-pentachlorobiphenyl	42	10	0.2	9.2	28	41	7	57	77	0.2	0.2	0.2	7.8	97	0.2	0.2	0.2	0.2
3,3'4,4',5-pentachlorobiphenyl	2.5	4	32	15	2.5	11	0.7	3	0.1	0.1	0.1	0.1	0.5	58	1.4	2.6	25	9.7
22',33',44'-hexachlorobiphenyl	6.2	3	1.8	3.8	0.2	5.6	2.8	4.4	0.2	0.2	0.2	5.8	2.2	0.2	0.2	0.2	0.2	0.2
2,2'3,44'5'-hexachlorobiphenyl	49	14	10	17	24	60	6.8	70	160	0.2	8.8	1.8	16	61	2.2	0.2	0.2	0.2
2,2'3,4'5'6-hexachlorobiphenyl	93	8	8.2	22	25	53	1.4	110	38	49	5.4	21	16	84	2.6	0.2	42	12
2,2'3,5,5'6-hexachlorobiphenyl	8	6	4	6.2	11	10	3.4	17	18	0.1	0.1	6.1	3.3	36	0.1	0.1	0.1	0.1
22',44',55'-hexachlorobiphenyl	37	13	8.5	12	24	28	13	67	190	47	10	28	11	57	2.1	0.1	0.1	0.1
22',44',66'-hexachlorobiphenyl	0.1	18	5.9	11	25	13	1.4	0.1	0.1	0.1	13	43	1.8	150	3.6	0.1	0.1	0.1
2,3,3'4,4'5-hexachlorobiphenyl	3.8	2	1.2	20	8.2	3.4	0.6	3.6	16	0.2	0.2	0.2	1	0.2	0.2	0.2	0.2	0.2
2,3,3'44'5'-hexachlorobiphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
2,3,3'4,4'6-hexachlorobiphenyl	2.4	0.1	0.1	1.7	0.1	3.2	0.1	1.7	15	0.1	0.1	0.1	1.1	0.1	0.1	0.1	0.1	0.1
23',44',55'-hexachlorobiphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
23',44',5'6-hexachlorobiphenyl	92	8	8.8	25	6.8	11	8.6	18	150	0.1	0.1	12	17	68	0.1	0.1	0.1	0.1
3,3'4,4'55'-hexachlorobiphenyl	1.3	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
22'33'44'5-heptachlorobiphenyl	3	0.2	0.2	2.4	8.6	8.8	4.2	10	6	0.2	0.2	7	0.2	0.2	0.2	0.2	0.2	0.2
22'33'44'6-heptachlorobiphenyl	1.6	0.2	0.2	0.8	0.2	1.6	1.2	1.6	20	0.2	0.2	0.2	1	0.2	0.2	0.2	0.2	0.2
22'33'4'56-heptachlorobiphenyl	0.9	0.1	0.1	2.3	0.1	5.7	0.4	7.9	31	0.1	0.1	0.1	2.7	0.1	0.1	0.1	0.1	0.1
22'33'55'6-heptachlorobiphenyl	79	4	14	30	41	61	14	150	260	0.2	13	13	20	110	3.8	0.2	0.2	0.2
22'344'55'-heptachlorobiphenyl	9.8	5	4	8.2	3.6	2.6	5.2	33	110	47	6.2	18	3.2	44	0.2	0.2	2	0.2
22'344'5'6-heptachlorobiphenyl	1.2	0.2	0.2	2.4	5.6	6.8	4.6	7.2	0.2	0.2	0.2	0.2	2.8	0.2	0.4	0.2	0.2	0.2
22'34'55'6-heptachlorobiphenyl	7.3	7	2.5	5.8	11	33	11	27	2.5	40	6.2	11	1.4	0.1	0.9	0.1	0.1	0.1
22'34'566'-heptachlorobiphenyl	3.3	0.1	0.1	4	6.9	8.9	5.2	3.6	26	0.1	0.1	0.1	4.5	0.1	0.1	0.1	0.1	0.1
233'44'55'-heptachlorobiphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.6	0.2	0.2	0.2
233'44'5'6-heptachlorobiphenyl	1	0.2	2.4	1.2	0.2	0.2	0.2	0.2	0.2	0.2	3	6	1.2	0.2	0.2	0.2	27	0.2
22'33'44'55'-octa(Cl)biphenyl	0.8	0.2	0.2	0.8	0.2	3.8	0.4	4.8	19	0.2	0.2	0.2	3	0.2	0.2	0.2	0.2	0.2
22'33'455'6'-octa(Cl)biphenyl	2.4	0.2	0.2	1.4	4	9	4.6	3	20	0.2	0.2	0.2	1.4	0.2	0.2	0.2	0.2	0.2
22'33'45'66'-octa(Cl)biphenyl	6.2	0.2	1.8	4.4	6.8	12	0.2	34	0.2	0.2	0.2	6.6	4.2	0.2	0.2	0.2	32	0.2
22'33'55'66'-octa(Cl)biphenyl	0.2	2	0.2	0.2	5.6	0.2	0.2	2.2	0.2	0.2	2.4	0.2	1.2	0.2	1.8	0.2	0.2	8.4
233'44'55'6-octachlorobiphenyl	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
22'33'44'55'6-nona(Cl)biphenyl	0.8	11	6	4	12	10	4.8	5.6	34	10	9.4	17	2.8	60	2.8	0.2	0.2	0.2
22'33'455'66'-nona(Cl)biphenyl	2.6	11	9	4.2	12	8.8	5.4	6.8	42	10	9.8	19	2.4	6	0.2	8.6	0.2	28
PCB congeners; total	2800	510	350	1900	1500	2300	220	2100	1900	770	270	760	600	2900	82	140	940	910

Table D2. Total PAHs in benthic invertebrate taxa (µg/g dry weight) – 2002 sites only.

Area	Site	BIOTA – Total PAHs			
		Chironomid	Amphipod	Oligochaete	Odonate
Reference	**BEC01**	-[a]	-[a]	-[a]	-[a]
	BEC02	-[a]	-[a]	-[a]	-[a]
	BLC01	**0.30**	**0.12**	-[b]	**0.22**
	BLC02	**0.44**	**0.24**	-[b]	**0.28**
	UC01	-[b]	-[b]	-[b]	-[b]
	TC40	-[b]	-[b]	-[b]	-[b]
Lyons Creek	LC01	-[b]	0.52	0.32	0.26
	LC03	-[b]	12.06	2.20	1.90
	LC06	-[b]	-[b]	-[b]	-[b]
	LC08	-[b]	-[b]	-[b]	-[b]
	LC10	-[b]	-[b]	-[b]	-[b]
	LC12	2.58	5.10	1.72	0.64
	LC14	-[b]	-[b]	-[b]	-[b]
	LC16	-[b]	-[b]	-[b]	-[b]
	LC17	0.56	0.56	0.22	1.56
	LC18	-[b]	-[b]	-[b]	-[b]
	LC19	-[b]	-[b]	-[b]	-[b]
	LC22	-[b]	-[b]	-[b]	-[b]
	LC23	-[b]	-[b]	-[b]	-[b]
	LC29	-[b]	-[b]	-[b]	-[b]
	LC38	-[b]	-[b]	-[b]	-[b]

[a] benthos not collected [b] taxa not analyzed

Table D3. Biota-sediment accumulation factors (BSAF) for total PCBs.

Area	Site	BSAF			
		Chironomid	Amphipod	Oligochaete	Odonate
	Mean % lipid	13.78	5.74	17.86	7.58
Ref. Creeks	BEC01	-[a]	-[a]	-[a]	-[a]
	BEC02	-[a]	-[a]	-[a]	-[a]
	BLC01	12.80	40.49	-[a]	42.66
	BLC02	24.15	61.88	-[a]	85.28
	UC01	8.83	8.80	1.37	5.71
	TC40	5.18	5.11	10.29	10.96
Lyons Creek	LC01	-[a]	10.64	1.70	2.71
	LC03	-[a]	0.28	0.01	0.001
	LC06	-[b]	-[b]	-[b]	-[b]
	LC08	0.04	0.47	0.01	0.02
	LC10	-[b]	-[b]	-[b]	-[b]
	LC12	0.22	1.24	1.91	0.09
	LC14	0.21	1.13	0.10	0.07
	LC16	0.55	3.16	0.88	0.44
	LC17	1.27	3.60	0.89	0.16
	LC18	1.44	4.25	1.67	0.60
	LC19	0.69	1.84	0.19	0.15
	LC22	-[b]	-[b]	-[b]	-[b]
	LC23	-[b]	-[b]	-[b]	-[b]
	LC29	1.24	2.39	3.64	0.82
	LC38	46.29	9.74	34.61	12.14

[a] insufficient tissue sample size – no analysis
[b] tissue not collected

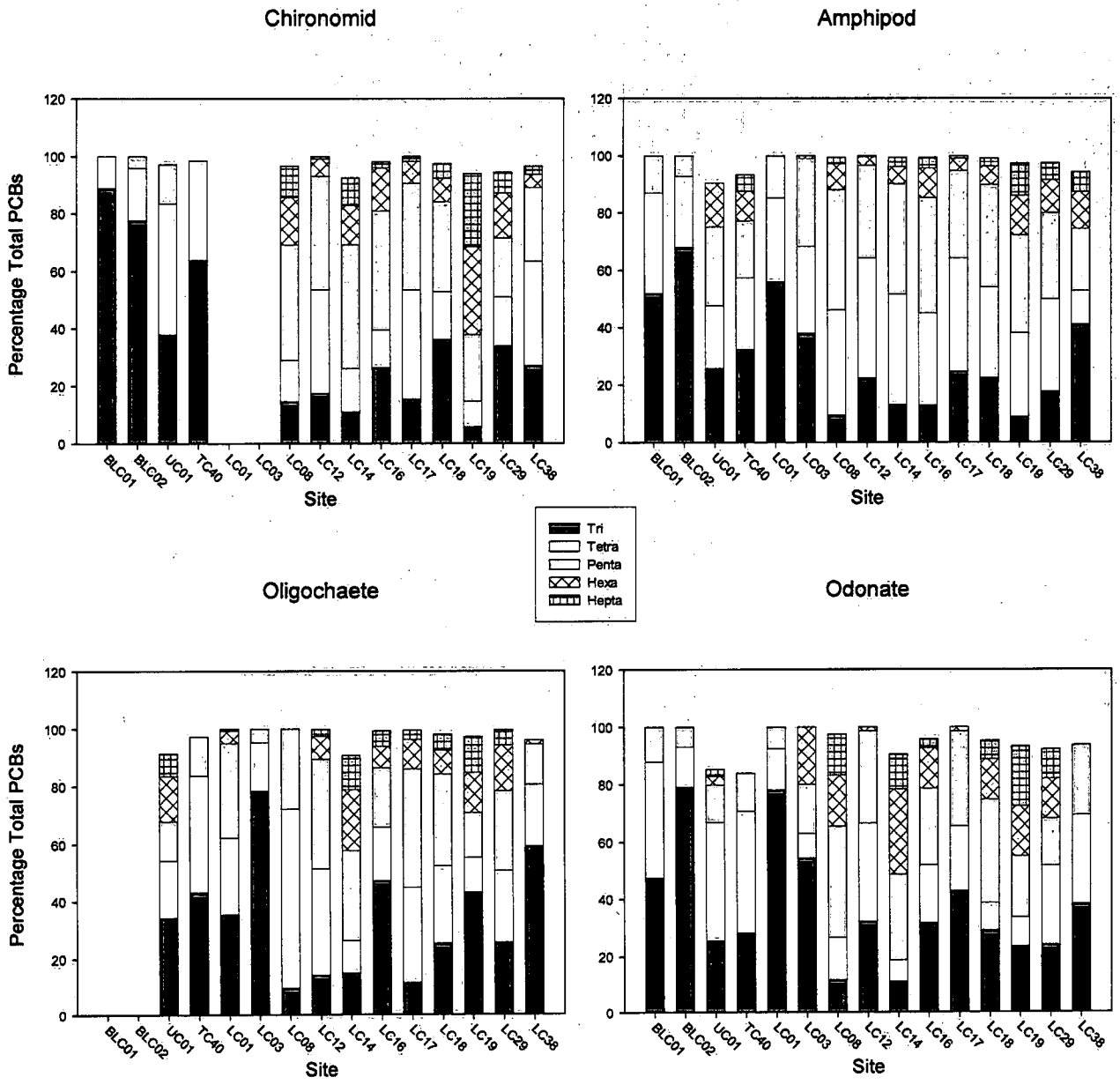


Figure D1. Isomeric composition (%) of benthic invertebrates from Lyons Creek (LC) and reference creek (BLC, UC, TC) sediment.

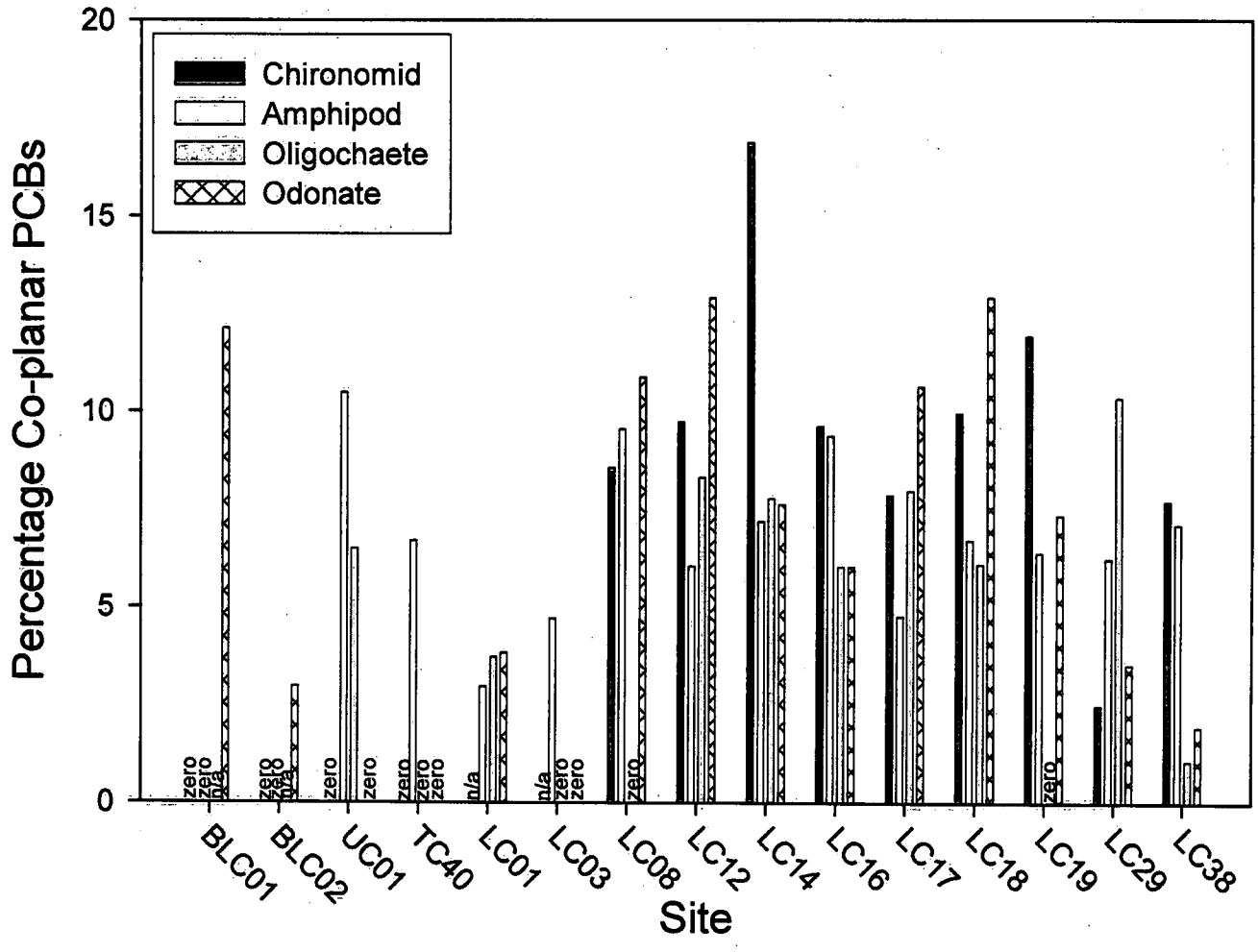
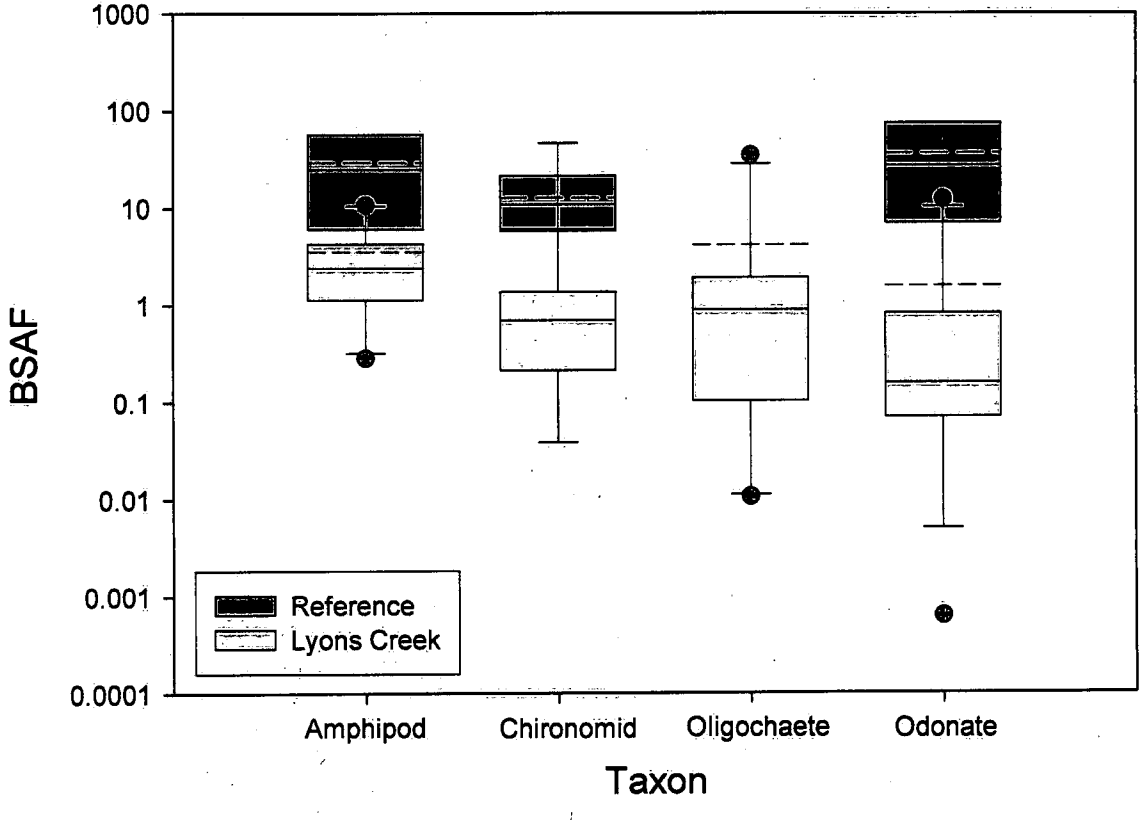


Figure D2. Percentage of co-planar PCBs in Lyons Creek (LC) and reference creek (BLC, UC, TC) benthic invertebrates.



Taxon	Mean % Lipids
Amphipod	5.74
Chironomid	13.78
Oligochaete	17.86
Odonate	7.58

Figure D3. Biota-sediment accumulation factors for Lyons Creek (grey) and reference creeks (green). The red dotted line is the mean for Lyons Creek sites and the black dotted line is the mean for reference sites. The solid lines are the median values. The bottom and top of the boxes represents the 25th and 75th percentiles, respectively. The lower and upper whiskers (Lyons Creek sites only) represent the 10th and 90th percentiles, respectively. Outliers are shown as solid circles.

APPENDIX E. Toxicity Ordinations and Toxicity-Contaminant Relationships

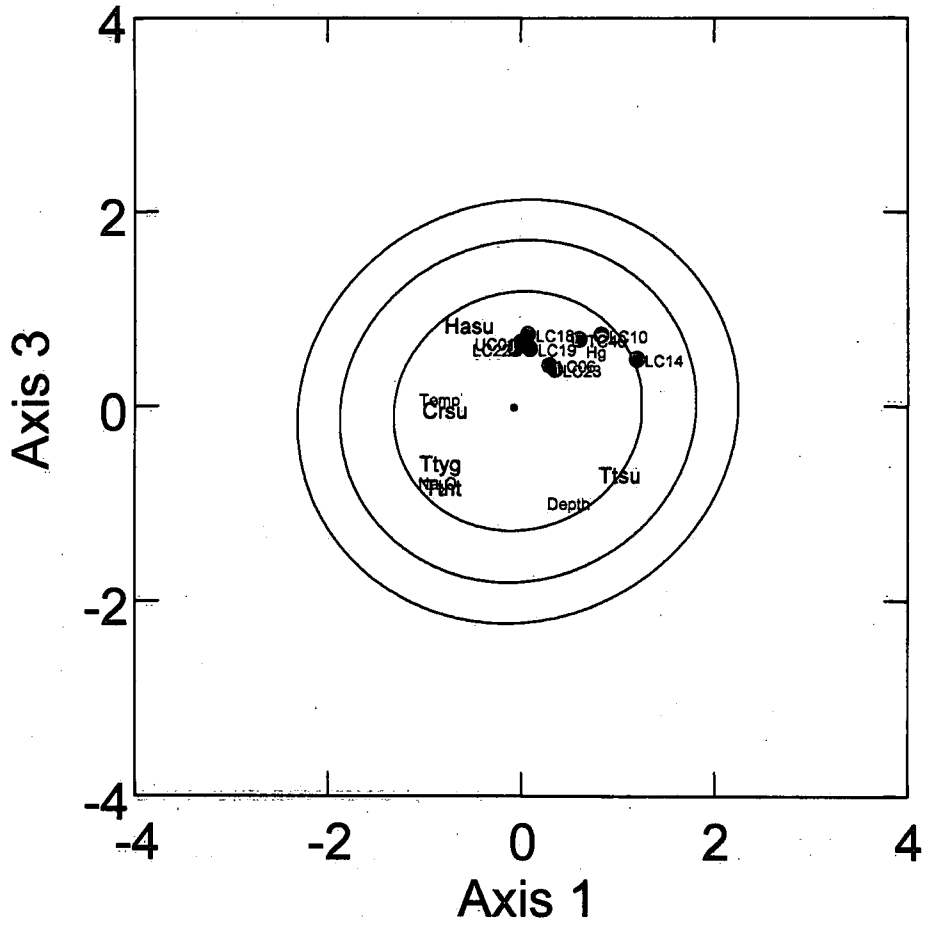


Figure E1. Ordination of subset of Lyons Creek sites using 10 toxicity test endpoints summarized on Axes 1 and 3, with 90%, 99%, and 99.9% probability ellipses around reference sites (not shown). Most significant toxicity endpoints and environmental variables are shown. [*Tubifex* young production (Ttyg), *Chironomus* survival (Crsu), *Hyalella* survival (Hasu), Tuvifex percent cocoon hatch and survival (Ttht, Ttsu)]. Maximum stress level = 0.09.

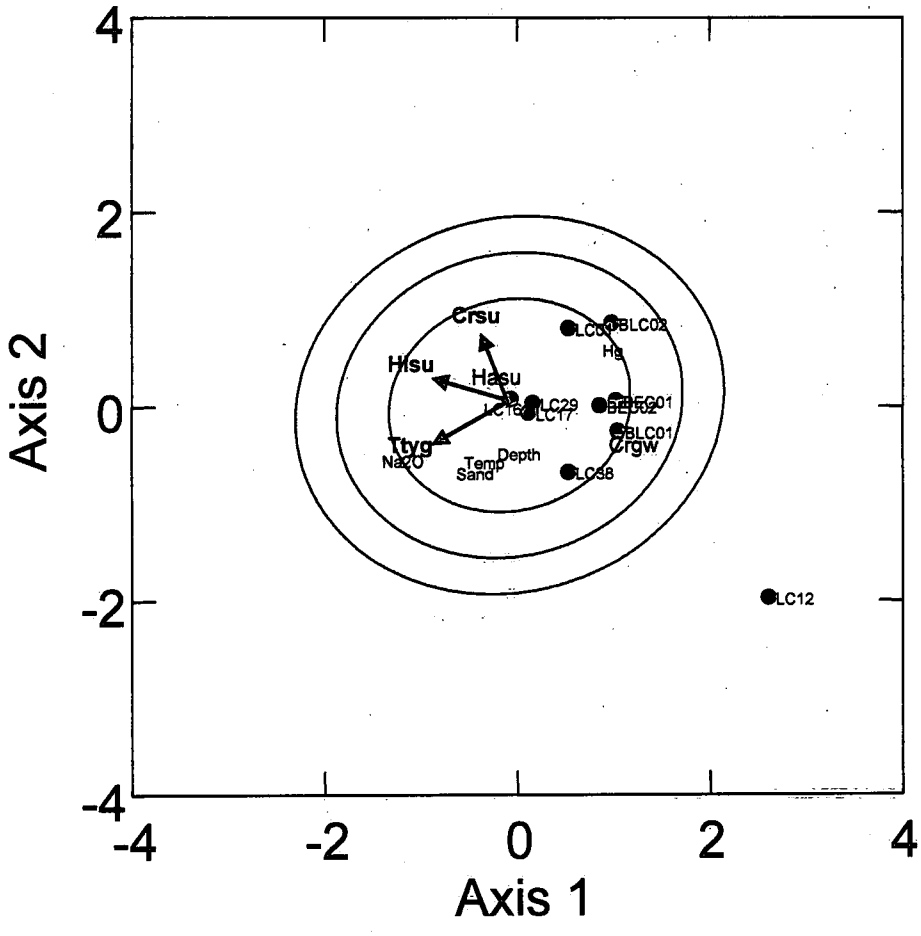


Figure E2. Ordination of subset of Lyons Creek sites using 10 toxicity test endpoints summarized on Axes 1 and 2, with 90%, 99%, and 99.9% probability ellipses around reference sites (not shown). Most significant toxicity endpoints and environmental variables are shown. [*Hyalella* survival (Hasu), *Tubifex* young production (Ttyg), *Chironomus* survival and growth (Crsu, Crgw), *Hexagenia* survival (Hlsu)]. Maximum stress level = 0.08.

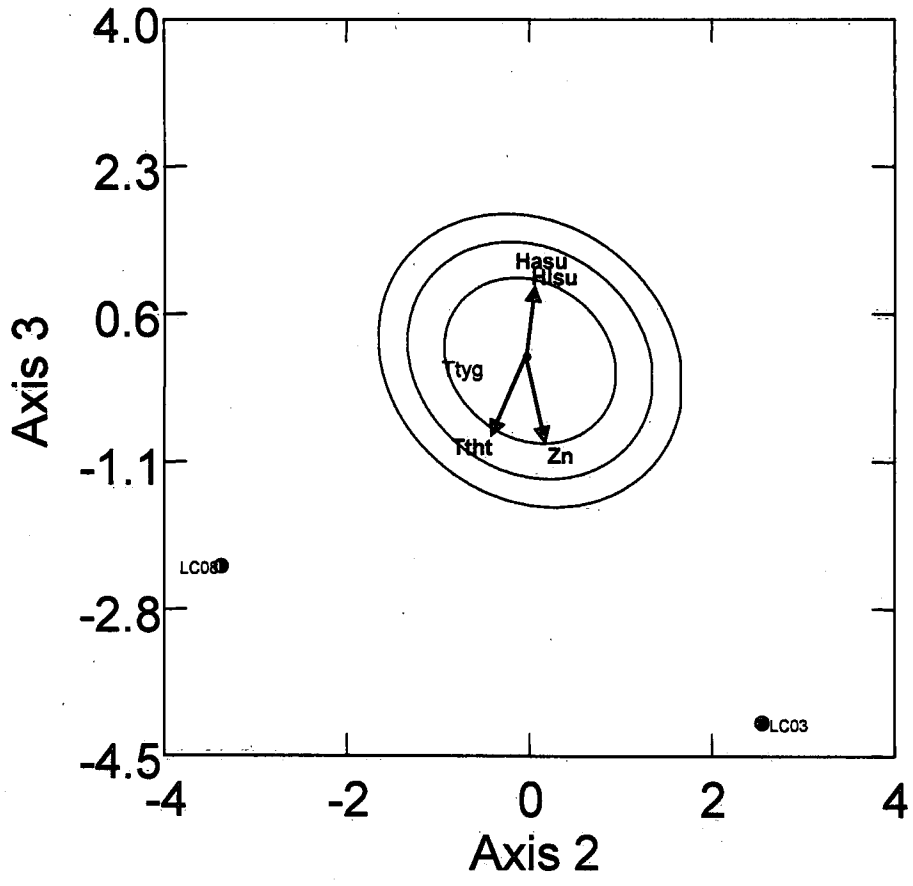


Figure E3. Ordination of LC03 and LC08 using 10 toxicity test endpoints summarized on Axes 2 and 3, with 90%, 99%, and 99.9% probability ellipses around reference sites (not shown). Most significant toxicity endpoints and environmental variables are shown. [*Hyalella* survival (Hasu), *Tubifex* young production (Ttyg), *Hexagenia* survival (Hlsu), *Tubifex* percent cocoon hatch (Ttht)]. Maximum stress level = 0.08.

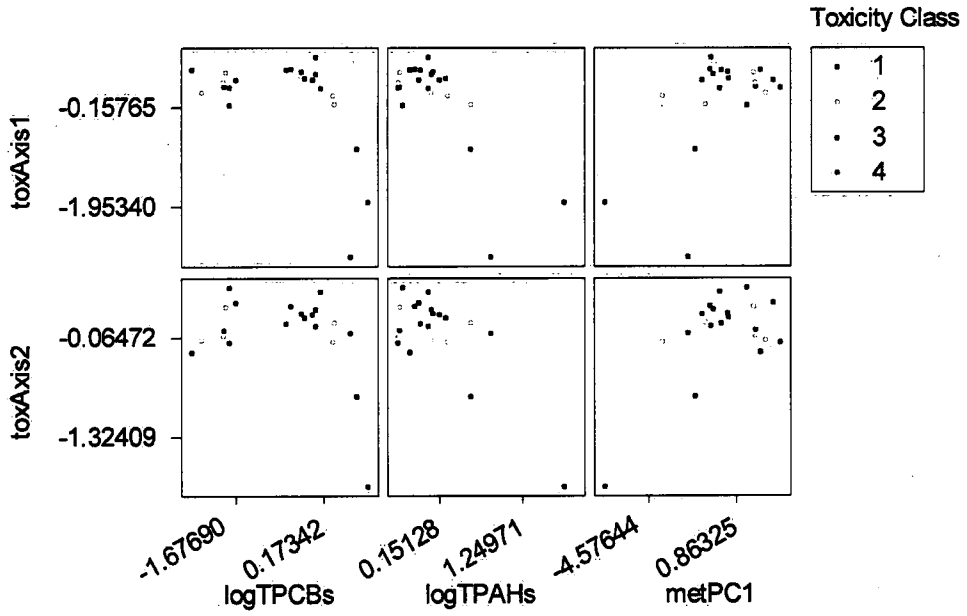


Figure E4. Lyons Creek sediment toxicity relationships to contaminant concentrations based on integrated descriptors. Low values for Axis 1 correspond to sites with high relative toxicity to *Hexagenia*, *Hyalella* and *Tubifex* survival. Low values for Axis 2 correspond to sites with high relative toxicity to *Chironomus survival*. Sites are colour-coded by toxicity class as determined by the BEAST assessment with reference sites.

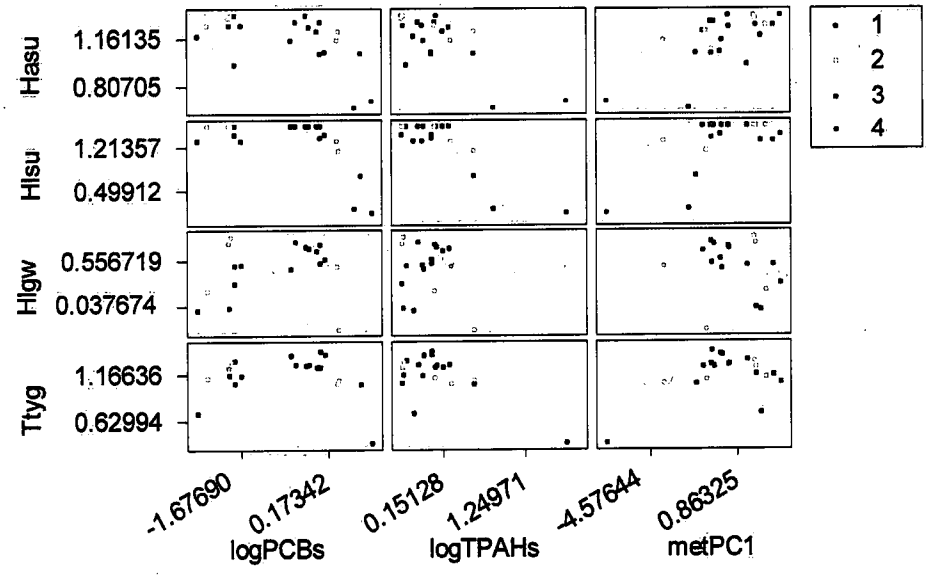


Figure E5. Lyons Creek sediment toxicity relationships to contaminant concentrations based on individual toxicity endpoint and integrated metal, PCB and PAH concentrations. "Hasu, Hlsu" = survival of *Hyalella* and *Hexagenia*, respectively, "Hlgw" = *Hexagenia* growth, "Ttyg" = *Tubifex* young production. Sites are colour-coded by toxicity class as determined by the BEAST assessment with reference sites.

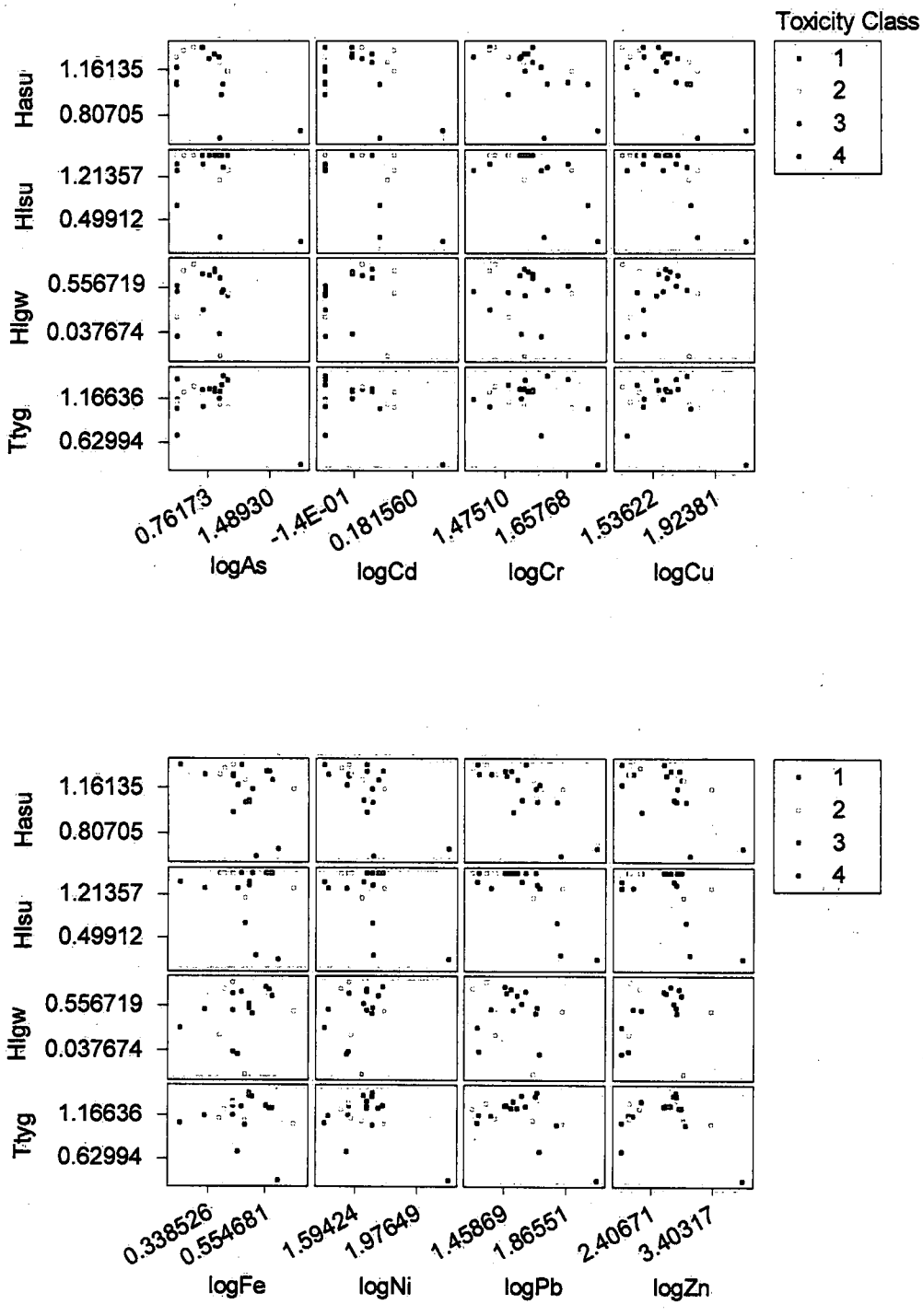
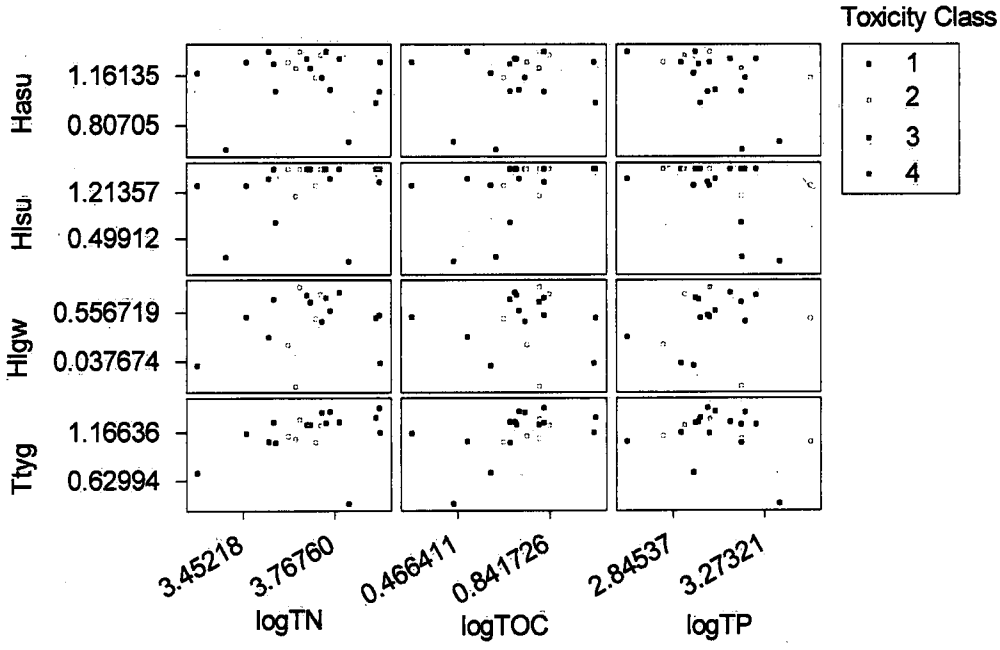


Figure E6. Lyons Creek sediment toxicity relationships to sediment contaminant concentrations based on individual toxicity endpoint and individual metal concentrations. "Hasu, Hlsu" = survival of *Hyalella* and *Hexagenia*, respectively, "Hlgw" = *Hexagenia* growth, "Ttyg" = *Tubifex* young production. Sites are colour-coded by toxicity class as determined by the BEAST assessment with reference sites.



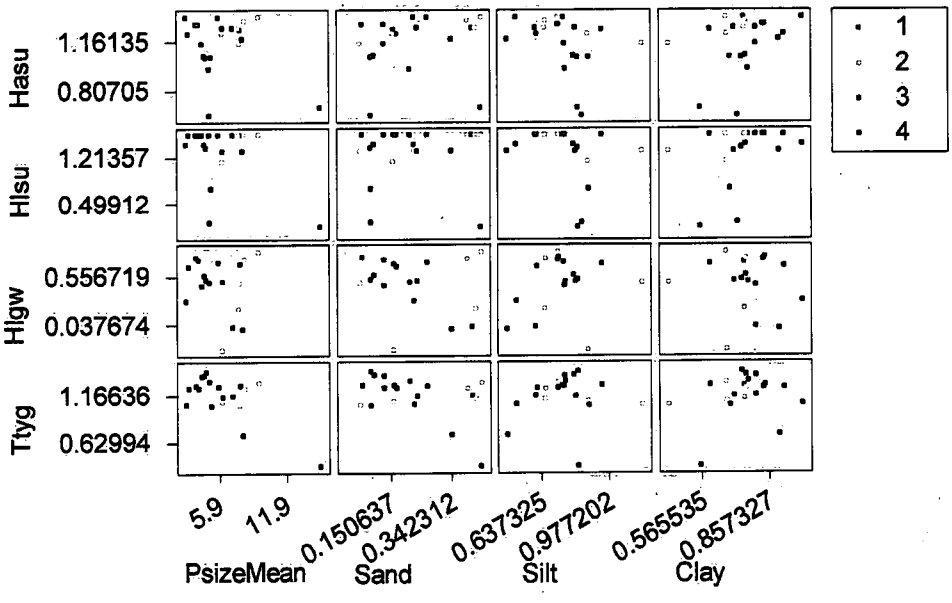


Figure E7. Lyons Creek sediment toxicity relationships to sediment nutrients (top) and particle size (bottom). "Hasu, Hlsu" = survival of *Hyalella* and *Hexagenia*, respectively, "Hlgw" = *Hexagenia* growth, "Ttyg" = *Tubifex* young production. Sites are colour-coded by toxicity class as determined by the BEAST assessment with reference sites.

APPENDIX F. Benthic Invertebrate Family Counts

Table F1. Mean abundance (per m^2) of invertebrate families.

Family	BEC01	BEC02	BLC01	BLC02	UC01	TC40	LC01	LC03	LC06	LC08	LC10
Ancylidae	0.0	0.0	0.0	0.0	241.3	0.0	35.9	0.0	120.6	0.0	361.9
Aoridae	60.3	60.3	107.6	71.7	0.0	0.0	26.9	301.6	0.0	0.0	0.0
Arrenuridae	0.0	0.0	9.0	0.0	0.0	0.0	9.0	0.0	0.0	60.3	0.0
Asellidae	904.7	1266.6	89.6	0.0	2533.2	5488.5	53.8	60.3	0.0	0.0	0.0
Aturidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Baetidae	0.0	241.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Bosminidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Caenidae	6513.9	3015.7	430.2	9.0	784.1	723.8	170.3	542.8	422.2	180.9	0.0
Candoniidae	1688.8	2352.2	806.6	546.7	603.1	8926.4	1084.5	0.0	0.0	0.0	0.0
Ceratopogonidae	4825.1	1146.0	923.1	376.4	120.6	542.8	385.4	1146.0	1387.2	180.9	241.3
Chaoboridae	0.0	120.6	44.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Chironomidae	27322.1	16887.8	2116.9	2723.4	5006.0	12786.5	6505.3	34077.2	49698.4	6091.7	3076.0
Chrysomelidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Chydoridae	482.5	603.1	35.9	0.0	1326.9	3498.2	9.0	0.0	1869.7	180.9	60.3
Coenagrionidae	60.3	180.9	44.8	116.5	60.3	120.6	412.3	1869.7	120.6	180.9	241.3
Corixidae	0.0	60.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Crangonyctidae	0.0	60.3	0.0	53.8	0.0	0.0	71.7	60.3	0.0	0.0	0.0
Culicidae	60.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cyclocyprididae	1447.5	2834.7	5512.0	1909.0	1628.5	3015.7	0.0	0.0	7056.7	0.0	0.0
Cyprididae	18697.2	25814.2	44.8	125.5	7177.3	18094.1	0.0	0.0	18697.2	120.6	60.3
Daphnidae	241.3	422.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Dreissenidae	0.0	120.6	35.9	0.0	0.0	0.0	2805.3	0.0	60.3	0.0	0.0
Dugesiidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	542.8	1990.3	1085.6
Elmidae	2653.8	542.8	349.5	510.9	361.9	1266.6	1335.4	301.6	0.0	0.0	0.0
Enchytraeidae	60.3	0.0	0.0	0.0	60.3	0.0	22.5	0.0	0.0	0.0	0.0
Ephemeridae	0.0	0.0	35.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Erpobdellidae	0.0	0.0	0.0	0.0	0.0	0.0	125.5	0.0	0.0	0.0	0.0
Gammaridae	180.9	120.6	62.7	206.1	0.0	0.0	107.6	422.2	361.9	60.3	0.0
Glossiphoniidae	0.0	0.0	0.0	0.0	180.9	0.0	9.0	60.3	60.3	0.0	60.3
Hyalellidae	0.0	180.9	9.0	26.9	0.0	120.6	573.6	482.5	60.3	120.6	60.3
Hydridae	0.0	0.0	0.0	0.0	0.0	0.0	26.9	0.0	120.6	60.3	482.5
Hydrobiidae	301.6	1387.2	35.9	0.0	0.0	1568.2	107.6	542.8	0.0	0.0	0.0
Hydrodromidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	60.3	0.0
Hydrophilidae	0.0	0.0	0.0	0.0	0.0	0.0	9.0	120.6	0.0	0.0	0.0
Hydroptilidae	60.3	0.0	26.9	35.9	241.3	0.0	0.0	0.0	180.9	0.0	0.0
Hydrozetiidae	60.3	0.0	0.0	0.0	0.0	60.3	0.0	0.0	0.0	0.0	0.0
Hygrobatidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Lebertiidae	0.0	0.0	0.0	0.0	0.0	0.0	9.0	0.0	60.3	0.0	0.0
Leptoceridae	0.0	301.6	80.7	26.9	60.3	60.3	98.6	0.0	0.0	0.0	0.0
Libellulidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Limnesiidae	0.0	0.0	0.0	0.0	0.0	0.0	80.7	0.0	0.0	241.3	120.6
Limnocytheridae	0.0	0.0	0.0	0.0	0.0	663.4	0.0	0.0	0.0	0.0	0.0
Lumbriculidae	0.0	0.0	0.0	0.0	0.0	120.6	0.0	0.0	0.0	0.0	0.0
Macrothricidae	0.0	180.9	0.0	62.7	1507.8	0.0	0.0	422.2	603.1	0.0	0.0
Muscidae	0.0	0.0	0.0	0.0	0.0	60.3	0.0	0.0	0.0	60.3	180.9
Naididae	60.3	1507.8	56.2	1473.0	361.9	0.0	45.0	603.1	0.0	120.6	603.1
Phrygaenidae	60.3	0.0	0.0	9.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Physidae	0.0	422.2	0.0	0.0	120.6	120.6	0.0	60.3	0.0	0.0	0.0
Pionidae	0.0	0.0	17.9	9.0	0.0	0.0	0.0	120.6	1507.8	603.1	60.3
Piscicolidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Plagiostomidae	0.0	0.0	71.7	0.0	0.0	663.4	0.0	301.6	784.1	60.3	0.0
Planariidae	0.0	0.0	9.0	0.0	0.0	0.0	233.0	1387.2	0.0	0.0	0.0
Planorbidae	0.0	120.6	17.9	170.3	0.0	60.3	62.7	844.4	60.3	0.0	965.0
Pleidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Polycentropodidae	60.3	0.0	44.8	35.9	0.0	0.0	71.7	0.0	241.3	0.0	0.0
Pyralidae	0.0	0.0	0.0	0.0	60.3	0.0	9.0	120.6	0.0	0.0	60.3
Sabellidae	0.0	0.0	0.0	0.0	0.0	60.3	0.0	0.0	0.0	0.0	0.0
Sialidae	0.0	0.0	9.0	26.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Sperchontidae	0.0	0.0	0.0	0.0	0.0	0.0	26.9	0.0	0.0	0.0	0.0
Sphaeriidae	482.5	180.9	288.6	90.7	120.6	1990.3	0.0	0.0	60.3	120.6	60.3
Spongillidae	0.0	150542.8	14833.0	752.9	3437.9	1146.0	0.0	0.0	0.0	0.0	482.5
Stratiomyidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Syllidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Syrphidae	0.0	60.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tabanidae	0.0	60.3	0.0	17.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tetrastemmatidae	0.0	0.0	0.0	0.0	0.0	0.0	555.7	180.9	0.0	0.0	0.0
Tipulidae	0.0	0.0	0.0	53.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Trhypachthoniidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	60.3	0.0	0.0
Trochochaetidae	120.6	60.3	0.0	0.0	0.0	0.0	3997.3	1688.8	0.0	0.0	0.0
Tubificidae	7659.8	11037.4	5048.5	697.1	7478.9	7659.8	5543.3	17370.3	40711.7	27744.3	17913.1
Unionicolidae	0.0	0.0	0.0	0.0	0.0	120.6	0.0	0.0	0.0	0.0	0.0
Valvatidae	0.0	361.9	9.0	0.0	120.6	301.6	0.0	0.0	0.0	60.3	0.0

Table F1. Continued.

Family	LC12	LC14	LC16	LC17	LC18	LC19	LC22	LC23	LC29avg	LC38
Ancylidae	0.0	965.0	0.0	60.3	784.1	2111.0	2171.3	301.6	0.0	0.0
Aoridae	180.9	0.0	542.8	422.2	0.0	0.0	0.0	0.0	382.0	0.0
Arrenuridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	120.6
Asellidae	0.0	965.0	0.0	180.9	6513.9	0.0	60.3	241.3	20.1	422.2
Aturidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	60.3
Baetidae	0.0	60.3	0.0	60.3	180.9	0.0	0.0	60.3	0.0	0.0
Bosminidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	20.1	0.0
Caenidae	0.0	6152.0	180.9	2352.2	1749.1	2955.4	2050.7	542.8	180.9	542.8
Candoniidae	0.0	180.9	0.0	0.0	120.6	301.6	1628.5	904.7	422.2	5066.3
Ceratopogonidae	301.6	1447.5	1206.3	4825.1	241.3	180.9	603.1	361.9	301.6	0.0
Chaoboridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Chironomidae	8202.7	58202.7	24306.4	92400.5	22798.6	21833.5	30096.5	27322.1	9469.2	15621.2
Chrysomelidae	0.0	0.0	0.0	0.0	0.0	0.0	60.3	0.0	40.2	120.6
Chydoridae	0.0	844.4	0.0	422.2	5187.0	482.5	1749.1	2352.2	40.2	361.9
Coenagrionidae	0.0	1206.3	180.9	1266.6	663.4	1266.6	844.4	422.2	20.1	361.9
Corixidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	40.2	0.0
Crangonyctidae	0.0	0.0	0.0	120.6	0.0	0.0	0.0	0.0	0.0	241.3
Culicidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cyclocyprididae	0.0	71290.7	0.0	51146.0	1085.6	723.8	11700.8	1869.7	100.5	844.4
Cyprididae	0.0	21712.9	8082.0	1025.3	2593.5	1206.3	6996.4	241.3	120.6	784.1
Daphnidae	0.0	603.1	0.0	0.0	1930.0	0.0	60.3	0.0	0.0	60.3
Dreissenidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Dugesiidae	0.0	7478.9	0.0	0.0	2352.2	1447.5	301.6	542.8	0.0	0.0
Elmidae	0.0	0.0	0.0	0.0	0.0	0.0	60.3	0.0	0.0	784.1
Enchytraeidae	0.0	0.0	0.0	0.0	60.3	0.0	0.0	0.0	20.1	0.0
Ephemeridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Erpobdellidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Gammaridae	0.0	422.2	180.9	1206.3	1930.0	1146.0	301.6	301.6	40.2	0.0
Glossiphoniidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	60.3	0.0	60.3
Hyalellidae	0.0	1326.9	1628.5	2653.8	1749.1	482.5	2050.7	1749.1	20.1	723.8
Hydridae	0.0	241.3	0.0	0.0	0.0	482.5	1206.3	0.0	0.0	0.0
Hydrobiidae	60.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1085.6
Hydrodromidae	0.0	0.0	60.3	0.0	0.0	120.6	0.0	0.0	0.0	0.0
Hydrophilidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hydroptilidae	0.0	120.6	60.3	60.3	60.3	0.0	60.3	60.3	0.0	120.6
Hydrozetiidae	0.0	0.0	0.0	0.0	301.6	120.6	0.0	0.0	0.0	0.0
Hygrobatidae	60.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Lebertiidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Leptoceridae	0.0	11037.4	23703.3	11459.6	2533.2	241.3	542.8	180.9	0.0	3196.6
Libellulidae	0.0	0.0	0.0	0.0	60.3	0.0	0.0	0.0	0.0	0.0
Limnesiidae	0.0	0.0	0.0	0.0	0.0	0.0	180.9	0.0	0.0	0.0
Limnocytheridae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	120.6
Lumbriculidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	60.3
Macrothricidae	241.3	361.9	0.0	1749.1	60.3	0.0	0.0	120.6	422.2	60.3
Muscidae	0.0	60.3	0.0	0.0	0.0	0.0	482.5	0.0	0.0	0.0
Naididae	663.4	6031.4	3920.4	1688.8	1568.2	361.9	603.1	60.3	100.5	1326.9
Phrygaenidae	0.0	241.3	0.0	0.0	0.0	120.6	0.0	120.6	60.3	0.0
Physidae	0.0	0.0	120.6	180.9	60.3	60.3	0.0	0.0	0.0	2111.0
Pionidae	0.0	60.3	120.6	0.0	0.0	0.0	60.3	0.0	0.0	0.0
Piscicolidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	60.3
Plagiostomidae	0.0	1447.5	482.5	663.4	784.1	422.2	180.9	180.9	20.1	1146.0
Planariidae	60.3	0.0	542.8	2774.4	0.0	0.0	0.0	0.0	0.0	301.6
Planorbidae	0.0	2653.8	0.0	723.8	241.3	422.2	5428.2	0.0	0.0	2955.4
Pleidae	0.0	0.0	0.0	60.3	0.0	0.0	0.0	0.0	0.0	0.0
Polycentropodidae	0.0	120.6	60.3	60.3	60.3	3015.7	241.3	482.5	0.0	0.0
Pyralidae	60.3	120.6	0.0	60.3	60.3	180.9	0.0	0.0	0.0	0.0
Sabellidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Sialidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Sperchontidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Sphaeriidae	60.3	120.6	0.0	241.3	844.4	0.0	60.3	0.0	0.0	422.2
Spongillidae	0.0	422.2	0.0	0.0	60.3	0.0	120.6	60.3	20.1	43365.5
Stratiomyidae	0.0	0.0	0.0	60.3	0.0	0.0	0.0	0.0	20.1	60.3
Syllidae	0.0	0.0	0.0	60.3	0.0	0.0	0.0	0.0	0.0	0.0
Syrphidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tabanidae	0.0	0.0	0.0	0.0	301.6	60.3	422.2	301.6	0.0	60.3
Tetrastemmatidae	60.3	0.0	0.0	60.3	0.0	0.0	0.0	0.0	0.0	0.0
Tipulidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Trhypachthoniidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Trochochaetidae	361.9	0.0	0.0	60.3	0.0	0.0	0.0	0.0	0.0	0.0
Tubificidae	12665.9	6272.6	3196.6	13992.8	542.8	2171.3	2412.5	3498.2	5106.6	2774.4
Unionicolidae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Valvatidae	0.0	60.3	0.0	60.3	0.0	0.0	0.0	0.0	0.0	0.0

APPENDIX G. Quality Assurance/Quality Control Results

Table G1. Coefficients of variation (CV) for field-replicated site (LC29).

Parameter	CV
Al2O3 (%)	0.8
Alkalinity (mg/L)	0.4
As (ppm)	14.0
CaO (%)	0.4
Cd (ppm)	-
Clay (%)	2.9
Co (ppm)	1.1
Conductivity (uS/cm)	-
Cr (ppm)	2.0
Cu (ppm)	1.3
Depth (meters)	-
DO (mg/L)	-
Fe (%)	0.9
Gravel (%)	-
Hg (ppb)	21.6
K2O (%)	1.2
LOI (%)	3.1
Mg (%)	1.8
Mn (ppm)	1.8
Na2O (%)	2.8
NH3 (mg/L)	4.0
Ni (ppm)	2.1
NO3NO2 (mg/L)	22.0
P2O5 (%)	2.3
Pb (ppm)	6.0
Sand (%)	5.4
Silt (%)	3.1
SiO2 (%)	0.5
TiO2 (%)	0.4
TKN (mg/L)	3.2
TN (ppm)	2.2
TOC (%)	4.2
TP(Sed) (ppm)	4.3
TP(Wat) (mg/L)	17.9
V (ppm)	1.0
Zn (ppm)	1.0
PCBs	4.7
PAHs	35.8
Range	**0.4 - 35.8**

Table G2. Laboratory duplicate results (Caduceon).

Analyte	Units	Laboratory Duplicate (site LC12)				Laboratory Duplicate (site LC16)		
		Det Limit	Concn 1	Concn 2	CV	Concn 1	Concn 2	CV
Ag	µg/g	0.5	44.27	43.99	0.44	-	-	-
Al	pct	0.01	1.28	1.23	2.48	-	-	-
Al			12787.93	12347.31	2.48	-	-	-
As	µg/g	5	<5	<5	-	-	-	-
Ba	µg/g	1	111.83	109.59	1.43	-	-	-
Be	µg/g	0.2	0.64	0.63	1.03	-	-	-
Bi	µg/g	5	<5	<5	-	-	-	-
Ca	pct	0.01	6.53	6.33	2.16	-	-	-
Ca	µg/g		65272.81	63313.20	2.16	-	-	-
Cd	µg/g	1	0.98	0.68	25.50	-	-	-
Co	µg/g	1	12.29	12.51	1.26	-	-	-
Cr	µg/g	1	52.39	51.60	1.08	-	-	-
Cu	µg/g	1	59.13	57.07	2.50	-	-	-
Fe	pct	0.01	2.97	2.90	1.68	-	-	-
Fe	µg/g		29711.21	29013.36	1.68	-	-	-
K	pct	0.05	0.34	0.33	2.04	-	-	-
K	µg/g		3376.64	3280.47	2.04	-	-	-
Mg			26.33	25.54	2.15	-	-	-
Li	µg/g	1	26.33	25.54	2.15	-	-	-
Mg	pct	0.01	0.04	0.04	1.54	-	-	-
Mn	µg/g	1	414.24	405.30	1.54	-	-	-
Mo	µg/g	1	5.00	4.00	15.71	-	-	-
Na	pct	0.01	0.04	0.05	10.12	-	-	-
Na	µg/g		407.03	469.74	10.12	-	-	-
Nb	µg/g	5	<5	<5	-	-	-	-
Ni	µg/g	1	50.00	49.59	1.58	-	-	-
Pb	µg/g	1	64.19	64.63	0.48	-	-	-
Sb	µg/g	5	<5	<5	-	-	-	-
Sn	µg/g	20	<20	<20	-	-	-	-
Sr	µg/g	1	148.80	145.60	1.53	-	-	-
Ti	µg/g	1	227.00	214.52	4.00	-	-	-
V	µg/g	25	15.81	15.88	0.29	-	-	-
W	µg/g	20	<20	<20	-	-	-	-
Y	µg/g	1	10.79	10.52	1.80	-	-	-
Zn	µg/g	1	926.22	890.04	2.82	-	-	-
Aluminum	pct	0.01	-	-	-	13.22	13.14	0.44
Barium	pct	0.001	-	-	-	0.05	0.04	0.51
Calcium	pct	0.01	-	-	-	2.91	2.86	1.15
Chromium	pct	0.01	-	-	-	0.02	0.02	6.80
Iron	pct	0.01	-	-	-	6.10	6.06	0.47
Potassium	pct	0.01	-	-	-	2.79	2.67	3.11
Magnesium	pct	0.01	-	-	-	2.53	2.51	0.41
Manganese	pct	0.01	-	-	-	0.04	0.04	1.89
Sodium	pct	0.01	-	-	-	0.66	0.66	0.28
Phosphorus	pct	0.03	-	-	-	0.30	0.34	7.30
Silicon	pct	0.01	-	-	-	50.39	49.96	0.60
Titanium	pct	0.01	-	-	-	0.69	0.69	0.62
Loss on	pct	0.05	-	-	-	20.60	21.20	2.03
Whole Rock	pct	-	-	-	-	100.24	100.13	0.07

Table G3. Matrix spike and reference standard results (Caduceon).

Analyte	% Recovery
Ag	102
As	100
Cd	100
Co	94
Cr	95
Cu	100
Fe	89
Mn	98
Mo	106
Ni	109
Pb	100
V	91
Zn	101
Aluminum	98
Barium	100
Calcium	99
Chromium	100
Iron	100
Potassium	94
Magnesium	99
Manganese	100
Sodium	98
Phosphorus	106
Silicon	103
Titanium	96
Loss on	96
Whole Rock	101
Mean	99

Reference Material	Expected Total Hg (ug/g)	Measured Total Hg (ug/g)	% Recovery
STSD-2	46	51	111
STSD-2	46	49	107
STSD-2	46	51	111
STSD-2	46	44	96
STSD-2	46	43	93
STSD-2	46	43	93
STSD-4	930	865	93
STSD-4	930	867	93
STSD-4	930	1010	109
STSD-4	930	876	94
STSD-1	110	115	105
STSD-1	110	104	95
STSD-1	110	117	106
STSD-1	110	117	106
		Mean	**101**

Table G4. Matrix spike recoveries for sediment samples (MOE Laboratory).

Site	d10-phenanthrene	d12-chrysene	d8-naphthalene
LC01	100	47	63
LC03	140	140	67
LC12	78	37	33
LC16	72	36	23
LC17	93	53	49
LC29-1	110	84	90
LC29-2	110	58	58
LC29-3	120	85	91
LC38	95	54	55
BLC01	140	98	100
BLC02	120	78	120
LC06	86	47	53
LC08-1	86	57	46
LC08-2	97	61	42
LC10	89	53	42
LC14	91	61	85
LC16	93	62	83
LC18-1	87	61	85
LC18-2	91	65	44
LC19	88	62	39
LC22-1	89	62	48
LC22-2	85	67	49
LC23	97	64	40
LC29	89	56	48
LC38	92	67	48
TC40	100	67	58
UC01	86	62	52
Mean	**97**	**65**	**60**

Table G5. Matrix spike recoveries for 2002 biota samples (MOE Laboratory).

Site	Taxa	d10-phenanthrene	d12-chrysene	d8-naphthalene
BLC01	CHIR	100	68	78
BLC01	AMP	87	68	67
BLC01	ODON	110	92	83
BLC02	CHIR	110	59	57
BLC02	AMP	100	67	65
BLC02	ODON	110	81	66
LC01	AMP	93	69	66
LC01	OLIG	93	70	41
LC01	ODON	97	88	85
LC03	AMP	88	52	61
LC03	OLIG	97	63	84
LC03	ODON	96	65	69
LC12	CHIR	110	76	75
LC12	AMP	87	60	59
LC12	OLIG	100	76	64
LC12	ODON	98	75	64
LC17	CHIR	92	64	60
LC17	AMP	93	0	52
LC17	OLIG	120	110	94
LC17	ODON	91	57	51
	Mean	**99**	**68**	**67**

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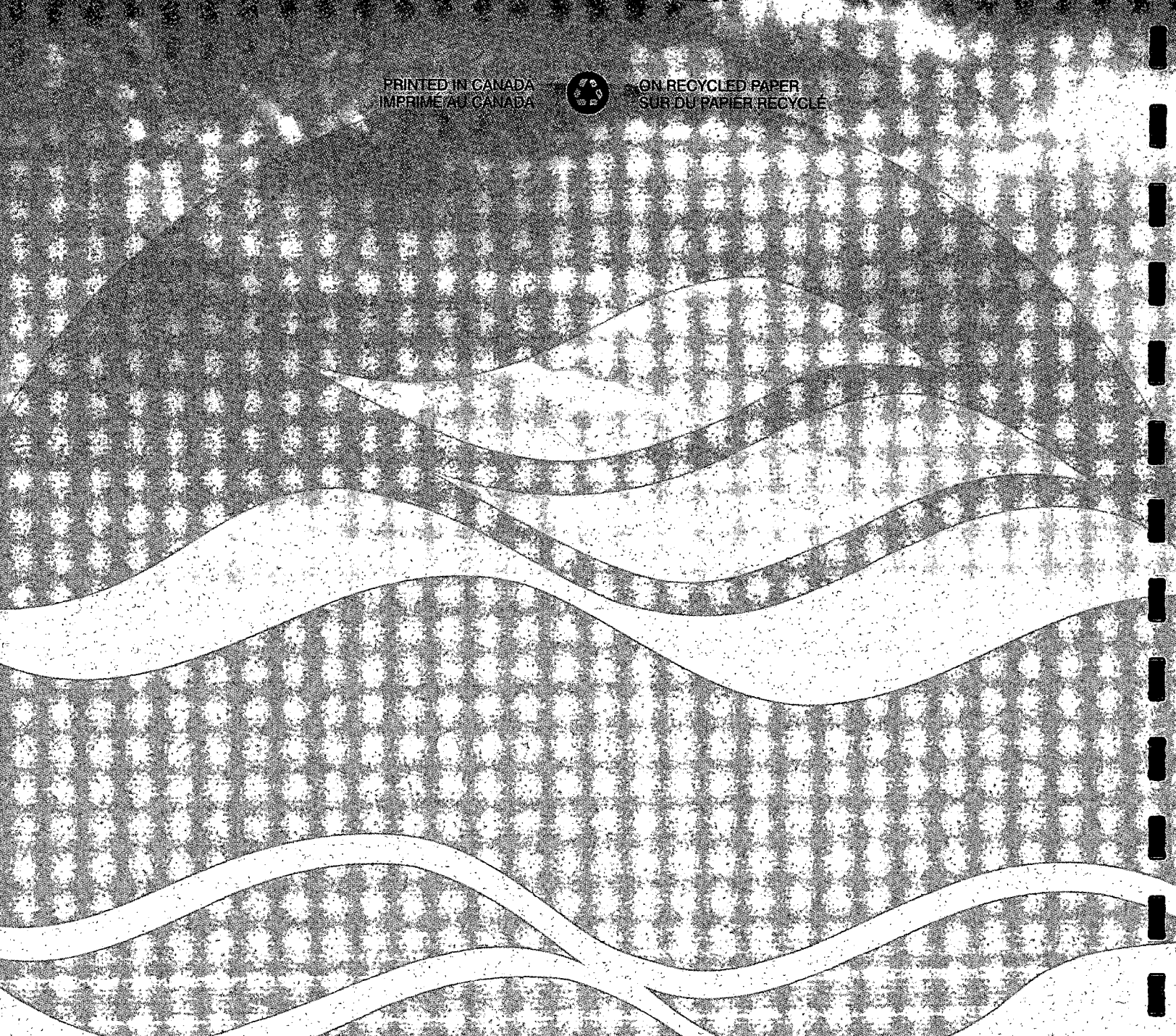

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