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Species richness and species occurrence of five taxonomic groups in relation to pH and other lake characteristics in southeastern Canada

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Species richness and species occurrence of five taxonomic groups in relation to pH and other lake characteristics in southeastern Canada

by

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Abstract

Biotic damage to aquatic ecosystems is among the most recognizable, deleterious effects of acidic precipitation in southeastern Canada. Using recent databases from a variety of sources (> 2000 water bodies), we determine the pH range of occurrence and regional pH minima for various aquatic biota (zooplankton, macroinvertebrates, fish, amphibians and waterbirds). We demonstrate that the proportional occurrence of aquatic taxa is highest near pH 6. In addition, we develop multivariate logistic and linear regression models to predict the occurrence and taxonomic richness of certain broadly-distributed taxa for use in current national acid rain modelling efforts. Our results confirm that pH 6 is an important threshold below which damage to aquatic biota occurs, and, more importantly, that acidification remains a major local and regional environmental problem for biodiversity across trophic levels.

Résumé

Les dommages biotiques causés aux écosystèmes aquatiques sont parmi les effets les plus évidents et délétères des précipitations acides dans l'est du Canada. En utilisant des bases de données récentes provenant d'une variété de sources (>2 000 lacs), nous déterminons la gamme de pH d'occurence et les pH régionaux minimums pour divers biotes aquatiques (zooplancton, macroinvertebrates, poissons, amphibiens et oiseaux aquatiques). Nous démontrons que l'occurence proportionnelle des taxons aquatiques est la plus élevée autour de pH 6. De plus, nous élaborons des modèles logistiques multivariés et de régression linéaire afin de prévoir l'occurence et la richesse taxinomique de certains taxons largement répartis, pour utilisation dans les efforts nationaux actuels de modélisation des précipitations acides. Nos résultats confirment que pH 6 constitue un seuil important au-dessous duquel des dommages aux biotes aquatiques apparaissent, et, ce qui est encore plus important, que l'acidification est encore un problème environnemental local et régional majeur en ce qui a trait à la biodiversité dans les différents niveaux trophiques.

1.0 General Introduction

Acidic precipitation (usually referred to as acid rain, but can also include dry deposition) is an environmental stressor caused by emissions of atmospheric pollutants such as sulphur dioxide (SO_2) and nitrogen oxides (NO_x). These may be transported vast distances by prevailing winds before being deposited as sulphuric or nitric acid. Over time, continued deposition of these acids on watersheds with poor buffering capacity (typical of those in much of southeastern Canada) has led to altered chemistries in soils, streams, rivers and lakes (RMCC 1990). For aquatic biota, reductions in the pH and acid-neutralizing capacity (alkalinity) of water and the associated increases in toxic trace metal concentrations have caused dramatic changes in the distribution, abundance and reproductive success of many taxa (reviewed in Schindler 1988, Schindler *et al.* 1989, Baker *et al.* 1990, RMCC 1990).

Acidic precipitation has been documented for more than a century (see Cowling 1982), but research efforts on the effects of this pollutant in southeastern Canada effectively started in the 1970s. Once the potential magnitude of the problem was recognized, scientific study of the effects of acidification on aquatic ecosystems, as well as efforts to reduce the threat from acid rain, accelerated (see RMCC 1990, Table 1 in McNicol *et al.* 1995a). In 1990, the LRTAP Assessment Report on Aquatic Effects, which included biotic effects, (hereafter the 1990 Assessment) was prepared as an up-to-date scientific technical document to support the negotiation of the bilateral Canada/US Air Quality Agreement (a similar document was prepared in the United States; see Baker *et al.* 1990). The 1990 Assessment summarized the pertinent results from intensive monitoring and research programs through the late 1970s and especially the 1980s, (particularly those in Canada), and identified those areas where knowledge gaps still existed. Two important conclusions of the 1990 Assessment were relevant to the present study:

1. Deposition of atmospheric sulphate (SO_4) was the principle cause of regional acidification in southeastern Canada, with the most sensitive and/or affected regions falling on the Canadian Shield.

2. Species richness and diversity of aquatic biota (particularly fish, plankton and some macroinvertebrates) began to decline in water bodies below pH 6.0.

Similarly, two key knowledge gaps that were identified in that document that pertain to this report were:

1. Information was lacking on the effects of acidification on the chemistry and biology of small water bodies (defined as wetlands to small lakes < 100 ha).

2. Further refinements were required to predict the effects of sulphur and nitrogen deposition on aquatic biota (RMCC 1990).

In 1995, a similar National Assessment project was initiated to reassess the effects of acid rain on aquatic ecosystems in southeastern Canada, using the knowledge base developed in the 1990 Assessment and focusing on what new information has been uncovered since that document was produced and to be incorporated into a new assessment document. The "1997

Assessment" was developed to serve as a technical support document to the long-term strategy on acid rain under the "Statement of Intent on Long-term Acid Rain Management in Canada" signed by the Federal/Provincial/Territorial Ministers of Environment and Energy in November 1994. Again, this information would be used to re-evaluate emission standards under the bilateral Canada/US Air Quality Agreement. As part of the 1997 Assessment, an evaluation of the new information (since 1990 or that unavailable for the 1990 Assessment) on the occurrence of various aquatic biota in relation to water chemistry was initiated. This technical report is a companion report to the 1997 Assessment and a product of that evaluation.

1.1 Scope

The work presented in this report covers a broad range of aquatic or semi-aquatic taxonomic groups in southeastern Canada, namely: zooplankton, macroinvertebrates, fish, amphibians, and waterbirds (in this report waterbirds are defined as seven waterfowl species and common loons). Data was supplied and collected by various provincial and federal agencies within this region, but in some cases does not encompass all of southeastern Canada. We would like to emphasize that the applicability of the models is relatively confined to the latitude and longitude ranges specified in the species richness and species occurrence models. However, previous studies have presented similar relationships to the ones depicted here and therefore the models may apply in other regions, like the Appalachian Mountain region in the United States (Reckhow 1988, Seigfried 1988). The datasets span from 1970 to 1995 but the majority of data were collected from 1985 to 1995 (see Section 2.0). The models presented here were generated for the 1997 Assessment and therefore did not use any data which had previously been used in other Assessments in order to establish new relationships or to verify existing ones. This tended to restrict the analysis spatially as well as functionally since not very many exhaustive datasets were available for southeastern Canada that covered large spatial areas, collected complete community data for all of the biological groups covered in this report, or sampled a complete array of chemical and physical parameters concurrently with biological data. Therefore, the chemical and physical attributes used in statistical analyses to test empirical relationships with species richness and occurrence were limited. (See Appendix 1 for list of variables.)

1.2 Rationale

Why do we need biological models? Firstly, and perhaps foremost, the public concern for the acid rain problem has manifested due to observed negative effects on aquatic biota which were subsequently linked to lake chemistry changes. It is the loss of fish (notably sportfish) or wildlife (especially the common loon) that triggered much of the initial and continued concern, and therefore models of responses of these organisms provide answers that have a meaningful component for scientists and the community at large. Secondly, biological models provide a final step in linking the effects of changes in lake pH and the ultimate effects of those changes on the aquatic ecosystem. It is important to recognize and incorporate other factors that affect biological diversity and not simply to estimate an overarching and general pH threshold that should be applied to all systems (for recent examples see Hämäläinen and Huttunen 1996, Logie *et al.* 1996). Finally, there is a need to continually assess the extent of biotic damage due to anthropogenic acidification, to monitor biotic communities after acidification and to predict improvements to these communities as a result of changes to emission standards across North America. It is important to recognize too that biological recovery usually lags behind chemical recovery and under certain situations may not even be possible. It is because of this disparity in recovery time lines that monitoring recovery as well as predicting recovery from biotic models are equally important.

1.3 Objectives

There were three objectives for this report on the biological effects of the long-range transport of airborne pollutants (LRTAP) or acid rain in aquatic ecosystems. Specifically, these were:

1. To update the existing database on aquatic biota, lake chemistry, and morphometry, that was available for acid rain assessments.

2. Develop new statistical models, and reassess existing models, for aquatic biota and the effects of pH changes by using a univariate (pH-only) and multivariate approach (single species and species richness or community models). A multivariate approach recognizes the importance of other physico-chemical factors and their effects on biota.

3. Provide information to the Integrated Assessment Model (IAM) work group to be used in scenario-testing for establishing acidic emission thresholds for the 1997 Assessment of acidic precipitation in southeastern Canada. The IAM has incorporated several of the models presented in this report into a pre-existing framework of integrated emission, deposition, hydrological and geochemical models. Various emission-deposition scenarios were tested using the integrated models to predict any subsequent changes in biotic integrity (species richness or range of occurrence changes). The results of several gaming and scenario testing will be reported elsewhere.

The following sections outline the approach that was taken in compiling and analyzing data from various sources (Section 2.0), and present three types of analyses and models which were used to determine the effects of pH and other factors on aquatic biota of southeastern Canada. Initially we present pH ranges for specific species and groups (Section 3.2), and then using these ranges we develop a theoretical species richness (or proportional response) distribution for several taxonomic groups (Section 3.3). Using multivariate approaches, we initially examine the important factors which affect distributions of certain species (Section 4.2), and finally we present information on how species richness (or the number of species in a taxonomic group) is affected by key factors (Section 4.3).

2.0 Datasets

The datasets that were used in the generation of the models presented in this report were generously supplied by various individuals and agencies (see Acknowledgements) in southeastern Canada. The list of contributors, the number of water bodies, the region sampled and the time frame of sampling are listed in Table 1. The datasets usually combined either presence / absence data or abundance data for a particular taxonomic group along with a suite of environmental parameters for the water bodies that were censused. A complete list of the

Table 1: List of data sources, codes, contributors, years, number of water bodies, and regions sampled for all the datasets used in statistical and modelling efforts.

Dataset Contributor	Dataset Code	Source / Contact	Sampling Years	# water-bodies	Region
Fisheries Data					
National Inventory Survey	NIS	Various federal and provincial agencies (see Kelso et al. 1986)	1981-1982	400	Southeastern Canada (excluding Ontario)
Ministère de l'Environnement du Québec	QLS	Québec Lake Survey -- J. Dupont (Dupont 1993)	1985-1990	250	Southern Québec
Ontario Mesoscale Study	MESO	Various Ontario ministries (see Kelso and Johnson 1986)	1982-1984	181	Central Ontario
CWS (Ontario) LRTAP Biomonitoring Program	CWS	Canadian Wildlife Service -- D. McNicol (McNicol *et al.* 1996)	1991-1994	590	Northeastern & Central Ontario
Fisheries and Oceans (Atlantic) & CWS (Atlantic)	DFO / CWS	DFO fisheries data and CWS chemistry data (Alexander *et al.* 1986, McNicol *et al.* 1996)	1971-1981, 1996	29	Northern and Southern Nova Scotia
Ontario Ministry of Natural Resources	OMNR	J. Gunn (partially from Gunn *et al.* 1988)	1983-1986, 1995	54	Central Ontario
Fisheries and Oceans (Atlantic)	DFO	W. White, (unpublished data)	1983-1987	43	Nova Scotia
Zooplankton Data					
Ontario Ministry of Environment and Energy	OMEE	B. Keller (Keller and Conlon 1994)	1981-1986	578	Northwestern and Central Ontario
University of Toronto	UofT	W.G. Sprules (Locke and Sprules 1993)	1981-1994	45	Central Ontario
Université de Montréal	UofM	B. Pinel-Alloul (from Pinel-Alloul *et al.*, 1990)	1981-1982	54	Southern Québec
Newfoundland Department of Environment and Lands	DEL	(from Blouin, 1989)	1981-1984	20	Nova Scotia
National Inventory Survey	NIS	Various federal and provincial agencies (See Kelso et al. 1986)	1981-1982	261	Southeastern Canada
Fisheries and Oceans (Atlantic Region)	DFO	W. White (from Strong, 1986)	1983-1984	36	Nova Scotia

Table 1 continued.

Dataset Contributor	Dataset Code	Source / Contact	Sampling Years	# water-bodies	Region
Macroinvertebrate Data					
CWS (Ontario) LRTAP Biomonitoring Program	CWS	Canadian Wildlife Service -- D. McNicol (McNicol *et al.* 1996)	1991-1994	66	Northeastern & Central Ontario
Wildlife Data (Amphibians and Waterbirds)					
CWS (Ontario) LRTAP Biomonitoring Program	CWS	Canadian Wildlife Service -- D. McNicol (McNicol *et al.* 1996)	1988-1994	629	Northeastern & Central Ontario
CWS (Atlantic) LRTAP Biomonitoring Program	CWS	Canadian Wildlife Service – J. Kerekes (McNicol *et al.* 1996)	1988-1995	46	Southwest Nova Scotia
Ontario Lakes Loon Survey	OLLS	Canadian Wildlife Service -- D. McNicol (data supplied by Long Point Bird Observatory)	1981-1995	821	Ontario

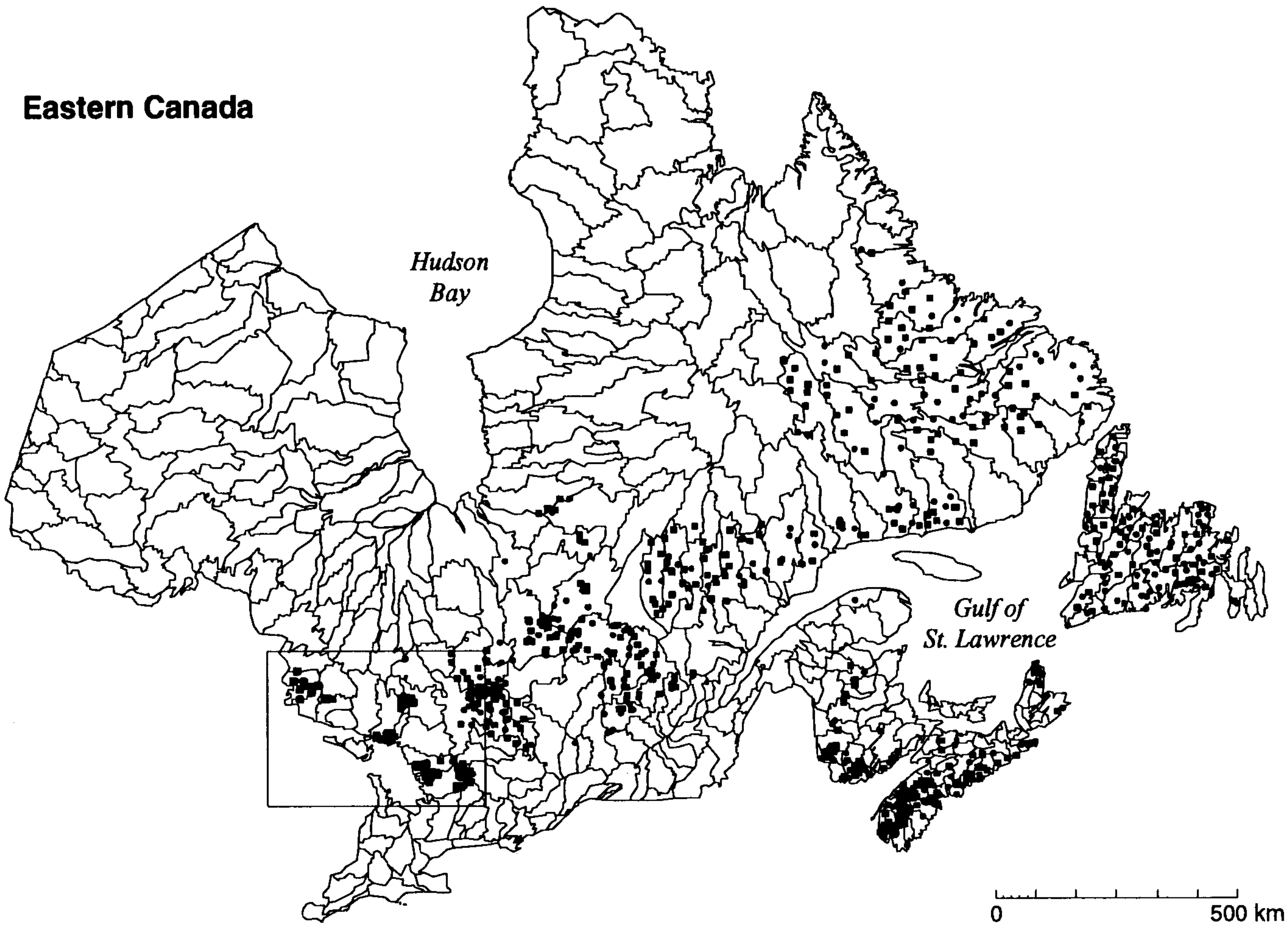

Figure 1A: Map of the complete set of water bodies compiled for the fish database used in this report for statistical and modelling efforts. All of the amphibian, macroinvertebrate and waterbird data that were supplied are encompassed by the boxed region on the map and overlap with the fish data supplied for the same region. This region does not include Québec. The boundaries indicate tertiary watershed limits. (Not all water bodies could be used in each of the different analyses. See Table 1 for a list of sources and regions covered by each individual dataset.)

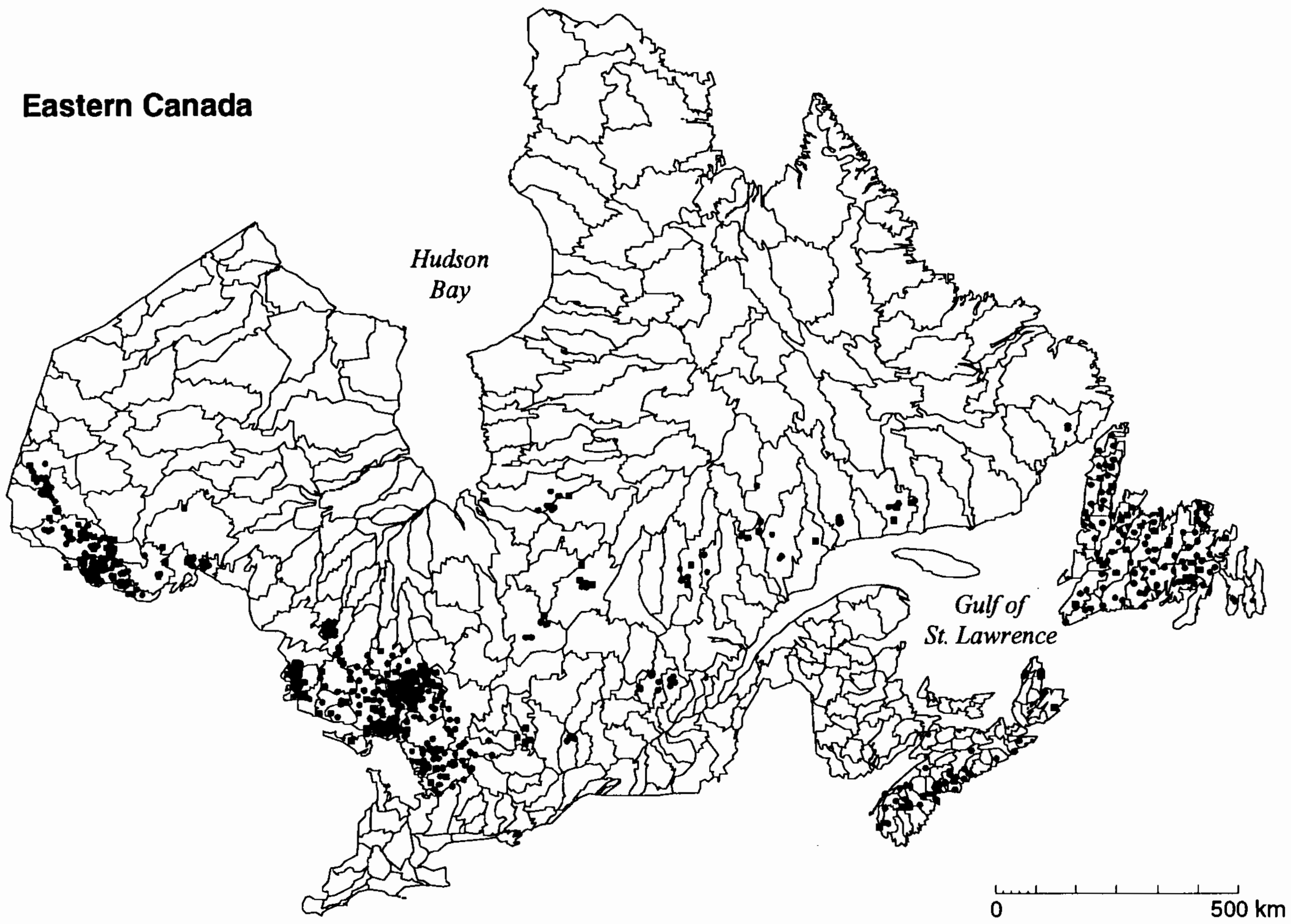

Figure 1B: Map of the complete set of water bodies compiled for the zooplankton database used in this report for statistical and modelling efforts. The boundaries indicate tertiary watershed limits. (Not all water bodies could be used in each of the different analyses. See Table 1 for a list of sources and regions covered by each individual dataset.)

variables (chemical and physical) that were supplied in each dataset, along with averages and standard deviations are listed in Appendix 1. Full details of dataset organization and collection methods are found in the references listed in Table 1. For reasons that are explained in later sections, certain variables needed to be derived from other highly correlated variables in the datasets provided. Dissolved organic carbon (DOC) was chosen as the variable of choice for representing coloured lake systems; therefore where DOC data was not supplied the values were derived by using linear relationships between DOC and either lake colour or secchi depth (The equations used for conversions are shown in Appendix 2).

Data were supplied from various regions across southeastern Canada, the maps in Figure 1A and B show the distribution of the complete set of water bodies used in modelling. The spatial area covered by water bodies for each taxonomic group should be taken into consideration when interpreting the results of modelling exercises, particularly the macro-invertebrate and waterbird models that are an accurate representation of small water bodies in central and northeastern Ontario but might not be applicable outside of this geographic location until further validation (models for loons are for larger lakes and are not regionally restricted).

3.0 Lake pH and species relationships

3.1 Introduction

Lake water acidity (hydrogen ion concentration or pH) is a key chemical characteristic of water bodies. It is easily measured and is highly correlated with many other chemical parameters (especially concentrations of base cations and trace metals), and influences the occurrence of many biota (see reviews in Schindler 1988, Schindler *et al.* 1989, Longcore *et al.* 1993). Early research on the relationship between anthropogenic acidification of water bodies and aquatic biota focused on determining the pH tolerance limits for reproduction and survival of specific groups of organisms, and the distribution of measured or expected damage to biota (Beamish 1974, Schofield 1976, Schindler and Turner 1982). Studies demonstrated that some taxa are relatively insensitive to reduced pH and are found across a broad pH range [For example, the rotifers *Keratella* spp. (Shaw and Kelso 1992), the dragonfly *Leucorrhinia glacialis* (McNicol *et al.* 1987b), yellow perch *Perca flavescens* (Matuszek and Beggs 1988, Swenson *et al.* 1989), common goldeneye *Bucephala clangula* (McNicol *et al.* 1987a, DesGranges and Houde 1989)]. Other species are affected deleteriously by direct (water toxicity) or indirect effects (habitat or predator/prey changes) when pH decreases [e.g. the amphipod *Hyallela azteca* (Stephenson and Mackie 1986), certain mayfly species Ephemeroptera (Allard and Moreau 1987), certain crayfish *Orconectes* spp. (France 1987), lake trout *Salvelinus namaycush* (Reckhow 1988), spotted salamander *Ambystoma maculatum* (Clark 1986a), common loons *Gavia immer* (McNicol *et al.* 1987a, Alvo *et al.* 1988, DesGranges and Houde 1989); see also extensive reviews in Schindler 1988, Schindler *et al.* 1989, Baker *et al.* 1990, RMCC 1990, Longcore *et al.* 1993]. Although some species can tolerate or even thrive at low pHs, usually species diversity is higher, and fecundity and breeding densities are greater at higher pH levels [e.g. microbial communities (reviewed in Baker *et al.* 1990), zooplankton (Keller and Pitblado 1984), snails (Økland and Økland 1986, Shaw and Mackie 1990), certain crayfish (Mierle *et al.* 1986, Davies 1989), fish and macroinvertebrates (Weatherley *et al.* 1988, Gunn and Keller 1990, Baker *et al.* 1990), amphibians (Clark 1986b, Glooschenko *et al.* 1992), waterbirds and loons (McNicol *et al.* 1987a, 1990)]. Consequently, aquatic food webs in acid-stressed water bodies usually become simpler (i.e. biodiversity is reduced due to the loss of acid-sensitive species), even though biomass may remain constant. Depending on the magnitude and types of species lost from a lake, predators (e.g. fish, amphibians, aquatic birds and mammals) that rely on these prey may also be deleteriously affected indirectly (see Longcore *et al.* 1993).

Studies conducted up to 1990 suggested that biotic degradation of aquatic ecosystems in North America and Europe occurred when acidic deposition caused the pH of water bodies to decline below pH 6.0 (Rago and Wiener 1986, Matuszek and Beggs 1988, Schindler *et al.* 1989, Baker *et al.* 1990, RMCC 1990). Information gathered since 1990 has confirmed the deleterious effects of acidification on aquatic systems below pH 6 and has provided even stronger documentation of the worldwide magnitude of the effects of acidic precipitation (see reviews in Muniz 1991, Appelberg *et al.* 1993, Stenson *et al.* 1993, Herrmann *et al.* 1993, Longcore *et al.* 1993, Havas and Rosseland 1995, Raddum and Skjelkvale 1995, Rask *et al.* 1995).

3.2 Species pH ranges

Research to date has suggested that certain taxa may have specific tolerance limits to pH below which they cannot reproduce or survive (e.g. Gosner and Black 1957, Eilers *et al.* 1984, Stephenson and Mackie 1986, Shaw and Mackie 1990, McNicol *et al.* 1996, and many others). These types of taxon-specific pH ranges may be useful tools in developing single species or community models to assess biological damage and potential recovery of watersheds from acidification (see Eilers *et al.* 1984, Baker *et al.* 1988, Schindler *et al.* 1989, Minns *et al.* 1990, Conlon *et al.* 1992, Matuszek *et al.* 1992, Nicholls *et al.* 1992, Hämäläinen and Huttunen 1996, Logie *et al.* 1996). We have used data collected from a variety of sources across southeastern Canada to examine the occurrence of various biota in relation to lake pH, to determine whether pH minima (the lowest pH at which a species has been recorded in our data) are similar across regions, and to look at the collective occurrence of species across the pH scale. Hereafter, we define organisms with a 'low pH minima' as those that are found at relatively lower pHs than most organisms, whereas those organisms with a 'high pH minima' have a minimum recorded pH that are higher than most species.

Species-specific pH ranges are listed for selected aquatic and semi-aquatic taxa in Table 2. This table represents data collected from water bodies in southeastern Canada between 1970 and 1995 (with the majority of sampling between 1985-1995, see Table 1). Taxa were sampled in water bodies with pHs that ranged between 3.8 to 9.8 but 95% of the water bodies where the various taxa were present had a pH > 5.1 (Appendix 1).

It is clear from Table 2 that the minimum pH of occurrence varies widely among taxa. However, some broad patterns do appear. Figure 2 shows the range for the average pHs over which different species within the five taxonomic groups (zooplankton, macroinvertebrates, fish, amphibians, waterbirds) occur. For example, many of the larger, semi-aquatic species such as waterbirds and amphibians, have low pH minima (approximately ≥ pH 4.2), whereas some of the zooplankton, macroinvertebrates and fish were not found below pH 5.0. This is not surprising, since mobile organisms (i.e. not restricted to the aquatic environment within a lake) may be able to exploit prey resources or suitable habitat conditions in acid-stressed water bodies while using other sites for breeding, in contrast to wholly aquatic organisms that may

Table 2: Taxonomic list of species pH ranges for the datasets used in this report. The number of water bodies (#) in which the species were recorded, the mean pH, standard deviation (± SD), median pH, and minimum and maximum pH value at which these species occurred are shown. (Zooplankton and fish statistics are from southeastern Canada data and amphibian, macroinvertebrate and waterbird statistics apply to small lakes in Ontario.)

TAXA	Common Name	Species Name	#	Mean pH	± SD	Median pH	Min pH	Max pH
ZOOPLANKTON								
Cladocera	Water Fleas	*Eubosmina tubicen*	64	5.83	0.61	5.78	4.54	6.95
		Polyphemus pediculus	58	5.87	0.97	5.79	4.26	7.35
		Latona setifera	19	5.66	0.60	5.80	4.63	6.54
		Daphnia ambigua	57	5.88	0.59	5.88	4.50	6.99
		Daphnia catawba	165	5.96	0.65	5.96	4.30	7.77
		Eubosmina longispina	198	6.12	0.77	6.15	4.26	8.11
		Diaphanosoma brachyurum	141	6.09	0.79	6.19	4.26	7.45
		Daphnia pulex	184	6.19	0.84	6.38	4.21	8.25
		Leptodora kindtii	64	6.36	0.59	6.43	4.79	7.44
		Holopedium gibberum	451	6.31	0.75	6.46	4.38	8.36
		Bosmina longirostris *	567	6.30	0.81	6.48	4.21	8.36
		Daphnia dubia	90	6.57	0.57	6.62	4.62	7.68
		Diaphanosoma birgei *	280	6.51	0.62	6.62	4.58	8.11
		Sida crystallina	32	6.43	0.79	6.63	4.73	8.00
		Daphnia longiremis	177	6.60	0.61	6.68	4.30	8.11
		Latonopsis fasciculata	10	6.52	0.46	6.69	5.75	7.15
		Daphnia galeata mendotae	386	6.63	0.65	6.72	4.30	8.36
		Daphnia retrocurva	239	6.64	0.56	6.72	4.59	8.11
		Chydorus sphaericus	204	6.58	0.71	6.75	4.13	7.75
Copepoda		*Ceriodaphnia lacustris*	83	6.77	0.56	6.81	4.90	7.80
		Limnocalanus macrurus	94	5.85	0.70	5.79	4.60	7.20
		Leptodiaptomus minutus	793	6.20	0.80	6.33	4.21	8.36
		Orthocyclops modestus	55	6.10	0.79	6.39	4.48	7.40
		Cyclops scutifer	216	6.31	0.72	6.40	4.54	8.30
		Mesocyclops edax	447	6.33	0.78	6.49	4.26	8.36
		Tropocyclops extensus	342	6.37	0.70	6.56	4.60	8.30
		Skistodiaptomus oregonensis	308	6.61	0.61	6.70	4.30	8.25

TAXA	Common Name	Species Name	#	Mean pH	± SD	Median pH	Min pH	Max pH
Copepods cont'd		*Diacyclops thomasi*	419	6.60	0.63	6.70	4.30	8.30
		Epischura lacustris	217	6.69	0.51	6.71	5.20	8.36
		Acanthocyclops vernalis	91	6.67	0.61	6.81	4.29	7.68
		Leptodiaptomus sicilis	65	6.81	0.44	6.82	5.20	8.20
Rotifers		*Keratella taurocephala*	66	5.97	0.69	6.10	4.56	7.30
		Polyarthra major	18	6.08	0.46	6.16	5.25	6.78
		Collotheca mutabilis	30	6.28	0.58	6.18	5.43	7.80
		Gastropus stylifer	83	6.19	0.59	6.18	4.84	7.77
		Polyarthra euryptera	45	6.20	0.63	6.22	4.90	7.70
		Polyarthra dolichoptera	32	6.30	0.84	6.24	4.84	7.96
		Pleosoma truncatum	36	6.37	0.63	6.26	5.41	7.96
		Conochilus unicornis	138	6.28	0.61	6.26	4.84	7.96
		Kellicottia longispina	262	6.29	0.69	6.29	4.30	8.30
		Keratella cochlearis spp.	291	6.27	0.70	6.30	4.30	8.30
		Filinia longiseta	35	6.37	0.80	6.40	4.90	8.30
		Asplanchna priodonta	79	6.48	0.74	6.40	5.00	8.30
		Keratella quadrata	70	6.35	0.75	6.40	4.30	8.30
		Polyarthra vulgaris	141	6.44	0.69	6.40	4.71	8.30
		Trichocerca cylindrica	123	6.39	0.61	6.40	4.71	8.00
		Kellicottia bostoniensis	123	6.37	0.68	6.50	4.30	7.70
		Trichocerca multicrinnis	41	6.45	0.81	6.50	4.90	8.00
		Ascomorpha saltans	22	6.53	0.61	6.60	5.30	7.70
MACROINVERTEBRATES								
Amphipoda	scuds	*Crangonyx richmondensis*	36	5.67	0.49	5.58	5.02	6.98
		Hyallela azteca	20	6.27	0.48	6.24	5.22	7.02
Coleoptera	diving & water	*Acilius athabascae*	5	6.12	0.58	6.06	5.26	6.79
	beetles	*Acilius semisulcatus*	4	6.17	0.61	6.26	5.38	6.79

TAXA	Common Name	Species Name	#	Mean pH	± SD	Median pH	Min pH	Max pH
Coleoptera cont'd		*Dytiscus alaskanus*	15	5.66	0.59	5.44	5.10	6.98
		Dytiscus verticalis	8	5.41	0.32	5.42	5.02	5.94
		Graphoderus fasciatocollis	9	5.74	0.42	5.79	5.02	6.32
		Graphoderus liberus	36	5.58	0.54	5.46	4.51	6.98
		Graphoderus perplexus	27	5.70	0.56	5.57	4.51	6.98
		Haliplus cribrarius	6	6.09	0.46	6.08	5.48	6.88
Decapoda	crayfish	*Orconectes propinquus*	4	6.38	0.27	6.37	6.12	6.66
Diptera	midges	*Chaoborus americanus*	22	5.57	0.63	5.39	4.51	6.98
Gastropoda	snails	*Gyraulus deflectus*	5	6.75	0.31	6.88	6.30	7.02
		Helisoma anceps	7	6.21	0.40	6.05	5.81	7.02
Hemiptera	back swimmers	*Notonecta borealis*	17	5.73	0.64	5.66	4.51	6.98
		Notonecta insulata	28	5.55	0.60	5.42	4.25	6.98
		Notonecta undulata	21	5.43	0.61	5.33	4.25	6.98
	giant water bug	*Lethocerus americanus*	26	5.60	0.56	5.50	4.39	7.02
	water boatman	*Dasycorixa hybrida*	5	5.56	0.28	5.66	5.21	5.90
		Sigara compressoidea	8	5.95	0.27	6.04	5.57	6.32
		Sigara decoratella	9	5.50	0.50	5.33	4.78	6.30
		Sigara defecta	10	5.45	0.70	5.20	4.78	6.98
		Sigara dolabra	5	5.41	0.45	5.40	4.78	6.05
		Sigara mackinacensis	14	5.99	0.45	6.04	5.22	6.88
		Sigara macropala	24	5.51	0.44	5.39	4.78	6.47
		Sigara penniensis	17	5.92	0.51	5.87	5.15	6.98
		Sigara solensis	5	5.78	0.32	5.68	5.48	6.32
	water strider	*Gerris comatus*	18	5.60	0.54	5.68	4.39	6.88
		Rheumatobates rileyi	5	5.66	0.57	5.80	5.02	6.46
		Trepobates inermis	6	5.67	0.51	5.74	5.02	6.46

TAXA	Common Name	Species Name	#	Mean pH	± SD	Median pH	Min pH	Max pH
Hirudinea	leeches	*Batracobdella picta*	23	6.03	0.45	6.05	5.22	6.98
		Erpobdella punctata	29	5.99	0.52	6.03	5.15	7.02
		Glossiphonia complanata	11	6.19	0.41	6.14	5.48	6.98
		Helobdella stagnalis	22	6.05	0.44	6.05	5.26	6.98
		Macrobdella decora	38	5.91	0.48	5.93	5.02	7.02
		Mollibdella grandis	17	6.09	0.53	6.06	5.38	7.02
		Mooreobdella fervida	8	5.68	0.31	5.67	5.33	6.16
		Nephelopsis obscura	35	5.91	0.50	5.87	5.10	7.02
		Percymoorensis marmoratis	54	5.75	0.53	5.70	4.78	7.02
		Placobdella ornata	11	6.26	0.50	6.03	5.68	7.02
Odonata (Anisoptera)	dragonflies	*Aeshna eremita*	31	5.69	0.60	5.69	4.39	6.98
		Aeshna interrupta	22	5.64	0.62	5.47	4.51	6.98
		Aeshna umbrosa	8	5.62	0.77	5.41	4.39	6.88
		Cordulia shurtleffi	54	5.72	0.63	5.69	4.25	7.02
		Didymops transversa	7	6.07	0.50	5.94	5.22	6.60
		Leucorrhinia frigida	4	5.70	0.59	5.77	4.93	6.32
		Leucorrhinia glacialis	53	5.62	0.60	5.50	4.25	7.02
		Leucorrhinia intacta	15	6.08	0.53	6.03	5.15	7.02
		Libellula julia	60	5.76	0.54	5.70	4.78	7.02
		Somatochlora cingulata	4	6.11	0.74	6.10	5.22	7.02
		Somatochlora walshii	4	6.27	0.39	6.19	5.90	6.79
		Somatochlora williamsoni	4	6.31	0.39	6.14	6.06	6.88
Trichoptera	caddisflies	*Agrypnia improba*	6	5.64	0.47	5.58	5.09	6.30
		Agrypnia straminea	10	5.74	0.39	5.80	4.93	6.17
		Banksiola crotchi	29	5.63	0.58	5.49	4.39	7.02
		Banksiola smithi	34	5.47	0.41	5.42	4.51	6.30
		Platycentropus amicus	45	5.63	0.55	5.49	4.39	7.02

TAXA	Common Name	Species Name	#	Mean pH	± SD	Median pH	Min pH	Max pH
FISH								
	brown bullhead	*Ameiurus nebulosus*	134	5.72	0.55	5.69	4.40	7.13
	golden shiner	*Notemigonus crysoleucas*	133	5.78	0.48	5.75	4.40	7.00
	yellow perch	*Perca flavescens*	455	5.78	0.60	5.81	4.23	7.27
	pumpkinseed	*Lepomis gibbosus*	171	5.83	0.51	5.81	4.57	7.53
	largemouth bass	*Micropterus salmoides*	52	5.81	0.48	5.83	4.57	7.24
	pike	*Esox lucius*	170	5.88	0.48	5.90	4.85	7.11
	fallfish	*Semotilus corporalis*	65	5.93	0.40	5.90	5.10	7.10
	rockbass	*Ambloplites rupestris*	75	5.95	0.42	5.91	4.96	7.24
	lake whitefish	*Coregonus clupeaformis*	62	5.94	0.45	5.93	4.85	7.10
	white sucker	*Catostomus commersoni*	219	5.98	0.52	6.00	4.50	7.27
	pearl dace	*Margariscus margarita*	158	6.00	0.52	6.00	4.86	7.44
	smallmouth bass	*Micropterus dolomieui*	65	6.01	0.40	6.02	5.12	7.24
	brook trout	*Salvelinus fontinalis*	331	6.07	0.69	6.04	4.40	8.63
	northern redbelly dace	*Phoxinus eos*	179	5.99	0.57	6.05	4.06	7.44
	walleye	*Stizostedion vitreum*	76	6.07	0.44	6.08	5.02	7.10
	creek chub	*Semotilus atromaculatus*	151	6.01	0.55	6.09	4.38	7.44
	finescale dace	*Phoxinus neogaeus*	67	6.09	0.60	6.10	4.79	7.44
	common shiner	*Luxilus cornutus*	64	6.15	0.51	6.10	4.86	7.44
	brook stickleback	*Culaea inconstans*	95	6.06	0.63	6.10	4.54	7.45
	fathead minnow	*Pimephales promelas*	75	6.16	0.49	6.11	4.90	7.44
	lake trout	*Salvelinus namaycush*	70	6.34	0.55	6.37	4.80	8.22
AMPHIBIANS								
	bullfrog	*Rana catesbeiana*	60	5.97	0.45	6.00	4.53	6.97
	green frog	*Rana clamitans*	111	5.63	0.52	5.65	4.37	7.06
	mink frog	*Rana septentrionalis*	89	5.59	0.47	5.61	4.45	6.98
	red-spotted newt	*Notophthalmus viridescens*	58	5.85	0.42	5.82	4.98	6.84
WATERBIRDS								
	common loon	*Gavia immer -- broods*	51	6.02	0.55	6.12	4.46	6.98
		-- pairs	188	5.95	0.56	6.01	4.25	7.25

TAXA	Common Name	Species Name	#	Mean pH	± SD	Median pH	Min pH	Max pH
Waterbirds cont'd	common merganser	*Mergus merganser -- broods*	47	6.25	0.34	6.33	5.60	6.97
		-- pairs	145	6.07	0.49	6.11	4.42	7.02
	hooded merganser	*Lophodytes cucullatus -- broods*	179	5.74	0.52	5.74	4.27	7.10
		-- pairs	308	5.71	0.57	5.72	4.25	7.42
	mallard	*Anas platyrhynchos -- broods*	27	5.68	0.38	5.67	5.10	6.62
		-- pairs	121	5.76	0.63	5.75	4.27	7.33
	black duck	*Anas rubripes -- broods*	111	5.76	0.53	5.72	4.67	6.98
		-- pairs	281	5.77	0.59	5.78	4.27	7.36
	wood duck	*Aix sponsa -- broods*	56	5.73	0.44	5.74	4.83	6.78
		-- pairs	87	5.70	0.50	5.75	4.41	6.78
	ring-necked duck	*Aythya collaris -- broods*	111	5.75	0.59	5.75	4.25	7.42
		-- pairs	195	5.75	0.64	5.68	4.25	7.60
	common goldeneye	*Bucephala clangula -- broods*	111	5.60	0.71	5.47	4.25	7.16
		-- pairs	196	5.54	0.71	5.43	4.25	7.29

* *Bosmina longirostris* is being used in this report even though the species has been reclassified (Demelo and Hebert 1994) and *Diaphanosoma birgei* includes *Diaphanosoma leuchtenbergianum* because of older and inconsistent nomenclature used in some of the original datasets.

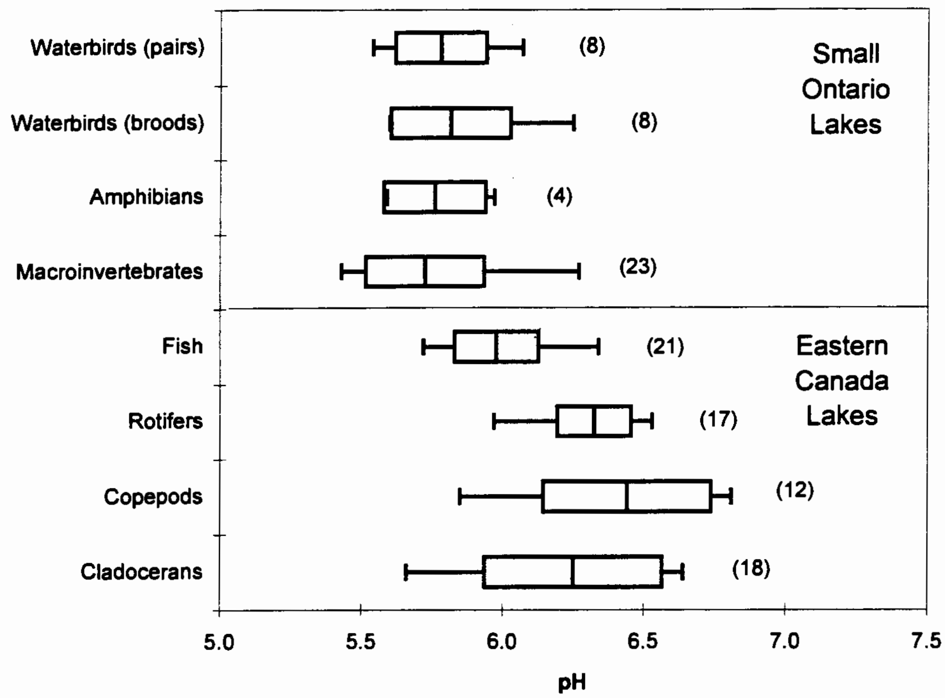

Figure 2: The average ± one standard deviation (boxes) pH of occurrence for species in several taxonomic groups. The minimum and maximum average pH at which a species within each taxonomic group was recorded is indicated by the upper and lower error bar limits. The data have been divided into two groups: small water bodies in Ontario (CWS contribution) and southeastern Canada. The number of species used in the calculations are shown in brackets and each species incorporated was present in a minimum of 20 water bodies in the dataset. See Table 2 for individual species' pH ranges.

persist in refugia in acid-stressed water bodies, but which will ultimately be lost from a lake if water quality requirements for reproduction are not met (see also Schindler *et al.* 1989, Minns *et al.* 1990). Species-specific characteristics (e.g. dispersal ability, reproductive strategy) will also control the nature and rate of the recovery of these organisms in lakes experiencing chemical improvements (e.g. Keller *et al.* 1992).

Across southeastern Canada, species-specific pH minima for many taxa appear similar, as demonstrated by the concordance in values between absolute minima derived from our data and those described in other studies. For example, minimum pH of occurrence for snails in Ontario (5.1; McNicol *et al.* 1995d) was very similar to that in Atlantic Canada (5.0 -- Peterson *et al.* 1987, 5.2 -- Schell and Kerekes 1988) and some observed laboratory values (5.0 -- Eilers *et al.* 1984), and also corresponds well with the minimum pH of 5.2 for gastropod occurrence in Norway (Økland and Økland 1986). The minimum pH for successful common loon reproduction across Ontario (4.5) is comparable to previous reports in Ontario (4.5 -- Alvo *et al.* 1988) and in Atlantic Canada (4.6 -- Kerekes 1990). Lake trout pH minima are remarkably consistent in several studies at a pH of 5.1 (Reckhow 1988, Kretser *et al.* 1989, Johnson *et al.* 1987); however, the minimum pH of lake trout occurrence was found to be slightly lower (4.8) in this assessment. Pike (pH 4.8) and lake trout had absolute pH minima which were higher than those observed for other species in the compiled datasets for southeastern Canada (e.g. white sucker, *Catostomus commersoni*, pH 4.4; brook trout, *Salvelinus fontinalis*, 4.4). Fish species, such as the fathead minnow (*Pimephales promelas*) and common shiner (*Notropis cornutus*), have been found to have high pH thresholds (pH 5.5 and 5.4 respectively -- Matuszek *et al.* 1990) similar to that determined for lake trout. Intuitively, taxa which cannot tolerate lower pHs as might be suggested by their pH minima are at greater risk from watershed acidification. From the taxa examined in this report, there are several examples of species which may be considered acid sensitive based on higher than average mean pH of occurrence. These are the cladocerans; *Daphnia galeata mendotae, Daphnia retrocurva,* and *Chydorus sphaericus* (mean pH ~ 6.6), the copepods; *Epischura lacustris, Acanthocyclops vernalis,* and *Leptodiaptomus sicilis* (mean pH > 6.7), the rotifers; *Kellicottia bostoniensis, Trichocerca multicrinnis,* and *Ascomorpha saltans* (mean pH > 6.4) and the fish; common shiner, fathead minnow, and lake trout (mean pH 6.1-6.3) based on the data provided from southeastern Canada. Alternately, based on data provided from small lakes in Ontario, the macroinvertebrates with a mean pH of greater than 6.2 were: *Hyallela azteca* (amphipod), *Gyraulus deflectus* and *Helisoma anceps* (snails), *Orconectes propinquus* (crayfish), *Placobdella ornatus* (leech) and *Somatochlora* spp. (dragonflies). The data collected for amphibians did not reveal a species with a mean pH of greater than 6.0, however two waterbird species, the common loon and the common merganser, did have a mean pH of greater than 6.0.

While pH minima are often used as pH thresholds for species and their probability of occurrence in water bodies, they may not be an accurate estimator of actual changes in the probability of finding a species as pH decreases for two reasons. First, species may persist for years in a lake even though conditions may not be suitable for successful reproduction, and they may be able to reproduce again should conditions improve (e.g. Gunn and Keller 1990). Moreover, pH minima for taxa are usually much lower than the normal pH range of occurrence for those species, as shown in Table 2. This may indicate that there is a range of pHs over which a species is less likely to be found, but may still occur (i.e. abundance decreases). This point is illustrated with bullfrogs (*Rana catesbeiana*), where the demonstrated minimum acidity tolerance is about pH 4.3 (Gosner and Black 1957). However, in the present study, they were captured in only one of 70 water bodies with pH < 5.0 (pH 4.5) in Ontario, but all other 59 (of 612) water bodies with bullfrogs had pH > 5.0. Similarly, critical pH values (values below which viable populations are not expected to persist, based on netting and field bioassay work; Beggs and Gunn 1986) for brook trout (pH 5.0) and lake trout (pH 5.5) are substantially higher than their known minimum pH of occurrence (see Table 2). A second reason why using pH minima may not provide accurate estimates is that these minima are often derived from field studies, and hence are dependent on the chemical distribution of the water bodies sampled, the sample size, and the frequency of sampling. With increased number of water bodies sampled, species are more likely to be found at lower pHs simply due to probabilities. For example, Schindler *et al.* (1989) used pH 6.0 as the minimum pH tolerance for common shiner and fathead minnow based on available data, but data published in 1990 dropped these minimum pHs by 0.5 pH units (Matuszek *et al.* 1990). In this study, the minimum pH of collection for fathead minnows was 5.0, and 5.1 for common shiners (from small lake sampling in Ontario; McNicol *et al.* 1996). For several other species, minimum pH tolerances are considerably lower in this study than previously noted, probably due to the larger number of water bodies used in this analyses than in most studies and the unfortunate lack of temporal trend data for lake pH in most datasets (a notable exception is the CWS dataset) which would give a better indication of the average annual pH. The use of small lakes in the dataset, which generally exhibit lower pH levels than larger lakes, could be one reason why pH minima are lower in this study.

These factors must be considered when developing predictive models of biotic impoverishment due to acidification or interpreting the results of various models. For example, in the northeastern United States, Schindler *et al.* (1989) used critical pHs for species and estimated that up to 14% of fishes had been lost in various subregions. Similarly, Matuszek *et al.* (1992) estimated that, in Ontario, 5.1% of known lake trout lakes, 1.6% of known brook trout lakes, 2.2% of known smallmouth bass (*Micropterus dolomieu*) lakes and 0.3% of known walleye (*Stizostedion vitreum*) lakes had lost their fish populations due to acidification. Most losses occurred in the Sudbury area. Modelling of fish losses in Norway has also used a critical pH approach, although it was derived in a slightly different manner. In this work, Rask *et al.* (1995) assumed that the critical pH for affected populations was the mean of the pHs of water bodies where reproduction was affected, not simply the minimum pH of occurrence. Using this approach, they showed that several fish populations have disappeared from lakes (i.e. 60% of roach populations in Finland), due to acidification sensitivity (roach *Rutilus rutilus* pH min = 5.2 and critical pH=5.8; European perch *Perca fluviatilis* pH min = 4.8 and critical pH=5.0; Rask *et al.* 1995).

Clearly, there are advantages and disadvantages to using pH-based models of acidification effects on aquatic biota. The biggest advantage is that these models require essentially one parameter to run the model and estimate effects, namely pH. Therefore, many data sources may have this information and can be used in modelling efforts, and if pH ranges are reasonably accurate, estimation and/or prediction of theoretical temporal and spatial changes in species distributions based on pH changes in water bodies is possible (see Schindler *et al.* 1989, Minns *et al.* 1990). Such models are also useful in that they establish threshold values that are much easier for resource managers or wildlife officials to use in making decisions regarding effects of acidification on habitats and subsequent management approaches. The principle disadvantage of these models is that, for many taxa, the models may be less precise or accurate than models which account for other factors. This will be particularly true for semi-aquatic organisms that can move among lakes or streams and thus are not entirely restricted by the chemical conditions in a water body. In these cases, additional variables (such as lake area or other chemical parameters) increase our ability to predict the occurrence of a species (see Section 4 for details).

3.3 Proportional community response to pH

Previous assessments (RMCC 1990, Minns *et al.* 1990) have predicted the response of taxonomic groups using pH minima only. However, each species should have both a minimum and maximum pH between which it occurs, (although it never occurs in all water bodies within this pH range). Based on species specific pH ranges, relationships between pH and potential species richness were generated for each dataset separately and combined (datasets are listed in Table 1), for four taxonomic groups: waterbirds, fish, zooplankton, and macroinvertebrates (data were insufficient for amphibians). Species which occurred in less than 5% of water bodies within each dataset were excluded from the analyses because we assumed the accuracy of pH ranges for those species would be questionable. This reduced the total number of fish in the datasets from 73 to 21 species, and similarly for zooplankton, from 128 to 49 species. After the exclusion of these species, the remaining ones within each taxonomic group were used as the total or maximum number of species. The proportion of the maximum number of species present at particular points over the pH range for each dataset was determined and curves were generated by using these proportions. All the proportional response curves could be described significantly by quadratic equations (RS/1 1990, Systat 1993; See Appendix 3). By calculating the upper limit for each quadratic equation, the lake pH at which the *maximum proportion* of the potential *species* pool (MPS) was present could be determined. MPS is different from the total number of species since usually not all species are present at the upper limit pH (i.e. some particular species ranges do not include the upper limit pH). Also, for the lower pH range of each curve, the pH at which 0% (no species), 50%, and 75% of the MPS occurred was determined (Table 3).

A significant relationship could not be determined for waterbird proportional response to pH. Waterbird breeding pairs (spring survey data collected during nest initiation) were not very sensitive indicators of lake acidity because of a broad range of occurrence for almost all the species assessed. Several reasons that would explain why the proportional response of waterbird species is not a good indicator of acidification effects are discussed in the summary.

There were slight differences in the proportional response to pH between the remaining taxonomic groups, and also between regions, within southeastern Canada (Figures 3 & 4). However, zooplankton, macroinvertebrates, and fish exhibited a similar trend in response; a loss in species richness as pH decreased. This finding is consistent with the pattern observed by Minns *et al.* (1990) where fish, molluscs, and rotifers were the most affected taxa, from a total of seven groups, by decreasing pH in Ontario.

Separate analysis of each dataset in this report revealed slight regional and lake size differences in taxonomic responses as well. Atlantic Canada appears to have consistently lower pH thresholds for fish species loss that may be due to the higher DOC levels in many water bodies which ameliorate the effect of pH (Havens 1993). For fish populations, the pH at which MPS occurred ranged from 5.6 to 5.8 in Nova Scotia, 5.8 in Québec, and 6.0 to 6.1 in Ontario (Table 3). Collectively, an overall average pH limit of 5.9 was found for southeastern Canada. Also, equivalent losses in MPS for small lakes or wetlands (Table 3; CWS data for

Ontario lakes generally < 20 ha) occurred at lower pH levels than for larger lake datasets. This would indicate that in addition to provincial differences in pH thresholds, there would be lake size differences within those regions as well. The regional trends in proportional response were not similar between fish and zooplankton, potential species richness of zooplankton did not indicate a gradient from east to west. Generally pH maxima were higher for zooplankton than for fish (Figure 4), especially in the Québec region where MPS was reached at pH = 6.7. The overall pH at which the maximum number of species are present was 6.1.

Based on the results of the proportional response analysis, if lake pH declined to below 5.5 from 6.0 then a 10% average loss in species richness would occur in fish and zooplankton populations in southeastern Canada. Similarly for macroinvertebrates in Ontario, if pH dropped below 5.3 from 5.8, there would be a 10% loss in richness. However, the error in estimating the proportional loss in species richness cannot be determined; therefore setting pH thresholds should be done in conjunction with other empirical analyses involving the prediction of species richness, such as the multivariate regression models presented in Section 4.3.

Table 3: Fish, zooplankton and macroinvertebrate proportional response results from separate regions within southeastern Canada and averaged over all regions. MPS is the maximum proportion of the species pool within an area (dataset) which can be theoretically achieved based on pH ranges, the pH at 100%, 75%, 50% and 0% of MPS (i.e. relatively the maximum species proportion, 25%, 50% and 100% proportional loss in species) is shown for each dataset.

Dataset	Region	MPS	pH 100%	pH 75%	pH 50%	pH 0%
FISH						
OMNR	Central Ont.	0.682	6.06	5.51	5.28	4.95
MESO	Central Ont.	0.596	5.99	5.19	4.87	4.40
DFO	Nova Scotia	0.962	5.81	5.17	4.91	4.52
DFO/CWS	Nova Scotia	0.895	5.57	4.87	4.57	4.18
QLS	Québec	0.700	5.82	5.23	4.99	4.65
CWS	Central Ont.	0.759	5.94	5.12	4.79	4.30
		Overall Average:	5.87	5.18	4.90	4.50
ZOOPLANKTON						
OMEE	N. & C. Ont.	0.856	6.11	4.94	4.55	3.79
NIS	Ont.,Que., Nfld.	0.994	6.20	5.25	4.84	4.30
UofM	Québec	0.946	6.68	5.81	5.46	5.20
UofT	Central Ont.	0.783	5.60	4.76	4.41	3.92
DFO	Nova Scotia	0.685	5.67	5.02	4.74	4.35
		Overall Average:	6.05	5.16	4.80	4.31
MACROINVERTEBRATES						
CWS	Central Ont.	0.860	5.67	4.80	4.42	3.89

As alluded to earlier, there were differences in the potential richness in large and small water bodies and along longitudinal gradients. Thus, pH is not the only variable affecting the probability of finding a particular species in a lake, or the variation in species richness among water bodies. In the past, other variables, such as lake area, geographic location, and lake colour have been shown to play an important role in determining species diversity, occurrence, and richness (Minns 1989, Pinel-Alloul *et al.* 1990, Hinch *et al.* 1991, Locke *et al.* 1994). These variables confound efforts to delineate the effects of pH alone on species richness or occurrence and therefore need to be taken into consideration in modelling efforts The relative contribution of other lake and regional characteristics are addressed in Section 4.

Some separation of proportional response was tested for DOC since the hypothesis of different responses of species in coloured and clear systems is a widely held tenet. Since it is impossible to calculate an error estimate for individual pH minima for particular species, any significant differences in proportional response of species richness over pH cannot be tested between coloured and clear lake groups. However, a comparison of several curves generated for DOC groups (groups were divided by the median DOC level for that area) within regions (or datasets) in southeastern Canada could indicate some separation in pH and DOC effects by the differences in shape, and the range of pH covered, by the curves. In Figure 5, there were no

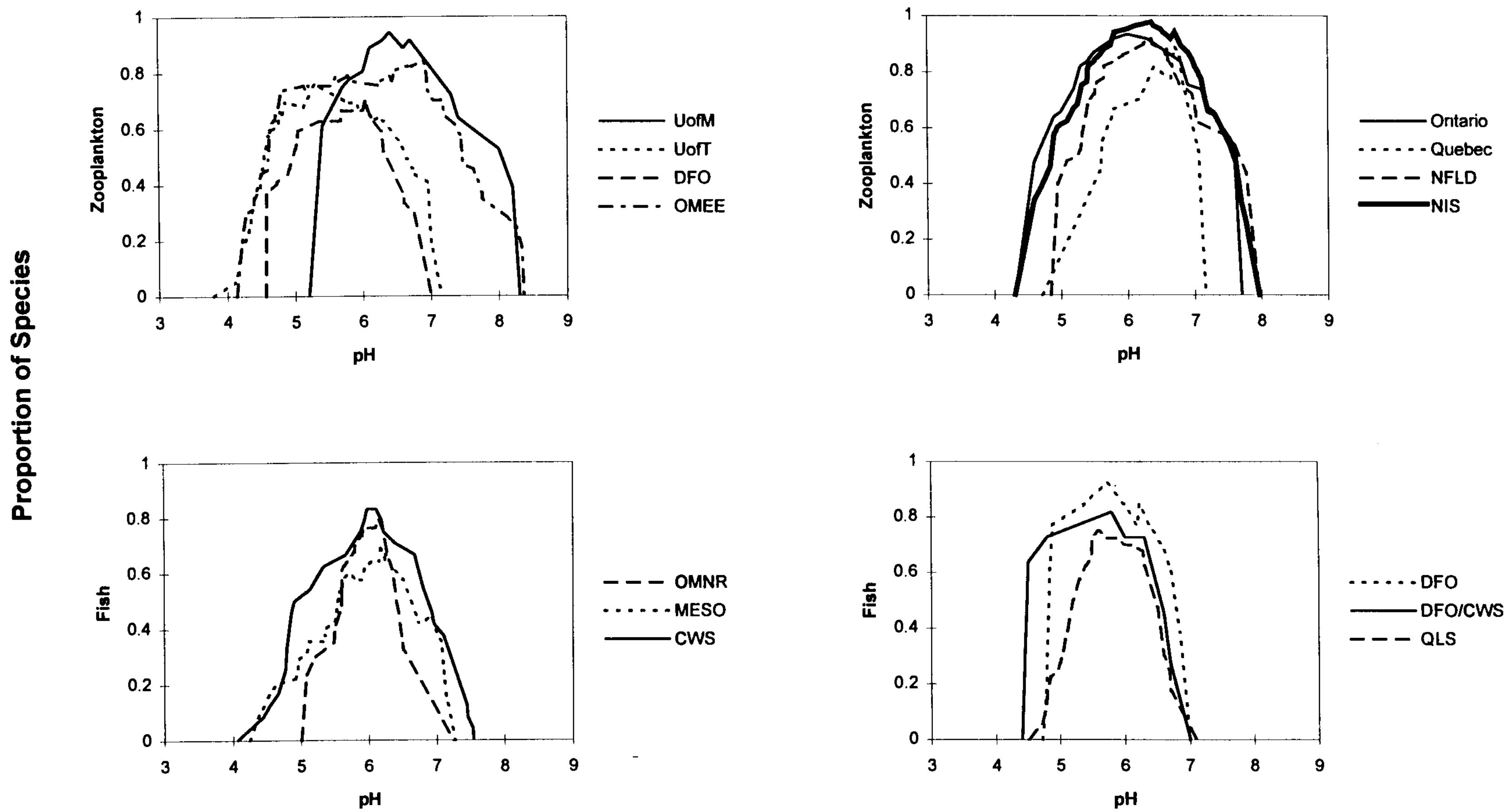

Figure 3: Proportional response curves for zooplankton (A and B) and fish (C and D) species richness response to pH for each of the datasets used in statistical analyses. See Table 1 for a list of the datasets referred to in the figure legends. The NIS dataset has been divided into its provincial subcomponents for comparison purposes with the general, combined NIS curve in B.

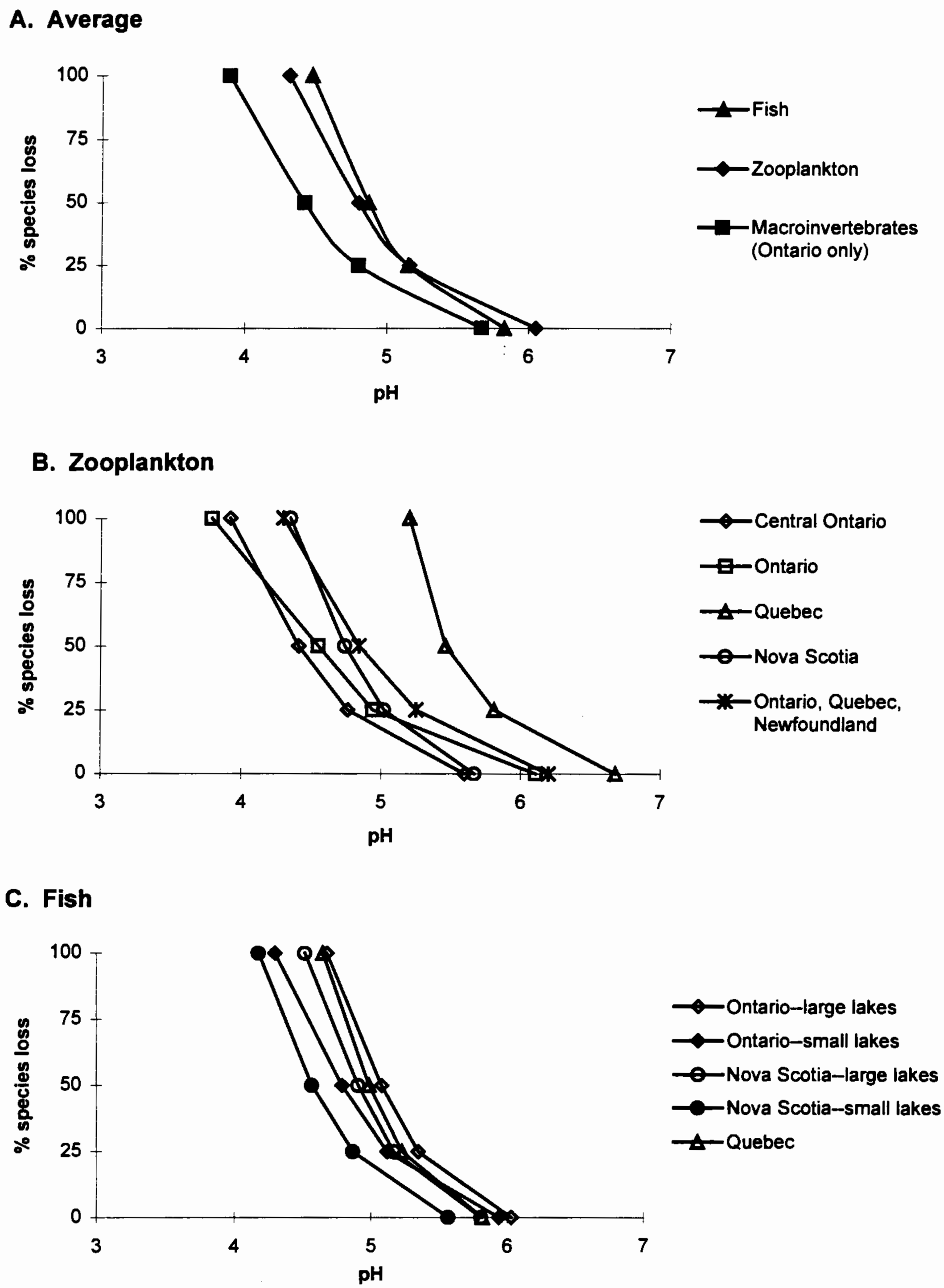

Figure 4: Proportional loss curves (potential proportion of species lost) for different taxa and regions in southeastern Canada (A) Fish and zooplankton loss curves averaged for southeastern Canada over all datasets, macroinvertebrate loss curve applies to small water bodies in Ontario only (CWS data). (B) Zooplankton loss curves for central Ontario (UofT), Ontario (OMEE), Québec (UofM), Nova Scotia (DFO), and Ontario, Québec, and Newfoundland combined (NIS). (C) Fish loss curves for large water bodies in Ontario (OMNR & MESO), Québec (QLS) and Nova Scotia (DFO), small water bodies in Ontario (CWS) and Nova Scotia (DFO/CWS).

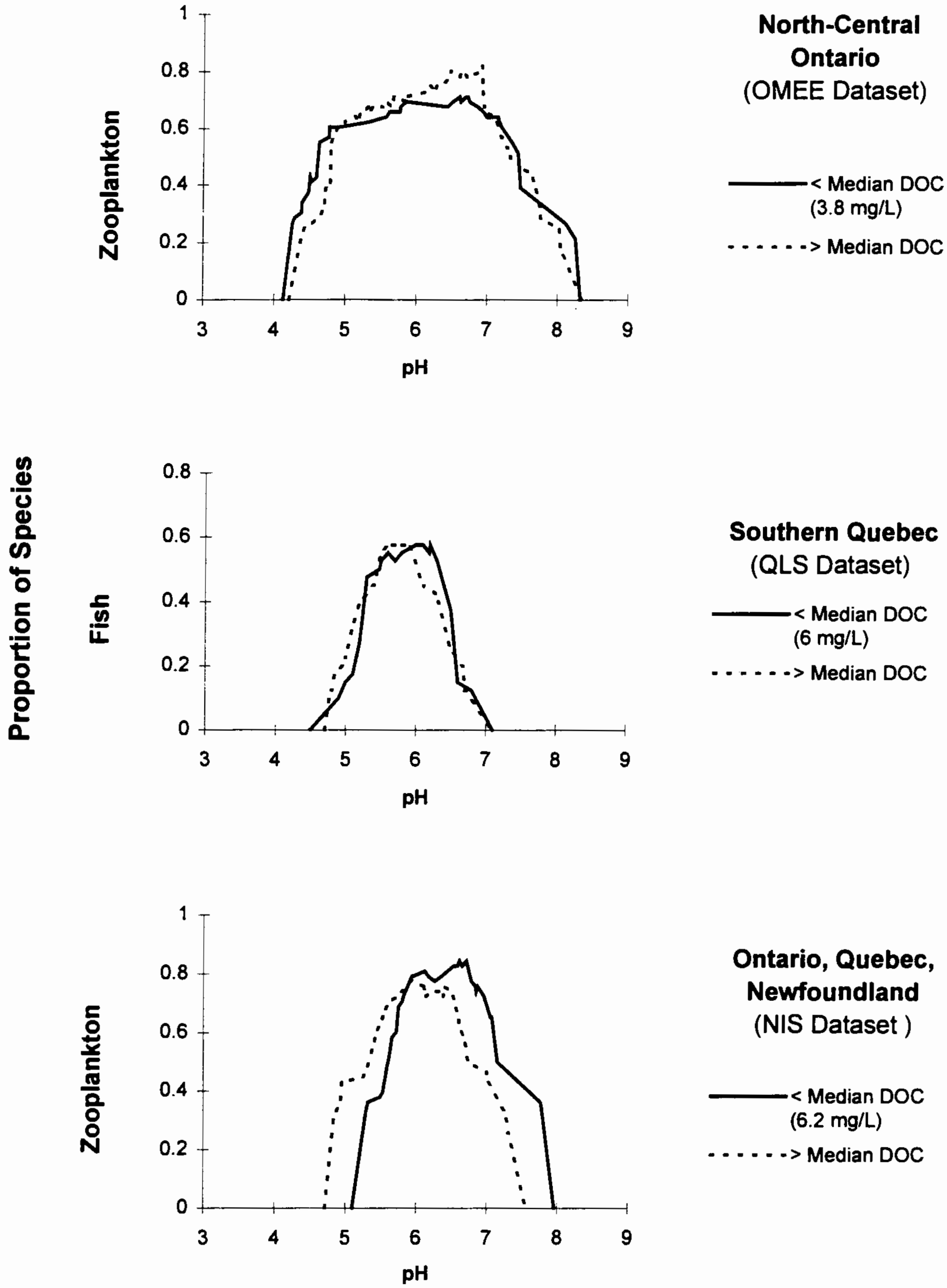

Figure 5: Proportional response curves were generated for DOC categories in three of the datasets (2 zooplankton and 1 fish) that cover different regions in southeastern Canada. The water bodies within each dataset were divided into two DOC groups by using the median DOC level. Proportional response curves were generated separately and are shown as solid (< median DOC) and dashed (> median DOC) lines.

apparent differences between clear and coloured lake groups in both zooplankton proportional response in Ontario and fish proportional response in Québec. However, when zooplankton proportional response is separated into clear and coloured lake sets in a larger dataset, such as the NIS dataset, then some separation in proportional response does occur indicating a potential difference in clear and coloured water bodies on a larger scale. The curve for coloured water bodies in the NIS dataset is shifted down the pH scale by approximately 0.5 pH units. This probably is a reflection of some correlation between lake colour, region (Newfoundland has fewer species per lake and more highly coloured lakes than other areas in southeastern Canada) and species richness, since this shift did not occur in more localized datasets. Since there does seem to be an effect of DOC on pH ranges, however under the current analyses no statistical difference can be determined, and there could potentially be confounding factors such as longitude or region that change the interaction between pH and DOC, then the multivariate analyses presented in the following sections (especially multiple regression analyses) are more valid for determining the relative contributions of particular lakecharacteristics than proportional response speculations. Therefore, for statistical purposes we need to look to these models.

3.4 Summary

Evidence gathered in North America and Europe since the 1970s confirms that damage to aquatic ecosystems occurs when acidic deposition causes the pH of water bodies to drop below pH 6. Deleterious effects on biota have been detected at all trophic levels (e.g. phytoplankton, zooplankton, macroinvertebrates, fish, amphibians and waterbirds; Schindler 1988, Schindler *et al.* 1989, RMCC 1990, Minns *et al.* 1990, Longcore et al 1993, see also Section 3.1). Using data from across southeastern Canada (summarized in Fig. 2), taxa from four broad, taxonomic groups occurred 95% of the time above pH 5.1. Reported pH limits for various regions were generally lower than 5.1; however these values agreed with those found in other datasets, suggesting similar pH minima across southeastern Canada. High DOC (or highly coloured) lake conditions typical of Atlantic water bodies, may have been associated with the tendency to have slightly lower pH minima for species in that region and could suggest that coloured water bodies under certain circumstances may have communities which react differently to pH changes than clear lake communities.

Generally, by plotting the theoretical distribution of species richness over pH (based on the pH ranges of species in southeastern Canada), the greatest proportion of species were present around a pH of 5.9. This value was obtained from curves which were based on the lowest pH at which species have been recorded in our data and therefore this pH should be considered to be a lower limit pH at which species richness could reach maximum levels; statistically the threshold would be higher. The findings augment the existing evidence that species diversity is consistently highest across taxonomic groups above pH 6. In all, these results (in conjunction with the taxon-specific models discussed in the following sections) provide strong regional support for the pH 6.0 criterion for the protection of aquatic biota as suggested in the 1990 Assessment. A point to stress is that the proceeding models do take into account the effect of other factors, in particular lake area, when making predictions about species occurrence or richness and therefore are superior from a predictive standpoint.

4.0 Multiple factor relationships

4.1 Introduction

In Section 3, we presented evidence that the occurrence of many aquatic organisms in southeastern Canada can be predicted using the pH of water bodies, and that theoretically species richness begins to decline below pH 6. Moreover, these relationships were generally consistent across regions with some slight variations. The findings were also compatible with previously-derived pH relationships, and with findings from other regions in North America. However, the occurrence of a particular species, or the species richness within a particular taxon, is affected by characteristics of water bodies other than pH alone. For example, Rask *et al.* (1995) identified labile aluminum (Al) as a key variable in determining which water bodies had fish populations affected by acid rain, and they acknowledged that calcium and organic carbon concentrations also influenced pH toxicity. Conlon *et al.* (1992) showed that a combination of pH, alkalinity and conductivity best predicted (>92% correct) the presence of lake trout in lakes near Sudbury, and Blancher *et al.* (1992) used combinations of lake area, fish presence, pH, DOC and total phosphorus in lakes to model the probability of occurrence of waterbird species in acid-stressed lakes of northeastern Ontario. Biogeographical factors (Mandrak 1995), physical environmental factors (Alvo *et al.* 1988, Pinel-Alloul *et al.* 1995), predatory processes (Hinch *et al.* 1991, Bendell and McNicol 1995), chemical factors (Holtze and Hutchinson 1989) and whether the species are introduced or native (Graham 1993), have all been shown to affect species richness or the occurrence of a particular species. In this section, we included other physical and chemical variables that may influence the occurrence of taxa (along with pH) in statistical analyses with two goals in mind:

1. To determine and partition the relative contributions made by significant physical and chemical, water body and tertiary watershed characteristics.

2. To develop accurate and representative models predicting species richness or occurrence in water bodies in southeastern Canada that may expand upon previous assessment work.

4.2 Variable Selection

The same datasets referred to in Section 2 were used in these analyses. However, variables which were sampled and provided in each dataset were not consistent across datasets (see Section 2). Because one of our goals was to develop models that were applicable regionally, we needed to use variables that were available or could be determined from as many datasets as possible. Thus, variables used in statistical analyses were chosen if they met the following criteria:

1. Preliminary analyses indicated that the variable may be an important determinant of the biotic response being tested (i.e. lake areas affect species richness).

2. Previous evidence or models suggested that the variable may have effects on aquatic biota (i.e. temperature affects species' distributions).

3. The variable was sampled consistently, or could be derived from other information, in most

of the datasets. For example, lake colour has been shown to be an important determinant of species richness in water bodies for several taxonomic groups. Lake colour data were not collected in some datasets, however, dissolved organic carbon (DOC) was usually tested in those datasets. Because lake colour and DOC are highly correlated, regional and local regression equations were developed and used to determine DOC content from lake colour where possible (see Appendix 2).

4. Variables that were highly autocorrelated were not used in the analysis; instead the variable which best represented a subset of correlated variables was chosen (i.e. pH was chosen from the subset; alkalinity, pH, calcium and magnesium concentrations). There were certain exceptions made if the inclusion of an additional variable could be justified (i.e. Al and SO_4 concentrations are often somewhat correlated with pH). A general rule of thumb of <30% correlation was applied.

The following list of variables were used in logistic and multiple regression analyses; however, geographic variables were only included where the dataset (pooled or not) covered a large enough area of southeastern Canada (e.g. across provinces) to warrant their inclusion (i.e. more than one localized area such as the Algoma district):

Geographic variables: Longitude, latitude, tertiary watershed (TWS) area, TWS elevation, TWS annual temperature

Physical variables: Lake area, midlake or maximum lake depth

Chemical variables: pH, DOC, SO_4, Al

Descriptive statistics and normality tests were used to determine if the variable data were normally distributed. If not, then the appropriate transformation was used to obtain as close to a normal distribution as possible since normality is a requirement for many of the analyses performed. The natural logarithm transformation was used in most cases and is indicated in the equations shown for each separate analyses (ln + 1 transform was used where necessary). In select cases, extreme outliers were discarded from the dataset. For example, data from Lake Superior was not incorporated due to the extreme lake area value which would drive any subsequent relationships if used in the analysis. Pearson's correlation matrices were generated for each of the datasets to test for greater than 50% correlation between variables (as mentioned, one or two select variables in a suite of autocorrelated variables were tested in analyses; many of the available variables were highly correlated with each other - see Mallory *et al.* 1997 for an example). Interaction terms and quadratic terms that involve the variables listed above were also used in analyses where appropriate. The tertiary watershed (TWS) codes were assigned to each water body by using latitudes and longitudes in a SPANS GIS database (TYDAC Technologies Inc. 1995). Tertiary watershed area, elevation, and mean annual air temperature were matched with each TWS assignment (see Minns and Moore, 1995).

4.3 Taxon-specific relationships

4.3.1 Introduction

Several studies have shown that the occurrence of certain species can be predicted in certain regions of eastern Canada using various physical and chemical characteristics of water bodies. For example, Kelso *et al.* (1986) found brook trout occurrence in southeastern Canada was predicted by lake depth, elevation, and lake colour. In the same article, a statistically significant eastern Canada model could not be developed for white sucker, yellow perch, or lake trout using discriminant analysis using the available parameters. However, lake depth, pH, elevation, and colour were separate and important predictors on a provincial scale for these species. Similarly, lake area, pH, and water colour have also been shown to be important predictors of common loon breeding pair occurrence and breeding success (pairs - Alvo *et al.* 1998, DesGranges and Houde 1989; broods - Kerekes 1990, McNicol *et al.* 1995c). Several species were selected from the southeastern Canada dataset for developing species specific models. The species for which pH was a significant determinant of their occurrence are discussed in the following sections.

4.3.2 Statistical Methods

Maximum-likelihood, logistic regression analysis (Systat 5.03 1993, SAS 6.10 1993) was used to determine the relationship, if any, between a species' presence or absence and the various lake characteristics mentioned in Section 4.2 If a significant logistic equation could be fit, then the probability of a particular species' occurrence could be determined given a key set of chemical and/or physical variables. Slightly different approaches were used for different taxonomic groups based on geographical differences in the datasets. They are discussed in the following sections.

Zooplankton and Fish

For the purposes of developing a southeastern Canada model, relatively ubiquitous species were chosen in order to avoid confusing regional and local effects. Therefore, a subset of species that were present in all datasets was determined and representative species that occurred in 30 to 70% of the water bodies within each dataset were chosen for logistic regression analysis. This range in species' occurrence was chosen arbitrarily to select species which were present in enough lakes to provide a statistically significant model, but not present in too many water bodies in order to exclude extremely tolerant species. The species that met this criteria included brook trout (*Salvelinus fontinalis*), lake trout (*Salvelinus namaycush*), golden shiner (minnow, *Notemigonus crysoleucas*), brown bullhead (catfish, *Ameiurus nebulosus*), pike (*Esox lucius*), pumpkinseed (sunfish, *Lepomis gibbosus*), white sucker (*Catostomus commersoni*), and yellow perch (*Perca flavenscens*) as representatives from several different fish families. The cladocerans, *Bosmina longirostris, Daphnia spp.,* and *Holopedium gibberum*, the copepods, *Epischura lacustris, Mesocyclops edax,* and *Tropocyclops prasinus*, and the rotifers, *Keratella taurocephala* and *Polyarthra spp.*, were chosen as zooplankton representatives. Several of these species have already been incorporated into other models. *Daphnia galeata mendotae*, *K. taurocephala*, and *B.*

longirostris have been shown to be good indicators of acidification (Keller *et al.* 1990, Siegfried 1988). Also, brook trout and lake trout logistic models have been developed for the eastern United States that link pH and calcium concentration to trout occurrence (Reckhow 1988); similarly for lake trout in Ontario (Beggs *et al.* 1985).

The logistic equation and loss function in Minns and Moore (1995) were used to determine the most important variables that affected the occurrence of the aforementioned species of fish and zooplankton (Systat 5.03 1993). An *a posteriori* test, the Kappa statistic (Titus *et al.* 1984, Monserud and Leemans 1992), was used to determine the significance level of each logistic equation developed (Table 4 for equations). Figure 6 shows the relationship between pH, other significant variables, and the occurrence of selected species as pH changes. Individual analyses of all of the species listed at the beginning of this section were performed; however only the species for which pH was a significant determinant are shown.

Macroinvertebrates and Wildlife

Representative macroinvertebrate, amphibian and waterbird taxa were used to depict the relationship between pH, other chemical and physical parameters, and the occurrence of these taxa. Although all of these data came from collections in Ontario (Table 1), taxa were chosen that were found across southeastern Canada.

Among macroinvertebrates, the mayfly *Caenis* spp., gastropods (dominated by *Amnicola limosa, Physella gyrina*, and *Ferrissia* spp.), the amphipod *Hyallela azteca*, and the scavenger leech *Percymoorensis marmoratis* were chosen. These taxa were known to have predictable distributions in relation to pH from previous studies (e.g. Stephenson and Mackie 1986, Bendell and McNicol 1991, 1993). Collection methods and timing are described in McNicol *et al.* 1995d. Because the datasets for these analyses were all from northeastern and central Ontario, and represented 66 small (< 20 ha) water bodies (92 water bodies for the leech, as data collected in an identical manner for 26 other Sudbury lakes were included in the analyses; Bendell and McNicol 1991, McNicol *et al.* 1995d), no geographic variables were included in stepwise, backward-elimination logistic regressions (PROC LOGISTIC, SAS 6.10 Institute 1993). Variables included in these regressions are described in Section 4.2.

For amphibians, the bullfrog was chosen as a representative species because it is common across southeastern Canada, it breeds in permanent (rather than ephemeral) water bodies, and because earlier studies had suggested that it might be sensitive to acidification (Clark 1986b). Data on bullfrog occurrence were collected from CWS (Ontario) baited minnow trap lake surveys between 1988-1994 (Table 1; McNicol *et al.* 1996). These lakes were principally small (< 20 ha), but did include some larger lakes (> 100 ha).

For waterbirds, piscivorous species that were common across southeastern Canada were chosen, namely the common loon (here loosely considered in the waterbird group) and the common merganser (*Mergus merganser*). Both of these species may be sensitive to acidification (Longcore *et al.* 1993). However, geographic variables were excluded because data were available only for Ontario water bodies. For loons, explanatory variables allowed to enter the equation were restricted to lake area and pH due to the limited number of chemical and other biological parameters (e.g. presence of fish) available for water bodies across

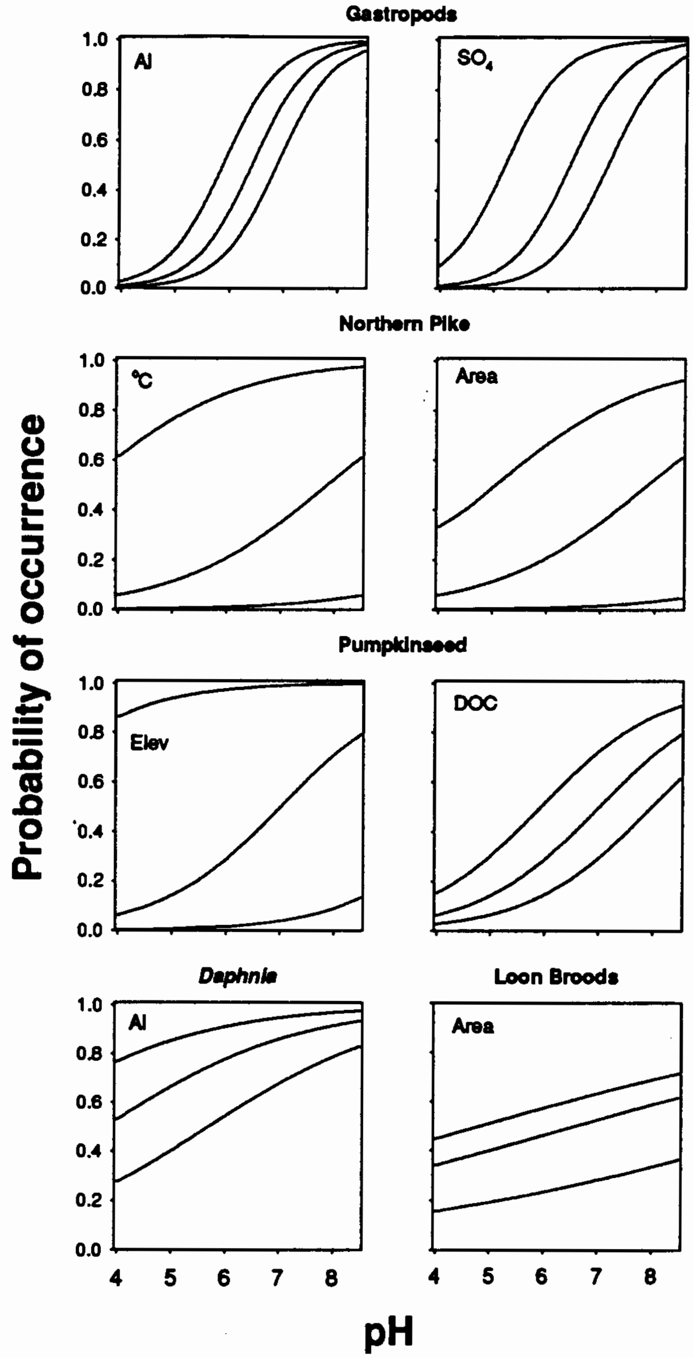

Figure 6: Logistic regression results showing the probability of occurrence of different taxa in relation to lake pH. Each pane shows the species occurrence versus pH at specified levels of the other significant variables (outside of pH) determined from our analyses. The three curves in each pane represent the range (upper and lower curves) and the mean (middle curve) for each of the non-pH variables. For gastropods: Al = 10, 112, and 200 $\mu g \bullet L^{-1}$ with SO_4 = 7 $mg \bullet L^{-1}$ (left panel) and SO_4 = 2, 7, 10 $mg \bullet L^{-1}$ with Al = 112 $\mu g \bullet L^{-1}$ (right panel); for pike: mean annual air temperature = 0, 4, 8 °C with lake area = 100 ha (left panel) and lake area = 1000, 100, 1 ha with temperature = 4 °C (right panel); for pumpkinseed: elevation = 200, 300, 400 m A.S.L. with DOC = 5 $mg \bullet L^{-1}$ (left panel) and DOC = 1, 5, 15 $mg \bullet L^{-1}$ with elevation = 300 m (right panel); for *Daphnia*: Al = 10, 100, 1000 $\mu g \bullet L^{-1}$; and for loon reproduction (minimum of 1 large young produced): lake area = 100, 50, 10 ha. The values are listed from upper to lower curves in each panel (see Table 4 for equations).

Ontario. Data on common merganser indicated breeding pairs and broods came from aerial surveys conducted by the CWS (Ontario) across central and northeastern Ontario, 1988-1995 (McNicol *et al.* 1996). Data on loon pairs and young, and lake chemistries, came from the CWS aerial surveys (McNicol *et al.* 1996), by volunteers of the Canadian Lakes Loon Survey (jointly with Long Point Bird Observatory; McNicol *et al.* 1995b), and from the Acid Precipitation in Ontario Study conducted by the Ontario Ministry of Environment and Energy (OMEE; Neary *et al.* 1990). Analyses were restricted to lakes between 2-500 ha in open water area.

We conducted a separate, site-specific analysis for loon-habitat relationships in Kejimkujik National Park in southwestern Nova Scotia because of some unique characteristics of this site, notably its exceptionally low loon breeding success (Kerekes *et al.* 1994). The Kejimkujik site is one of the "clusters" used in the 1997 Assessment, and represents the extremely acid-sensitive, highly colored (Section 3.3), nutrient-poor and calcium-poor habitat found in some regions of the Maritimes. Lakes in the park (n=46) were surveyed by CWS (Atlantic) personnel between 1988-1994 by canoeing the perimeter of each lake and recording the presence of adult loons and large young (Kerekes 1990; similar to the methods used in Ontario). The resulting dataset represents a relatively complete record in a unique, sensitive habitat of an entire population of loons. Chemical data for this analysis were means of annual average growing season chemistries from 1983-1985.

4.3.3 Results and Discussion

Zooplankton and Fish

The probability of occurrence of pike, pumpkinseed, brook trout, and *Daphnia* spp. in southeastern Canada was linked to the pH of water bodies. In addition to pH, lake area and average annual air temperature were important, positive determinants of the pike distribution in southeastern Canada. Even though the relationship is not depicted in Figure 6, longitude, lake area, and pH can be used to predict the probability of brook trout occurrence in water bodies. That is, brook trout are more likely to occur in the Atlantic provinces, in larger water bodies with higher pHs. Pumpkinseed occurrence increases at higher elevations, low DOC levels, and low pHs (Table 4).

The occurrence of other fish species used in the analysis were correlated with other variables than pH; although this may not necessarily rule out the possibility of indirect, acidic precipitation effects, such as spring freshets, or synergistic effects with other environmental stressors, such as Hg or UV-B. Lake trout, white sucker, and yellow perch occurrence were all affected to some extent by the aluminum concentration in water bodies (these equations are not shown in this report). However, due to the lower number of water bodies in which aluminum was measured, the results are not conclusive. Sulphate concentrations, latitude, and lake area were correlated with brown bullhead distribution, while latitude was the only variable that predicted golden shiner occurrence. With the exception of the golden shiner, the distributions of all of the fish species assessed were affected by pH or an associated variable, such as SO_4 or aluminum concentration.

Meanwhile, the only significant logistic regression which could be determined for zooplankton was one for *Daphnia* spp, where presence in southeastern Canada was coincident with low aluminum concentrations and higher pH levels. For the same reasons as cited earlier, total Al was measured on a sporadic basis and therefore these results, although significant should be validated with further data collection or statistics.

The findings for zooplankton and fish are consistent with previous results (Wales and Beggs 1986), but may be more valuable than other models in trying to determine the relative effects of both regional and local variables on certain species' occurrences in southeastern Canada and the relation to pH shifts. A multivariate approach is better when trying to delineate pH effects in different systems. All the evidence indicates that reductions in lake pH would directly or indirectly detrimentally affect several species of zooplankton and fish. Therefore based on the relative contribution of the other significant factors which affect the probability of a particular species' occurrence in an area, we can make better estimates of the types of systems in which those particular species are at greater risk to pH changes. For example, pike are not present in warm climates with small water bodies (8°C annual temperature and 1 ha), therefore the probability of occurrence for pike would be most affected by pH changes in water bodies with an annual average air temperature of 4°C and 100 to 1000 ha in lake area. Many Precambrian Shield lakes fall into this category and therefore would be at a greater risk under acidification stress. By defining ranges for important variable this way, we could determine areas of greater concern for particular species and delimit these areas of concern for preserving biota from the detrimental effects of acidification.

Macroinvertebrates and Waterbirds

For *Caenis* spp., gastropods, *Hyallela azteca* and *Percymoorensis marmoratis*, pH had a significant, positive effect on the probability of their occurrence in water bodies (Table 4). In water bodies; macroinvertebrate communities, and particularly the nektonic taxa (McNicol and Wayland 1992, Bendell and McNicol 1995a, McNicol *et al.* 1995d), analyses were separated for water bodies with (n=33) and without fish (n=33). Among fishless water bodies, pH was the only variable to have a significant effect ($F=19.5$, $df=1$, $p<0.001$) after accounting for the significant effect of region ($F=30.7$, $df=2$, $p<0.001$), with a model $r^2 = 0.76$ (note that Al was excluded from the final regression - when entered, this variable split part of the variance explained by pH such that neither were statistically significant). In contrast, only region contributed to macroinvertebrate species richness among water bodies with fish ($F=3.9$, $df=2$, $p=0.03$, $r^2 = 0.32$), in part due to the reduction in the range of pH and other chemistries of water bodies containing fish in this dataset.

For modelling purposes, we excluded the categorical variable "region" from the analyses, and ran stepwise multiple regressions on taxonomic richness in water bodies with and without fish. For fishless water bodies, we obtained the following regression:

> Richness = -37.16(12.68) + 8.74(2.31)*pH + 6.89(3.08)*lnDOC + 7.28(2.41)*lnAREA,
> ($p<0.001$, $r^2 = 0.54$, n=33 water bodies, where DOC is in mg/L and AREA is in ha).
> Algoma, Muskoka, and Sudbury regions of Ontario (n=629 water bodies; 1988-1995).

Eight species commonly nest on these water bodies (McNicol *et al.* 1987a, McNicol *et al.* 1995c), and a range of 0-8 species were recorded as breeding on any lake. Each lake was entered into the analysis once, and was scored as having a species present if that species was

Table 4: Logistic equations describing the probability of species occurrence in several taxonomic groups. The following equation describes the general form of the logistic equation that the parameter estimates below apply to:

$$\text{Probability of occurrence} = \frac{EXP\,(B0 + B1 * \text{variable1} + B2 * \text{variable2} + B3 * \text{variable3})}{(1 + EXP\,(B0 + B1 * \text{variable1} + B2 * \text{variable2} + B3 * \text{variable3})}$$

Species	B0 (intercept)	B1*variable [a]	B2*variable	B3*variable	Kappa statistic	p value
Brook Trout	9.73 (0.88)	-0.19 (0.01) * LONG	0.20 (0.09) * lnAREA1	0.38 (0.09) * pH	0.376	< 0.001
Northern Pike	-6.47 (0.87)	-0.81 (0.05) * TWStemp	0.88 (0.06) * lnAREA1	0.72 (0.14) * pH	0.544	< 0.001
Pumpkinseed	58.96 (4.19)	-11.16 (0.75) * TWSelev	-0.88 (0.22) * lnDOC1	0.89 (0.15) * pH	0.334	< 0.001
Epischura lacustris (zooplankton)	-4.28 (1.42)	0.05 (0.04) * DOC	-0.25 (0.14) * lnAl	0.70 (0.17) * pH	0.076	0.106 (NS)
Daphnia spp.	--	-0.48 (0.06) * lnAl	0.57 (0.05) * pH	--	0.245	< 0.001
					Concordance (%)	
Caenis spp.	-19.49 (7.88)	-0.03 (0.01) * Al	0.62 (0.28) * DOC	3.35 (1.36) * pH	93.4	< 0.001
Gastropods (snails)	-7.80 (6.47)	-0.45 (0.24) * SO4	-0.01 (0.01) * Al	1.89 (0.94) * pH	86.6	< 0.001
Hyallela azteca (amphipod)	-22.87 (5.67)	-0.60 (0.25) * DOC	4.32 (1.10) * pH	--	92.7	< 0.001
Percymoorensis marmoratis (leech)	-8.05 (2.71)	1.69 (0.50) * pH	--	--	75.2	< 0.001
Bullfrog	-9.45 (1.58)	0.36 (0.14) * lnAREA	1.20 (0.47) * lnSO_4	-0.41 (0.16) * lnAl	74.5	< 0.001
Common Merganser pairs	-3.20 (1.36)	0.05 (0.01) * lnAREA	-6.40 (1.91) * lnAl	0.36 (0.21) * pH	74.5	< 0.001
Common Loon broods	-4.03 (0.63)	0.64 (0.07) * lnAREA	0.25 (0.09) * pH	--	69.8	< 0.01
Common Loon broods (Kejimkujik)	-12.64 (2.22)	1.00 (0.15) * lnAREA	1.36 (0.36) * pH	--	--[b]	<0.05
Common Loon pairs	-4.64 (0.56)	1.17 (0.06) * lnAREA	0.30 (0.09) * pH	--	88.9	< 0.001

[a] Variable definitions: Al = aluminum concentration ($\mu g \cdot L^{-1}$), AREA = lake area (hectares), DOC = dissolved organic carbon ($mg \cdot L^{-1}$), SO_4 = sulphate concentration ($mg \cdot L^{-1}$), LONG = longitude (degrees), TWStemp = average annual temperature in TWS (°C), TWSelev = elevation of TWS (m A.S.L.), ln'VAR'1 = ln + 1 transformation.

[b] Performed with a statistical package that did not give concordance

Note: The higher the Kappa statistic, the more significant the equation.

observed in any of the years of observation. As with the macroinvertebrate analysis, we divided the water bodies into those containing (n=343) and lacking fish (n=277) because earlier studies had suggested that the presence of fish can influence waterbird use of water bodies (e.g. McNicol and Wayland 1992).

For fishless water bodies, we obtained the following regression:

$$\text{Richness} = 0.75(0.22) + 0.33(0.04)*\ln(\text{AREA}+1) - 0.01(0.01)*\text{pH}^2 + 0.22(0.07)*\ln(\text{DOC}+1) + 0.57(0.39)*\ln(\text{Al}+1) - 0.02(0.01)*\text{DEPTH},$$
$$(p<0.001,\ r^2 = 0.20,\ n=277).$$

In fact, for the leech, pH was the only significant explanatory variable, which was consistent for this group's reported sensitivity to pH (Eilers *et al.* 1984, Bendell and McNicol 1991). For both the mayfly and snails, increased concentrations of Al reduced the probability of their occurrence in water bodies; snails were also less likely to occur under higher SO_4 conditions (Figure 6). For example, in the top left graph, the probability of the occurrence of snails in small Ontario water bodies increases dramatically between pH 5.0 -7.0 when concentrations of Al and SO_4 are held constant at mean values for the data (Figure 6, middle curve). Under low Al conditions, the probability of snail occurrence is higher at all pHs (top curve), whereas at high Al the probability of occurrence is lower at all pHs. *Caenis* was more commonly found in higher DOC water bodies, whereas the amphipod occurred more commonly in clear (i.e. low DOC) water bodies. Logistic model concordance, or percent rank correlation (Snedecor and Cochran 1989), was at least 75% (complete concordance is 100%) in all of these analyses, and models were highly significant.

Bullfrogs were more commonly found on larger water bodies with higher SO_4 and lower Al concentrations (Table 4). In a subsequent analysis, we included "site" (Algoma, Muskoka or Sudbury study areas) as a coded variable, and found that bullfrogs were more commonly caught at Sudbury, and, after accounting for this, bullfrogs were more likely to be caught on higher pH water bodies (CWS (Ontario) unpublished data).

For waterbirds, the occurrence of breeding merganser pairs was significantly more probable on larger, high pH water bodies with low aluminum concentrations (Table 4). Although we lacked sufficient data to test relationships for merganser broods, Parker *et al.* (1992) found common merganser broods more often on high pH water bodies in southwest New Brunswick. Both pH and area entered positively for loon pairs and broods of large young, indicating that the occurrence of breeding pairs (i.e. breeding attempts) and of young (i.e. breeding success) were significantly higher on large, high pH water bodies in Ontario. In a similar analysis for Kejimkujik National Park, Nova Scotia, the occurrence of loon broods was also higher on large, high-pH water bodies (Table 4). Pairs were also more common on large lakes of lower DOC, possibly because darker waters are less suitable for loons to view their prey, but perhaps linked to higher mercury (Hg) methylation and bioaccumulation in darker lakes (Burgess *et al.* 1996). In general, these results are consistent with earlier results from Ontario (Alvo *et al.* 1988), Québec (DesGranges 1989), and Nova Scotia (Kerekes *et al.* 1994), and clearly implicate the negative effects of pH on higher trophic levels in large, aquatic systems. For these fish-eating birds, the mechanism by which pH affects these species is probably to reduce or eliminate fish populations in water bodies, thereby harming the prey resource necessary for successful reproduction (Kerekes 1990, Kerekes *et al.* 1994, McNicol *et al.* 1995b).

Clearly, fish populations are negatively influenced by increasing acidity (Section 4.2.3), as has been shown in many studies. Unfortunately, we lacked fish community data for many of the lakes used in the analyses above. However, Blancher *et al.* (1992) used an approach similar to that above to develop logistic equations relating the presence of loons and mergansers to lake characteristics in the Algoma and Sudbury regions of Ontario. They incorporated a fish presence/absence variable in their analyses, and they found that fish entered positively for breeding pairs and broods of both species. In fact, common merganser broods were never found on any of the 67 fishless lakes in their study. Similarly, Kerekes (1990) was able to show that nutrient levels in acid-stressed, nutrient-poor Kejimkujik lakes was related to fish production, and this was related to loon occurrence on lakes. Hence, the relationship we found between piscivorous bird breeding success and the physical and chemical characteristics of lakes is probably manifested through food chain effects. These include acidity-related chemical influences on fish populations, as well as recent concerns about elevated Hg levels in loon blood and feathers from Atlantic sites (including Kejimkujik) which point to uptake of Hg in these stressed systems as having an indirect effect on reproductive success.

4.3.4 Summary

Collectively, these analyses on zooplankton, fish, macroinvertebrates, amphibians and waterbird species point conclusively to the fact that reduced lake pH has a deleterious effect (whether direct or indirect) on certain aquatic taxa at several trophic levels and could undoubtedly have affected their distributions in southeastern Canadian water bodies. If species-specific models are used in conjunction with species richness models, this provides a powerful tool in determining the effects of pH changes on particular species at different trophic levels as well as overall species diversity and ecosystem health. If a particular taxon disappears from a lake, a link has been removed within the food web and therefore changes would ensue in food web dynamics (Locke and Sprules, 1994). Other variables which are important in determining the distribution of particular species (longitude, lake size, elevation) help us to delimit a geographic and physical range for that species. This would allow for the determination of more specific pH thresholds to be determined based on these range limitations. Still other factors that are consistently important, such as DOC and annual air temperature, could change over time just as pH could. Therefore, knowing the relationship between species occurrence and these variables, including their interactions, can help isolate the effects of pH alone given the specific models presented in this assessment. Instead of attempting to divide groups of water bodies into different categories based on certain characteristics (i.e. DOC groups), these logistic models can be used to determine the boundaries for those characteristics by choosing a minimum acceptable probability of finding a species within a lake and thereby determining minimum acceptable levels for pH change.

4.4 Species richness relationships

4.4.1 Introduction

Species richness or diversity is a strong indicator of ecosystem health (Schindler *et al.* 1989, Baker *et al.* 1990, RMCC 1990, Biodiversity Science Assessment Team 1994). The variables listed in the Section 4.1 were tested in stepwise, multiple regression analyses (RS/1 1990, SYSTAT, SAS 1993) to determine the factors which most affect the species richness of four taxonomic groups: fish, zooplankton, macroinvertebrates and waterbirds (reliable data for several amphibian species were insufficient, and thus amphibians were not included as a group). Fish and zooplankton results, and macroinvertebrate and waterbirds results, are presented individually below because analyses for these groups were carried out separately and the results apply to different geographic regions of southeastern Canada and Ontario (see maps, Figure 1).

Macroinvertebrate community composition and species richness is affected by many factors, including pH (Schindler *et al.* 1989, Griffiths and Keller 1992). Numerous studies have demonstrated that the presence of fish in water bodies has a strong influence on macroinvertebrate abundance and community structure, while chemical conditions appear to influence species diversity (Eriksson *et al.* 1980, Schell and Kerekes 1988, McNicol and Wayland 1992, Mallory *et al.* 1994a, Stephenson *et al.* 1994, Bendell and McNicol 1995, McNicol *et al.* 1995d).

Unlike most fish, zooplankton, and macroinvertebrates, many aquatic and semi-aquatic birds (e.g. swallows, dippers, herons, ospreys, loons, waterbirds) are indirectly affected by acidification, sometimes through toxic effects (e.g. Scheuhammer 1991), but more often through changes in the food chain (i.e. fish and macroinvertebrate communities, Longcore *et al.* 1993). Among waterbirds, some species (e.g. common goldeneye, *Bucephala clangula*) preferentially exploit acidified, fishless water bodies because of the abundance of acid-tolerant macroinvertebrate prey (Mallory *et al.* 1994b, McNicol and Wayland 1992), while others, such as loons and common mergansers, prefer higher pH water bodies with fish (see above, as well as McNicol *et al.* 1995c). Where fish are present, insectivorous waterbird species prefer water bodies that have non-competitor fish species [i.e. fish such as northern redbelly dace (*Phoxinus eos*) whose gapes are too small to effectively feed on the same macroinvertebrates as the waterbirds]. These water bodies can produce enough macroinvertebrate prey for waterbirds (McNicol and Wayland 1992).

Several models predicting species richness within a taxon have been developed which incorporate pH as a significant determinant. Characteristics which have been discussed earlier in species-specific models, often play a significant role in determining local and regional species richness as well. Geographic location, elevation, water clarity, climate, pH, and lake size appear to be consistent variables in many zooplankton, fish and waterbird models (Kelso *et al.* 1990, Minns *et al.* 1990, Keller and Yan 1991, Blancher *et al.* 1992, Locke *et al.* 1994, Pierce *et al.* 1994, Mandrak 1995). In addition to species richness, biotic interactions, such as the number of predatory and competitive links in food web structure, have also been connected to lake pH and other variables (Havens 1993, Locke and Sprules 1994). Generally species richness declines below pH 6.0 to 5.0 in several taxonomic groups and several regions (Seigfried 1988, Schindler *et al.* 1989), but the rate of decline could be determined by other factors. Local or lake characteristics, such as pH, are usually more important in determining lake species richness than regional ones. In a model developed by Minns (1989), lake area, tertiary watershed species richness, elevation, sampling year, pH, and mean lake depth influenced fish species richness in Ontario. Whereas, pH and SO_4 concentrations (and lake morphometry to some extent) were shown to affect species assemblages of zooplankton in Québec water bodies (Pinel-Alloul *et al.* 1990).

4.4.2 Statistical Methods

For the statistical analyses of all taxonomic groups, we used step-wise, linear, multiple regression analyses (with the same independent variables as described in Section 4.1) to determine relationships between species richness and physical and chemical characteristics of the habitat. Only variables which explained a significant portion of the variance in species richness were included to a maximum of three to four variables. As in logistic analysis, macroinvertebrates and waterbird data were available only for Ontario water bodies, but because the water bodies span a broad range of physical and chemical characteristics, the results should be broadly applicable. Also, because these data were collected from water bodies clustered in certain regions (Algoma, Muskoka and Sudbury; McNicol *et al.* 1996), we ran the analyses with and without coded variables to examine whether inherent characteristics of these sites influenced species richness.

4.4.3 Results & Discussion

Zooplankton and Fish

In this assessment, lake area and DOC, in addition to pH, contributed significantly to the variability in species richness for both fish and zooplankton in specific areas or datasets, and all of southeastern Canada or the pooled data (Figure 7). One problem with most significant determinants of species richness, especially when performing multiple regression analysis, is that several key variables can be highly correlated in some areas, such as lake area and pH (Matuszek and Beggs 1988). As a result, these variables could be excluded from regression equations for more localized datasets, even though they may be important. In comparing the zooplankton species richness at different spatial scales or in different regions, (the other significant variables determined in statistical analyses were held constant at average levels), the rate of change in richness (slope) and the minimum species richness (intercept) vary between datasets (Figure 7). For example, zooplankton richness increases at a faster rate as pH increases in Sudbury area lakes then in water bodies sampled in northeastern and central Ontario. The species richness responses in Ontario and Nova Scotia are very similar for both zooplankton and fish. The rate of increase in fish richness is greater in Québec, however, the maximum number of fish species in all three provinces are roughly equivalent. The rate of increase of fish species richness in small water bodies in Ontario is comparable to larger water bodies in Ontario and Nova Scotia, although the actual number of fish species is always lower in small ones.

When the individual datasets were pooled and analysed, a general model incorporating geographic variables was obtained. The geographical variable, longitude, was important in determining species richness in southeastern Canada models, indicating a gradient from Ontario to the Atlantic provinces associated with glaciation history and freshwater species dispersal (Minns and Moore 1995, Mandrak 1995). However, longitude only affected species richness secondarily, since pH and either lake area or DOC were the primary determinants of species richness in zooplankton and fish in southeastern Canada (Figure 8). Zooplankton richness was highest (according to a linear model) at high pHs and high DOC levels. Fish species richness averaged 10 species per lake in 1000-hectare water bodies with higher pHs. Any predictions of species richness within a lake that are based on these relationships should

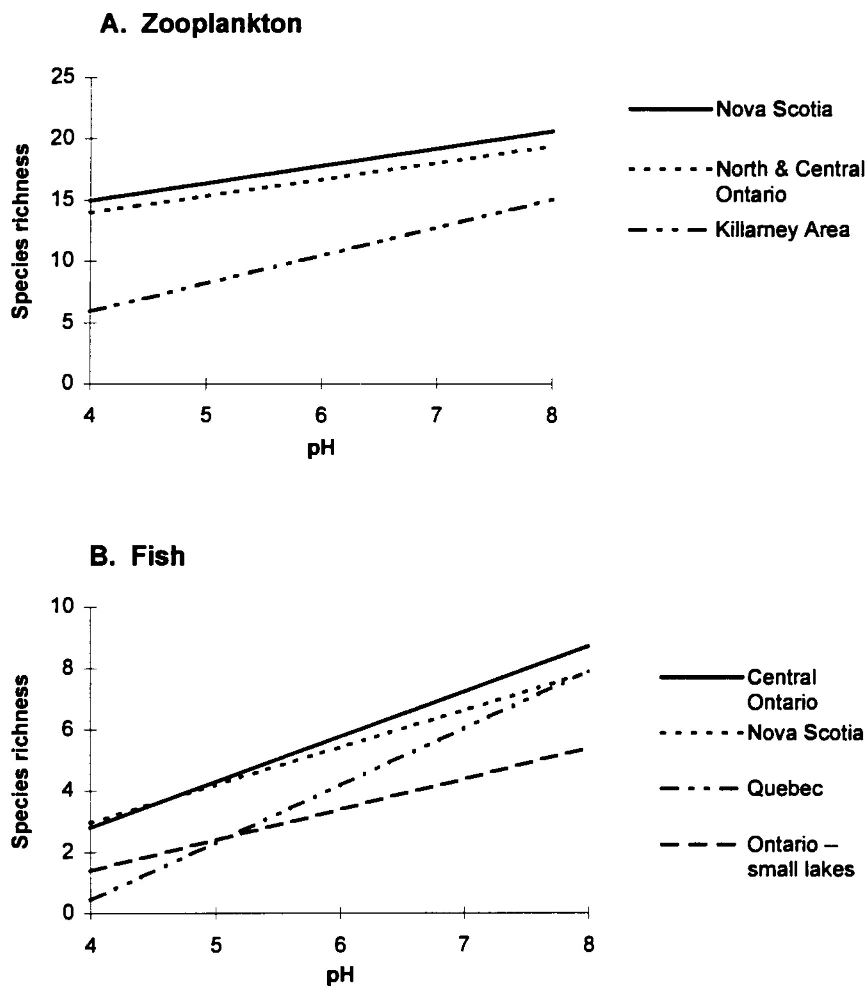

Figure 7: Species richness (total number of species) relationships with pH are shown for separate (A) zooplankton datasets and (B) fish datasets in southeastern Canada. Other significant variables were held constant at the average level for that variable in each dataset. See Table 5 for multiple regression equations used.

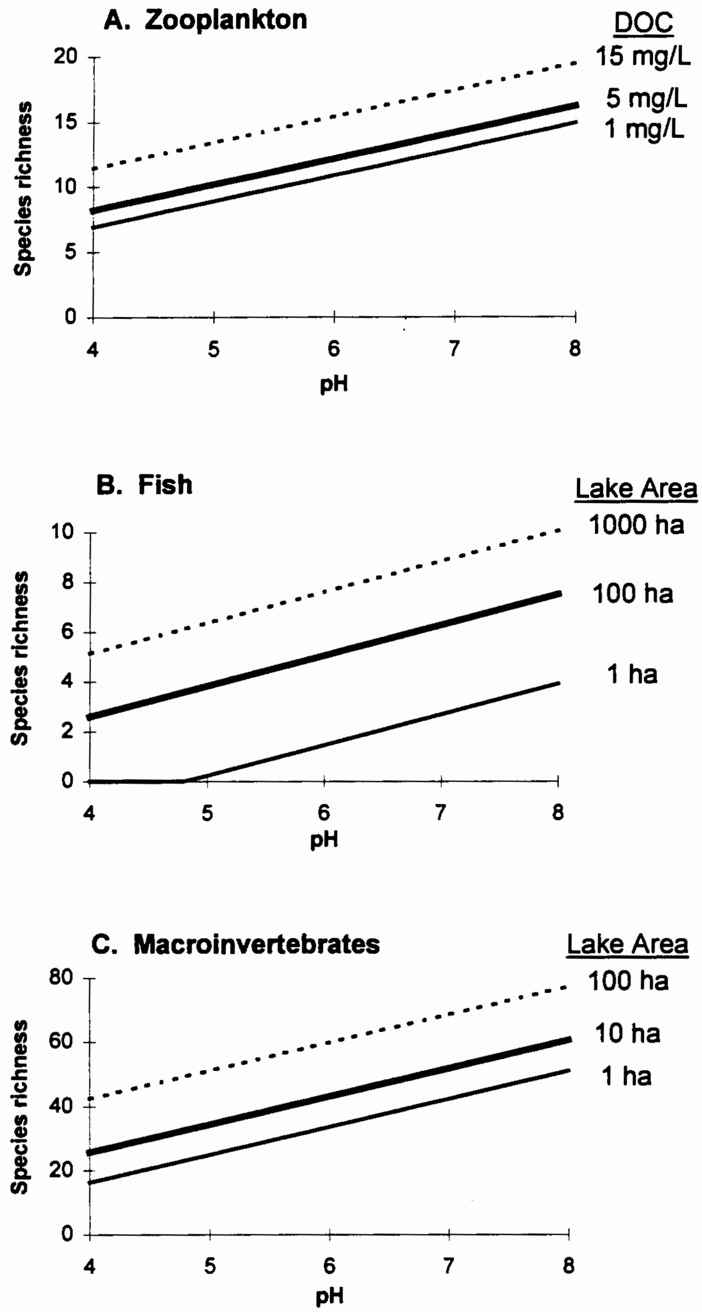

Figure 8: Species richness (total number of species) and pH linear regression models for (A) zooplankton species richness and (B) fish species richness in southeastern Canada or (C) macroinvertebrate species richness in small Ontario water bodies. The other significant variables in the regression equations were held constant at an average level [A. Longitude = 75 degrees (between Ottawa and Montreal) and lake area = 100 ha; B. Longitude = 75 degrees and DOC = 5 $mg{\bullet}L^{-1}$; C. DOC = 5 $mg{\bullet}L^{-1}$] with the exception of the second-most important variable in the equation which was set at three levels to generate a confidence area for that variable in conjunction with pH. See Table 5 for multiple regression equations.

be a range in average species richness and not an absolute number of species. On average two to three species should be added and subtracted to each prediction of species richness.

Macroinvertebrates

For this evaluation, analyses of macroinvertebrate taxonomic richness in relation to lake characteristics were restricted to small water bodies from three regions of central and northeastern Ontario (Algoma, Muskoka, Sudbury; McNicol *et al.* 1995d). Data were available on 159 taxa (genera or species) from 66 water bodies ranging in size (1.3 - 14.2 ha) and chemical characteristics (e.g. pH 4.25 - 7.02, DOC 0.98 - 15.91 mg/L, TP 3.08 - 17.73 ug/L, Al 9.7 - 470.6 ug/L, SO_4 3.49 - 10.66 mg/L or see Appendix 1). Region was included as a coded variable, and mixed model ANOVA/regression was used (PROC GLM; SAS 6.10 1993) to determine which physical and chemical characteristics influenced taxonomic richness. Becausethe presence of fish strongly influences the abundance and composition of The equation developed for water bodies containing fish was not statistically significant. These results support the supposition that the presence of fish must be considered when analyzing macroinvertebrate communities in relation to lake chemistry (McNicol *et al.* 1995d), particularly if communities include both nektonic and benthic organisms (McNicol and Wayland 1992). Furthermore, they demonstrate that, in the absence of fish predators, low lake pH has a negative impact on macroinvertebrate taxonomic richness, where many acid sensitive species disappear (see Section 3), leaving only acid-tolerant ones (Eriksson *et al.* 1980, Schindler *et al.* 1989, McNicol *et al.* 1995d). Using the above equation, a 10-hectare, fishless lake in Ontario with 5 $mg{\bullet}L^{-1}$ DOC would probably experience a 30% loss of taxonomic diversity with a reduction from pH 6 to 4.5. Loss of biodiversity, particularly key prey species, has consequences for higher levels in the food chain (e.g. fish, amphibians, waterbirds) that rely on these invertebrate prey as a food source (reviewed in Longcore *et al.* 1993). Consequences of the loss of preferred prey include:

1. Changes in the distribution, abundance and fecundity of various predatory species.

2. Potential calcium stress for predators (since many acid-sensitive invertebrate species are also calcium-rich).

3. Increased exposure of predators to trace metals, which are often higher burdens in macroinvertebrates from more acidic water bodies and which may accumulate up the food chain (see Scheuhammer 1991, Scheuhammer *et al.* 1997).

Thus, not only does lake acidification result in a loss of diversity among macroinvertebrates, but it can lead to deleterious effects up the food chain as well.

Waterbirds

As with the other taxonomic groups, species richness at a high trophic level may be influenced by acidification. For waterbird species richness analyses, we used data on the number of indicated breeding pairs of waterbird species collected from aerial surveys conducted during

Table 5: Fish and zooplankton species richness (number of species) equations for different regions within southeastern Canada and all regions (pooled datasets) combined. The equations are the results of stepwise linear regression analysis.

REGION	TAXON	DATASET	EQUATION [a]	N, R, SDR, p *
ONTARIO	FISH	CWS	-6.253(0.632) + 0.589(0.090) * **lnlkar1** + 1.000(0.096) * **pH** + 0.518(0.174) * **lndoc1**	523, 0.53, 1.60, <0.001 (all water bodies--incl. fishless)
			-3.929(1.230) + 0.719(0.181) * **pH** + 0.417(0.128) * **lnlkar1** + 0.814(0.296) * **lndoc1**	262, 0.34, 1.73, <0.001 (water bodies with fish only)
		MESO	-8.834(2.690) + 0.817(0.117) * **lnlkar1** + 1.483(0.423) * **pH** + 1.077(0.657) * **lndoc1**	144, 0.60, 2.45, <0.001
		OMNR	-29.688(3.357) + 1.738(0.255) * **lnlkar1** + 5.195(0.618) * **pH**	54, 0.88, 2.35, <0.001
	ZOOP	OMEE	8.755(1.454) + 1.336(0.163) * **pH** - 1.708(0.227) * **lnSO4** + 0.488(0.075) * **lnlkar1**	543, 0.60, 2.74, <0.001
		UofT	-14.163(3.731) + 1.180(0.378) * **lnlkar1** + 2.276(0.740) * **pH** + 1.104(0.344) * **DOC**	42, 0.71, 2.24, <0.001
QUÉBEC	FISH	QLS	-8.485(1.717) + 1.871(0.248) * **pH** + 0.799(0.326) * **lndoc1**	250, 0.43, 1.90, <0.001
	ZOOP	UofM	No suitable model.	
NOVA SCOTIA	FISH	DFO	-5.025(2.885) + 1.228(0.430) * **pH** + 0.666(0.194) * **lnlkar1**	43, 0.52, 1.59, 0.002
		DFO/CWS	No suitable model.	
	ZOOP	DFO	No suitable model.	
		DEL	5.712(4.482) +0.723(0.255) * **DOC** + 1.403(0.707) * **pH**	19, 0.62, 3.15, 0.019
SOUTH-EASTERN CANADA	FISH	ALL	-10.472(1.345) + 1.227(0.103) * **pH** + 1.113(0.051) * **lnlkar1** + 0.032(0.014) * **long** + 0.434(0.167) * **lndoc1**	1042, 0.65, 2.16, <0.001
	ZOOP	NIS	-11.195(3.249) + 0.223(0.022) * **long** + 1.821(0.402) * **pH** + 0.744(0.217) * **TWStemp**	258, 0.59, 4.23, <0.001
		ALL	-0.039(0.009) * **long** + 0.311(0.078) * **lnlkar1** + 2.012(0.137) * **pH** + 0.325(0.052) * **DOC**	823, 0.96, 3.42, <0.001

[a] List of variable abbreviations: lnlkar1 = ln (lake area + 1), hectares; lndoc1 = ln (dissolved organic carbon + 1), mg•L^{-1} ; long = longitude (degrees); lnSO4 = ln (sulphate conc.), mg•L^{-1} ; TWStemp = average annual temperature in TWS (° C);

* N = number of water bodies used in regression, R = multiple regression coefficient of variation, SDR = standard deviation of regression, p = significance level.

the spring (mid-May) nest building period in the Algoma, Muskoka, and Sudbury regions of Ontario (n=629 water bodies; 1988-1995). Eight species commonly nest on these water bodies (McNicol *et al.* 1987a, McNicol *et al.* 1995c), and a range of 0-8 species were recorded as breeding on any lake. Each lake was entered into the analysis once, and was scored as having a species present if that species was observed in any of the years of observation. As with the macroinvertebrate analysis, we divided the water bodies into those containing (n=343) and lacking fish (n=277) because earlier studies had suggested that the presence of fish can influence waterbird use of water bodies (e.g. McNicol and Wayland 1992).

For fishless water bodies, we obtained the following regression:

$$\text{Richness} = 0.75(0.22) + 0.33(0.04)*\ln(\text{AREA}+1) - 0.01(0.01)*\text{pH}^2 + 0.22(0.07) *\ln(\text{DOC}+1) + 0.57(0.39)*\ln(\text{Al}+1) -0.02(0.01)*\text{DEPTH},$$
$$(p<0.001,\ r^2 = 0.20,\ n=277).$$

For water bodies containing fish, the following regression was obtained:

$$\text{Richness} = 7.21(2.99) + 0.20(0.03)*\ln(\text{AREA}+1) - 1.99(0.99)*\text{pH} + 0.14(0.08)*\text{pH}^2 + 0.41(0.07)*\text{Algoma} + 0.21(0.07)*\text{Muskoka},$$
$$(p<0.001,\ r^2 = 0.21,\ n=343).$$

In both fishless water bodies and those with fish, waterbird species richness is higher on larger water bodies, but it declines slightly with increasing pH. The latter result is not surprising, since certain waterbird species that are common in these regions often use more acidic water bodies during the breeding season because of reduced competition with fish for invertebrate prey (see Eriksson 1983, McNicol and Wayland 1992, Mallory *et al.* 1994a,b). In water bodies containing fish, the inclusion of coded variables for each region indicates that the historically-damaged Sudbury region has significantly lower species diversity than either Muskoka or Algoma. Perhaps the most important result is that the explained variation in each analysis is low at approximately 20% (about half of which is attributable to lake area), which indicates that many other factors (e.g. territoriality, annual survival, and site fidelity) influence waterbird species diversity in these regions. Therefore, because of many confounding factors, lake chemistry parameters that are strongly influenced by acidic deposition (pH, Al, SO_4) appear to be poor predictors of waterbird species richness at this trophic level, consistent with results from some other recent studies (DesGranges and Houde 1989, Elmberg *et al.* 1994, Fox and Bell 1994).

5.0 General Summary

The importance of the long-range transport of air pollutants (particularly acidic deposition) as a continuing threat to biota in southeastern Canada cannot be overstated; acidic deposition and nutrient loading were ranked as the most serious threats among environmental pollutants at the ecoregion level to biodiversity (Biodiversity Science Assessment Team 1994). The empirical evidence provided in this Assessment supports previous work, and also aids in further quantifying the effects of other variables, such as dissolved organic carbon content, lake size, and geographic position, on biotic richness and the occurrence of specific pH-sensitive species in eastern Canada.

Based on the theoretical proportional response of taxonomic groups to pH (Section 3.3), species-specific models (Section 4.2), and empirical species-richness models (Section 4.3) developed in this report, a better understanding of the interactions between lake characteristics and the species response to pH has been achieved. The biotic models developed can be linked to existing chemical and hydrological models that predict the pH changes in water bodies based on deposition scenarios (see Sections 1.2). However, there are some limitations to the predictability and applicability of the empirical analysis presented here. Clearly, a variety of other factors (chemical, physical and biological) can influence the presence of certain species (see Section 4.0). Other variables which were not sampled in most datasets, such as food abundance, could have more significant effects on species richness and occurrence than the variables available. Generally, only one trophic level was assessed in each dataset and other biotic effects, such as the presence of predators, should be measured. Metal concentration data in water bodies was poorly represented in most datasets and could be an important, but correlated, factor to consider when attempting to delimit changes in biotic response based on pH. Another assumption which was made for this assessment was that temporal differences between datasets would not influence the results obtained. However, precautions were taken to not include the same lake twice. Given these limitations, pH was a consistent and significant variable in determining the species richness and occurrence of selected taxa in southeastern Canada.

In the 1997 Assessment, lake pH predictions were made for four clusters in southeastern Canada by using projected acidic deposition in those areas based on different IAM scenarios (See section 1.2). The pH of each lake determined from these scenarios was used in the general southeastern model given in this report to predict the species richness of fish in the cluster lakes. In comparing the values obtained for these lakes to the original fish species richness (based on predicted original pH for each lake) and extrapolating the richness predictions to the whole resource at risk in those areas, one is able to determine the loss in fish populations under the different scenarios. Even given the inherent gaps alluded to in the datasets, such as inconsistent data collection, when all the biotic models are applied in a similar fashion as the fish richness model, we should get a good indication of how biodiversity as a whole is affected by acidic deposition as long as we identify the limitations of our predictions.

The next steps are to identify the current knowledge gaps and future directions of acid rain research and monitoring after determining how biota respond to chanages in lake pH. Many of the knowledge gaps identified in the 1990 Assessment have been addressed, but some remain and yet others have become apparent since then. It is clear that acidification is not an issue unto itself since interactions with other environmental stressors, including climate change/variability, toxics (e.g. mercury), UV-b, catchment nutrient export, to name a few, have major implications for the chemical and biological status and recovery of aquatic ecosystems. While efforts to understand and address these interlinked environmental issues have expanded in recent years, research and monitoring to define the pathway and rate of recovery in aquatic ecosystems damaged by acidic precipitation have been reduced despite the fact that we know very little about recovery pathways and trajectories. Also, it is clear that methods for estimating the recovery (or loss) of biological populations in water bodies would be vastly improved if factors other than pH (e.g. DOC, lake size, etc.) were taken into account, together with a better understanding of natural variability in aquatic communities at low pH. Altogether, the development and application of time-dependent models to predict the rate of chemical and biological recovery (or degradation) remains incomplete and thus is limiting the effectiveness of empirical biological models. In addition, the regional impact of episodic acidification on aquatic biota remains largely unexplored. Taking all this information together, it is clear that much remains to be done to ensure protection of our invaluable aquatic resources from the effects of acidification.

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8.0 Appendices

Appendix 1: A list of all the physical and chemical variables which were supplied with each of the fish and zooplankton datasets used in statistical analyses in the report. The means and standard deviations (in parentheses) are shown. For a description of the source of each dataset see Table 1 in the report.

	Fish (Waterbird, Macroinvertebrates[a]) Datasets						
Attribute	**QLS**	**CWS[a]**	**NIS**	**OMNR**	**DFO**	**MESO**	**CWS/DFO**
[Al] ($\mu g \cdot L^{-1}$)	140.32 (85.06)	157.79 (144.22)	99.90 (77.50)[b]			109.58 (60.45)	
[Ca] ($mg \cdot L^{-1}$)	97.54 (51.80)	2.68 (2.05)	86.28 (137.90)	2.14 (0.40)		126.44 (58.04)	0.97 (0.78)
[Cd] ($\mu g \cdot L^{-1}$)			0.20 (0.10)[b]			0.87 (0.29)	
[Cl] ($mg \cdot L^{-1}$)	56.40 (58.88)	0.39 (1.35)	59.54 (60.46)			30.29 (50.86)	
[Cu] ($\mu g \cdot L^{-1}$)		3.98 (10.18)	0.60 (0.50)[b]			1.21 (0.82)	
[DIC] ($mg \cdot L^{-1}$)	1.05 (0.79)						
[DOC] ($mg \cdot L^{-1}$)	6.47 (2.56)	6.43 (3.26)		2.79 (0.97)		5.36 (1.98)	6.93 (3.34)
[Fe] ($\mu g \cdot L^{-1}$)		178.72 (221.12)	59.50 (71.10)[b]			507.87 (511.30)	
[H] ($\mu eq \cdot L^{-1}$)	3.23 (5.07)						
[HCO_3] ($mg \cdot L^{-1}$)			81.07 (171.01)				
[Hg] ($\mu g \cdot L^{-1}$)						0.02 (0.01)	
[K] ($mg \cdot L^{-1}$)		0.31 (0.15)	5.07 (3.16)			8.97 (3.97)	0.19 (0.11)
[Mg] ($mg \cdot L^{-1}$)	45.24 (23.61)	0.64 (0.40)	40.41 (49.26)			59.12 (24.48)	0.20 (0.15)
[Mn] ($\mu g \cdot L^{-1}$)		48.42 (42.12)	59.00 (51.00)[b]			60.05 (47.29)	
[Na] ($mg \cdot L^{-1}$)		0.77 (0.86)	64.20 (48.83)			48.44 (42.91)	0.41 (0.27)
[NH_3] ($\mu g \cdot L^{-1}$)	147 (111)[b]	55.35 (68.12)				70.52 (61.70)	
[Ni] ($\mu g \cdot L^{-1}$)		9.46 (6.46)	0.50 (0.50)[b]			1.74 (4.14)	
[NO_x] (mg/L)	1.31 (1.43)	0.02 (0.02)				1.33 (7.19)	
[Organic acid] ($\mu eq \cdot L^{-1}$)						45.43 (15.96)	
[Pb] ($\mu g \cdot L^{-1}$)		0.74 (0.78)				6.39 (5.96)	
[PO_4] ($\mu g \cdot L^{-1}$)						10.37 (11.98)	
[SiO_2] ($mg \cdot L^{-1}$)		2.10 (1.71)				1.62 (1.03)	
[SO_4] ($mg \cdot L^{-1}$)	82.27 (44.71)	7.10 (2.39)	58.67 (29.65)	7.34 (1.21)		130.26 (34.63)	2.32 (1.73)
[TIC] ($mg \cdot L^{-1}$)		0.88 (1.67)				1.09 (0.54)	
[Total C] ($mg \cdot L^{-1}$)						6.40 (2.85)	
[Total N] ($mg \cdot L^{-1}$)		0.46 (0.32)				0.47 (0.27)	
[Total P] ($\mu g \cdot L^{-1}$)		9.35 (12.25)				12.93 (11.74)	
[Zn] ($\mu g \cdot L^{-1}$)		9.52 (6.02)	1.60 (0.90)[b]			8.42 (5.68)	
Acidity (mg $CaCO_3 \cdot L^{-1}$)					6.58 (4.98)		

Attribute	Fish (Waterbird, Macroinvertebrates[a]) Datasets						
	QLS	CWS[a]	NIS	OMNR	DFO	MESO	CWS/DFO
Alkalinity ($\mu eq \cdot L^{-1}$)	51.55 (55.90)	36.31 (106.41)	77.52 (172.07)	2.01 (3.81)	2.07 (2.55)	53.32 (70.44)	0.89 (1.59)
Bedrock area (ha)						73.30 (333.46)	
Bog area (ha)						171.47 (477.91)	
Colour (Hazen units)	37.26 (22.57)		36.09 (32.52)		46.79 (28.00)	28.64 (20.96)	
Conductivity ($uS \cdot cm^{-1}$)		29.60 (13.35)	21.29 (22.11)	29.32 (7.84)	40.45 (33.77)	23.65 (10.40)	36.78 (15.21)
Depth--mean (m)				8.91 (12.73)			2.74 (2.84)
Depth--max (m)	8.09 (8.73)	4.95 (4.97)	9.59 (12.46)	20.71 (15.71)			7.75 (6.18)
Depth--sample (m)					10.07 (8.98)	3.33 (1.88)	
Depth--station (m)						9.09 (6.57)	
Dissolved O_2 ($mg \cdot L^{-1}$)					5.90 (3.14)	5.54 (2.16)	
DistNW* (m)		261.96 (207.66)					
Forested shoreline (m)		573.23 (922.90)					
Hardness (mg $CaCO_3 \cdot L^{-1}$)					6.70 (4.15)		
Lake area (ha)	60.19 (35.16)	7.19 (10.68)	131.69 (175.05)	101.36 (134.09)	100.18 (156.43)	199.17 (476.46)	66.66 (79.84)
Lake elevation (m A.S.L.)	413.29 (116.74)	414.78 (71.68)	285.20 (165.71)	222.59 (32.39)			
Lake order				1.82 (0.72)		2.60 (2.33)	
Lake perimeter (km)		1630 (1525)		8.01 (9.52)			
Lake volume (10^4 m^3)				766.49 (1133.78)			
No. of instreams		1.02 (1.25)					
No. of islands		0.63 (1.55)					
No. of outstreams		1.04 (0.73)					
No. wetlands in 500 m^2		2.39 (1.52)					
pH	5.75 (0.51)	5.65 (0.74)	6.11 (0.78)	5.68 (0.55)	5.89 (0.61)	5.77 (0.59)	5.68 (0.78)
Riparian area (m^2)		21742 (22753)					
Riparian perimeter (m)		864 (1098)					
Riparian shoreline (m)		429 (410)					
Secchi depth (m)			3.01 (1.90)	4.94 (3.23)	3.14 (1.81)	3.14 (1.64)	
Temperature (°C)					21.11 (2.68)	15.14 (5.18)	
Watershed area (ha)	12005 (4781)		1353 (2741)			2256 (5936)	
WTRAB[c] (ha)						459 (1609)	

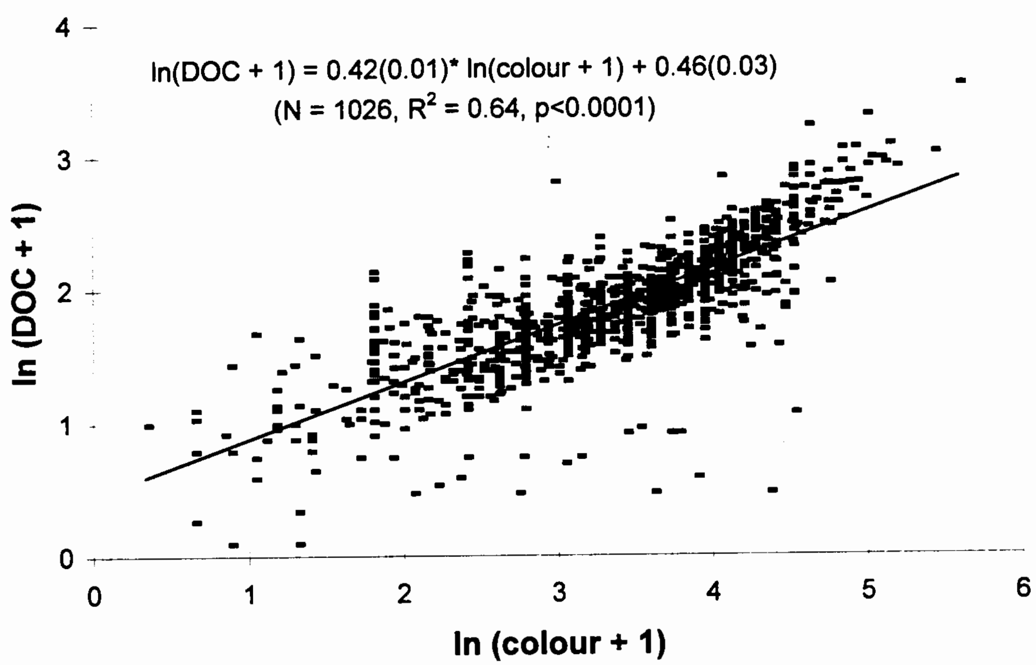
Appendix 1: cont'd

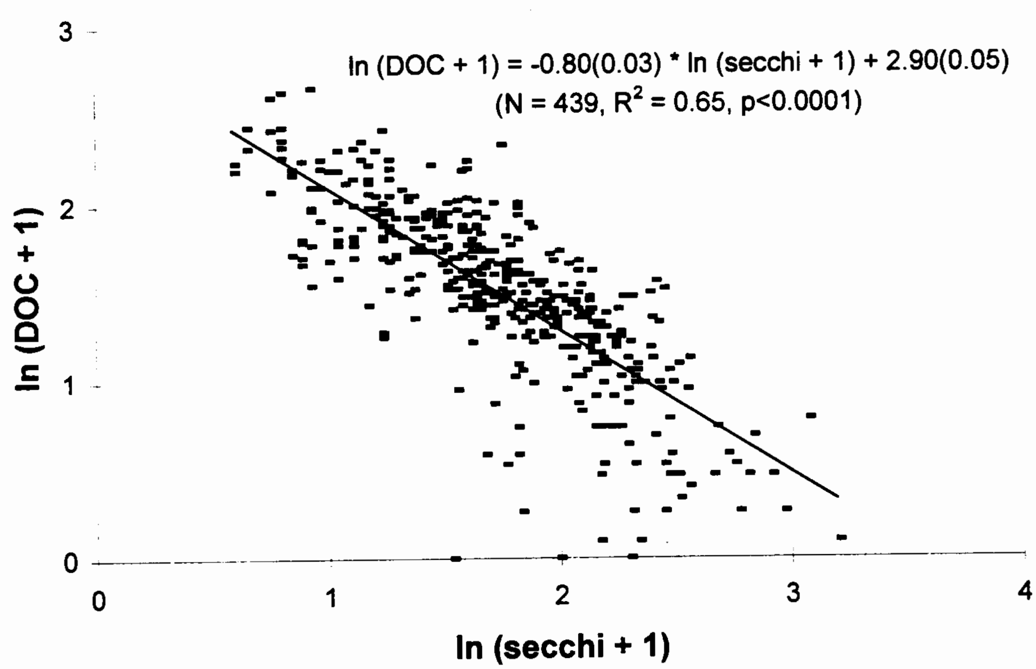
Attribute	Zooplankton Datasets UofM	NIS	DFO	DEL	UofT	OMEE
[Al] ($\mu g \bullet L^{-1}$)	104.48 (60.44)	104.2 (96.1)[b]				96.82 (120.36)
[Ca] ($mg \bullet L^{-1}$)	125.59 (150.84)	161.41 (177.70)			3.50 (2.33)	3.58 (2.88)
[Cd] ($\mu g \bullet L^{-1}$)	0.24 (0.16)	0.2 (0.1)[b]				
[Chlorophyll *a*] ($\mu g \bullet L^{-1}$)					1.27 (0.69)	
[Cl] ($mg \bullet L^{-1}$)	11.76 (4.46)	48.24 (50.77)			2.47 (10.00)	0.84 (3.54)
[Cu] ($\mu g \bullet L^{-1}$)	1.41 (1.37)	1.5 (1.0)[b]			14.73 (64.12)	3.53 (20.01)
[DIC] ($mg \bullet L^{-1}$)	1.52 (1.31)					
[DOC] ($mg \bullet L^{-1}$)	5.94 (2.03)					4.06 (2.22)
[Fe] ($\mu g \bullet L^{-1}$)	99.87 (82.11)	64.6 (84.1)[b]			70.62 (87.54)	105.74 (201.11)
[K] ($mg \bullet L^{-1}$)	6.62 (3.52)	3.69 (2.67)			0.45 (0.17)	0.41 (0.20)
[Mg] ($mg \bullet L^{-1}$)	43.47 (45.63)	63.99 (63.99)			1.18 (0.64)	1.00 (1.05)
[Mn] ($\mu g \bullet L^{-1}$)	11.78 (7.63)	4.6 (3.8)[b]				38.61 (48.77)
[Na] ($mg \bullet L^{-1}$)	24.38 (6.86)	54.15 (40.20)			1.74 (5.06)	0.99 (1.84)
[NH3] ($\mu g \bullet L^{-1}$)					25.69 (16.48)	
[Ni] ($\mu g \bullet L^{-1}$)	1.50 (1.42)	0.8 (1.2)[b]			44.98 (138.65)	9.25 (52.33)
[NO2] ($\mu g \bullet L^{-1}$)					1.11 (0.49)	
[NO3] ($\mu g \bullet L^{-1}$)	0.19 (0.21)				64.27 (56.93)	
[Pb] ($\mu g \bullet L^{-1}$)	1.02 (0.14)				9.08 (19.08)	
[Si] ($mg \bullet L^{-1}$)					0.91 (0.54)	
[SO4] ($mg \bullet L^{-1}$)	76.38 (41.25)	98.87 (66.15)			13.36 (5.27)	7.48 (4.45)
[Total N] ($\mu g \bullet L^{-1}$)				150 (70)[b]	141.04 (63.99)	262.69 (115.55)
[Total P] ($\mu g \bullet L^{-1}$)				18.09 (9.78)	4.67 (2.73)	8.12 (5.11)
[Zn] ($\mu g \bullet L^{-1}$)	6.50 (2.76)	4.1 (3.8)[b]			14.69 (17.63)	5.03 (6.89)
Acidity (mg $CaCO_3 \bullet L^{-1}$)			5.05 (3.17)			
Alkalinity ($\mu eq \bullet L$)	107.46 (164.86)	130.46 (221.13)	1.70 (2.29)	5.74 (13.88)	1.24 (1.13)	7.24 (11.81)
Colour (Hazen units)	36.94 (25.72)	43.91 (36.88)	35.56 (32.20)	34.08 (38.59)		21.57 (18.01)
Conductivity ($uS \bullet cm^{-1}$)	22.31 (20.31)	28.32 (23.70)	35.19 (13.38)		48.61 (44.65)	38.42 (29.31)
Depth--average (m)	4.94 (3.70)			3.60 (3.52)		
Depth--max (m)		10.12 (6.43)	12.34 (9.30)		30.27 (14.98)	
Depth--midlake (m)	20.93 (18.98)					
Hardness (mg $CaCO_3 \bullet L^{-1}$)			5.87 (2.79)			
Lake area (ha)	208.43 (134.21)	411.78 (1086.19)	111.19 (169.83)	1286.52 (5441.22)	316.07 (351.79)	790.87 (4121.47)
Lake elevation (m A.S.L.)	362.09 (134.49)	317.92 (121.39)				
Lake length (km)	3.40 (1.30)					

	Zooplankton Datasets					
Attribute	UofM	NIS	DFO	DEL	UofT	OMEE
Lake order	1.50 (0.72)					
Lake volume (10^4 m^3)	3741 (4989)					
Lake width (km)	1.18 (0.51)					
pH	6.50 (0.73)	6.22 (0.68)	5.82 (0.72)	5.48 (1.05)	5.06 (0.59)	6.26 (0.87)
Secchi depth (m)	3.72 (1.56)	2.93 (1.71)	3.27 (1.80)		9.62 (4.40)	5.27 (3.08)
Temperature (°C)			20.50 (3.16)			
Watershed area (ha)	3855 (5696)	2010 (5087)				5874 (31636)

(a - Macroinvertebrate and Waterbird data are summarized under the CWS contribution for fish; b - Values may be suspect due to lack of documentation; c - WTRAB = area of watershed above (draining into) the lake that was sampled.)

Appendix 2: Equations used for DOC (mg $C \bullet L^{-1}$) conversions between lake colour (Hazen units) and secchi depth (m). The plots indicate the goodness of fit of the linear regression relationships (error estimates for the slope and constant are shown in parentheses in each equation), as well, the regression coefficients (R^2) and p values are noted for each equation.





Appendix 3: Quadratic equations developed for proportional response of species richness to pH changes in Section 3.3. The equations for each province where sufficient fish and zooplankton data were collected are shown. (See Figure 3 for plots of individual dataset curves.)

TAXON	PROVINCE	Quadratic Equation
Zooplankton	Ontario	$-0.225 * (pH)^2 + 2.88 * (pH) - 6.84$
	Québec	$-0.355 * (pH)^2 + 4.56 * (pH) - 13.82$
	Newfoundland	$-0.292 * (pH)^2 + 3.69 * (pH) - 10.83$
	Nova Scotia	$-0.394 * (pH)^2 + 4.47 * (pH) - 11.99$
Fish	Ontario	$-0.337 * (pH)^2 + 4.12 * (pH) - 10.56$
	Québec	$-0.507 * (pH)^2 + 5.89 * (pH) - 16.46$
	Nova Scotia	$-0.523 * (pH)^2 + 5.87 * (pH) - 16.14$