SPATIAL ANALYSIS OF POPULATION TRENDS OF AMERICAN MINK

(*Neovison vison*) IN ONTARIO

By

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Abstract

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The trend in number of American mink (Neovison vison) harvested each year in the province of Ontario suggests their population may be in decline. A number of environmental variables have the potential to negatively affect Ontario’s mink, including: polychlorinated biphenyls, mercury, habitat loss, regional declines in muskrat populations, and hybridization or competition with feral mink. The likelihood mink harvest records reflect actual population trends was examined. Since the putative causes of declines are all expected to be spatial processes, declines in mink harvest due to these causes should be spatially structured. I used this spatial structure to test the hypothesis that if any of the environmental variables are directly or indirectly affecting mink population health, then a spatial correlation between mink population trend and the identified environmental variable should be present. From my analysis, I found muskrat population trend, had a significant, positive correlation with trends in the mink population. Both current muskrat harvest and current fur farm density had significant, positive correlations with current mink harvest.

Keywords: American mink, Neovison vison, polychlorinated biphenyls, mercury, habitat loss, muskrat, Ondatra zibethicus, feral, spatial analysis.
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Introduction

The American mink (*Neovison vison*) is a widely distributed, yet rarely seen, mammal. Mink share general characteristics of all mustelids (elongated, streamlined body, short limbs, and an unpronounced, small head) (Dunstone 1993), but they are particularly well adapted to semi-aquatic environments where they are often found (Novak 1987). Their distribution ranges throughout Canada and the United States – with the exception of the extreme north and arid southwest (Novak 1987). Their feeding habits are opportunistic and highly varied, with sources of food ranging from muskrat (*Ondatra zibethicus*) and other small mammals to insects. Heavy harvests of wild populations can be compensated by their potentially high reproduction rates, with litter sizes ranging from one to eight kits (Novak 1987). Despite what would appear to be a very resilient and adaptable mammal, Ontario’s mink populations may be on the decline.

Ontario’s historic fur harvest records show a clear decline in mink harvest levels (Figure 1). Following high harvests in the 1940s, harvest numbers have gradually declined to recent yearly harvests well below 20 000 individuals. Whether the observed declines reflect actual changes in the population has been argued from both sides. To date, studies have reached varying conclusions; some state that declines are most likely occurring while others argue that declines in harvest numbers may not reflect the actual population status. Wren (1991) suggested that the observed declines in harvest may be due to a decrease in incidental captures resulting from the improvement in muskrat trapping methods and equipment. Changes in economic factors, such as pelt price, have also been suggested as potential causes for the observed decline in harvest numbers (Addison et al. 1991). However, although it is likely that short-term changes in harvest
records do not provide a clear indication of changes in mink population, longer-term increases or declines in harvest records should reflect long-term trends in mink abundance (Novak 1987). A critical issue in analyzing harvest records concerns the method of data analysis. Similar data sets in other areas of biology have successfully been used in analysis of population trends, which will be discussed later on. What can be concluded from this debate is that given the contrasting opinions on the usefulness of harvest data it is clear that careful consideration and thorough justification of methods is necessary when choosing to use these harvest data as a source of population trend information.

Whereas a number of potential causes for mink declines have been identified, the impacts of polychlorinated biphenyls (PCBs) have been more thoroughly documented than most. Being a top mammalian carnivore, mink are particularly prone to accumulation of relatively high concentrations of persistent contaminants (Somers et al. 1987). Wren (1991) suggests that PCB contaminations in Ontario likely began shortly after WWII. Sources of contamination are all due to imported PCBs, as Canada has never produced PCBs; between 1929 and 1979 Canada imported 40,000 tons of this material. The sensitivity of mink to PCBs was initially discovered in the mid 1960s when ranch mink were fed Great Lakes fish, resulting in reproductive impairment through reduced litter size and kit survival (Aulerich and Ringer 1977). Further testing found adverse effects on kit survival from parental female mink fed diets containing 0.5 mg/kg, wet weight, or more, of total PCBs (Restum et al. 1998). Mink are one of the most sensitive mammals to these compounds (Aulerich and Ringer 1977). However, while many laboratory toxicity studies have shown mink to be particularly sensitive, difficulty
in field data collection has limited linking wild mink populations with the same health risks (Haffner et al. 1998).

Along with PCBs, a second toxin that has received intensive study is mercury. Like PCBs, mercury accumulates through the food chain (Kucera 1983). It is a well-known and widespread environmental contaminant, particularly in aquatic environments (O’Connor and Nielsen 1980). Fresh and salt waterbodies accumulate the greatest proportion of mercury from anthropogenic sources. Methylmercury, alkylated by microorganisms present in sediments, typically absorbs into the sediment debris and biotic surfaces (O’Connor and Nielsen 1980). This is the most potent alkylmercury compound due to its stability, ionic behaviour, ability to be absorbed through biotic membranes, and physiological effect as a neurotoxin and teratogen (an agent that causes malformation of the embryo). When introduced into an organism, the physical properties of methylmercury result in high intestinal absorption and slow excretion. This causes an accumulation of the toxin, which continues to increase in concentration as it moves up the food chain, resulting in bioaccumulation in mink. In addition to available mercury concentrations, the increased presence of methylmercury in an environment can be influenced by environmental factors like water pH and alkalinity (Wren et al. 1987). The physiological effects seen in mink include declines in general health through anorexia and nervous system damage, as well as declines in reproductive output (Evans et al. 2000; O’Connor and Nielsen 1980). Wren et al. (1987) found mercury levels in mink to closely follow mercury levels in typically preyed upon food items. A published report of mercury poisoning in a wild mink found mercury in local fish species to be the cause (Wobeser and Swift 1976). Mercury has been suggested as a potential cause for declines
in certain populations of wild mink elsewhere (e.g. Osowski et al. 1995), but this connection has not been established in Ontario.

Habitat loss is another identified potential cause of mink population decline and has been researched using wild populations; however, few studies have been conducted. The loss of vegetation can affect the distribution of mink; particularly mature willow, saplings and shrubs, as well as woody debris (Dunstone 1993). The vegetation of exceptional concern is that found in riparian habitat, as the majority of mink activity in North America and Europe occurs within 100 – 200 m of water’s edge (Dunstone 1993). Racey and Euler (1983) found a significant negative correlation between mink density and a measure of shoreline development on seven lakes in southcentral Ontario. The probable causes of decline were cited as loss of available food, hunting locations, and suitable denning sites. As human encroachment in the remaining undisturbed shorelines throughout the province continues, the loss of vegetative cover and debris preferred by mink will inevitably continue.

The potential for muskrat to influence long-term declines in mink populations has not been considered in previous research, despite the suggestion that available food is the main factor in determining the presence of mink (Novak 1987). Muskrat are a primary food source used by mink in areas of range overlap, and mink exhibit a preference for them even in the presence of an abundance of alternative food sources (Errington 1954). Similar to the popularized lynx-hare relationship, research using Hudson’s Bay Company harvest records for mink and muskrat found significant patterns of population synchrony between the two species (Viljugrein et al. 2001). Mink predation upon muskrat is particularly high in years when muskrat populations are high, or during bouts of drought,
which leave the muskrat more exposed (Errington 1954). By late summer the majority of
the muskrat population surplus has been taken and mink typically move on to other food
sources. However, preying upon muskrat tends to resume from mid-winter to early
spring as muskrat regularly leave their dens in search of food (Errington 1954;
McDonnell and Gilbert 1980). Because of their connection with muskrat, mink may
serve as indicators of the health of local muskrat populations (Errington 1954). The
greatest selection pressure affecting muskrat populations is not mink predation; it is the
presence of water, either through excessively high water levels or periods of drought
(Errington 1954). Despite the fact that there are plenty of locations devoid of muskrat
that support mink populations (Errington 1954; Errington 1961), many believe that there
is a relationship (be it a habitat quality connection, or a prey connection) (Erb et al. 2001;

Another factor that has received little attention is the effect of feral, farmed mink
on wild populations in North America, despite being a conspicuous cause of concern in
countries where mink are not native. Instances of escape and subsequent interaction of
farmed species with wild populations have been well documented in a limited number of
species. For example, farmed Atlantic salmon (*Salmo salar*) are known to escape and
the consequences of interbreeding, competing, or transmitting disease with wild
individuals have been identified as a major concern (Naylor et al. 2005). Outside of its
native-range, the feral mink is a frequently studied species, and often cited as a major
cause of concern to native species through competition with native mustelids and
predation of other species (Bonesi and Palazon 2007; Fournier-Chambrillon et al. 2004;
However, the effects of feral mink in all of the aforementioned studies pertain either to the impact on other native species, or introduction of Aleutian mink disease virus (ADV) to native species. The American mink (which is the species used in mink ranching) is native to North America, and because of this the issue of feral mink has been of little interest. Within North America, the issues are the same as seen with Atlantic salmon, resulting in a negative impact on wild American mink, either through increased competition, transmission of disease or hybridization (Bowman et al. in press). Thus, while it is well known that feral mink are having a detrimental impact on ecosystems outside North America, the potential impacts of feral individuals on wild members of their own species has only recently been identified.

The value in assessing the status of mink in Ontario can be seen from both economic as well as environmental perspectives. From an economic perspective, fur harvesting remains a viable activity, despite a large portion of mink pelts coming from fur farms, and any declines in wild populations can affect the fur trade industry. From an environmental perspective, in addition to their inherent biodiversity value, mink may be a valuable, underutilized resource for assessing environmental health. Because of their relatively small home-range, and their year around occupancy, mink can potentially provide an accurate reflection of their surrounding environment, particularly where there are concerns of bio-toxicity (Addison et al. 1991; Evans et al. 2000; Kucera 1983; Loukmas and Halbrook 2001; Martin et al. 2006). However, for mink to serve as an indicator I must first determine the current status of mink populations within the province, and assess the spatial and temporal relationships of mink to potential threats.
The purpose of the current study is to evaluate whether American mink populations are experiencing an actual decline, and if so, assessing what may be the cause of the decline. Determining what variables are involved in the decline in Ontario’s mink population is still inconclusive. Here the hypothesis of declines occurring in American mink is assessed in two stages. First, I assess the suitability of harvest data as a measure of mink population trends. Assuming this is true, and mink populations are experiencing significant declines, then regions of decline above and below average provincial trends should be identifiable through spatial analysis of the temporal trends in mink harvest records. I then test the hypothesis that if one or more of the environmental variables are directly or indirectly affecting mink, then a spatial correlation between the temporal trends in mink populations and the identified environmental variable should be present. My goal is to determine the regions experiencing the greatest rates of change in mink numbers and the factors most strongly associated with this change.

Methods

Data

Mink and Muskrat Harvest Records, PCB and Mercury Concentrations, Shoreline Development, and Fur Farm Density

Harvest data were provided by the Ontario Ministry of Natural Resources (OMNR). The data consisted of trapping records from individually issued licences (n = 109 123) for the trapping of 22 different furbearer species. Of these 22 species, mink and muskrat data were used in the analysis. Trapping records analyzed spanned 1972-1973 to 2004-2005, with the exception of records missing for 1975-1976, 1986-1987, and 1989-
1990. Because the situation exists where multiple trappers can work the same trapline, trapping records from individual licences were totalled for each trapline. I could not determine what proportion of a trapline a trapper used; therefore, the area of each trapline was calculated and the number of fur-bearers captured was transformed into a density measure of the number of fur-bearers captured / km².

Due to the existing doubt regarding the usefulness of harvest data for assessing population trends, it was critical that I thoroughly addressed the strengths and shortcomings of the data. The primary concern in using harvest data is the potential variables that may cloud or bias any underlying population trend. The key to determining if harvest data provides a meaningful surrogate for density is based on spatial and temporal patterns (Link and Sauer 1998). Bias in population estimation can occur when spatial or temporal patterns exist which could be inaccurately interpreted as changes in population size. In the case of harvest data, certain covariables likely influence mink harvest trends at the provincial scale. For example, pelt price is a variable that is uniform throughout the province. Thus, with the exception of some variability, the impact of pelt price is one that would be observed province-wide (see Appendix for further assessment of the spatial correlations between harvest levels and pelt price). As with pelt price, any slight differences among traplines due to covariables such as effort or change in trapping equipment would represent measurement error, which can be overcome by the substantial number of traplines included in the analysis. As well, rigorous trapline selection criteria can limit the effects of these covariables by selecting traplines that have continued to be actively trapped, irrespective of price, trapping popularity, or demand. From the actively trapped lines, I was interested in regions where harvest numbers significantly differed
from overall provincial trends. The spatial structure of these regions was verified using a correlogram (Figure 2). Once these areas were identified, I could then go about determining if any of my identified environmental factors share the same spatial distribution.

Traplines were identified by taking trapline information found nested within each trapper license number. For northern Ontario, the trapping license was processed by taking the two-character district code and combining it with the three-digit trapline number. For southern Ontario, the trapping license was processed by taking the four-digit geographic township found within the license code.

How the data is filtered and modelled can greatly influence the resulting trends (Villard and Maurer 1996). Many of the data quality and analysis challenges encountered in using harvest data are similar to problems seen in data collected for the Breeding Bird Survey (BBS) (Sauer et al. 1997). The BBS is a roadside bird survey program conducted by volunteer birders. Similar to a trapline, each survey volunteer is assigned a route that is approximately 39 km long, with fifty stops at equal intervals along the route. At each stop a 3-minute point-count is conducted, which consists of recording all birds heard or seen. The BBS program was launched in 1966, consisting of 600 survey routes in the U.S. and Canada. Currently, there are approximately 3700 actively surveyed routes. It is believed the lessons learned through research using the BBS data can be applied to similar data sets (Link and Sauer 1998; Thomas 1996). Thus, I applied insights found through research using the BBS data to the furbearer harvest data. James et al. (1996) reported that BBS survey data were ‘noisy’, and compensated for this noise by using only the most complete surveyed sites (sites that had been sampled at least once every 5 years
over the 25 year time-series). I applied this 5-year limitation to the furbearer harvest data. Within the furbearer harvest data certain reporting practices were not consistent throughout all the records, specifically inconsistent reporting of 0 values: in some records 0 indicated no pelts were harvested, whereas in other records a 0 was used to represent blanks, or no data, left in the harvest reports. Due to this inconsistency, records containing a 0 were excluded from the analysis. Another inconsistency in the data occurred where records from 1997-1998 to 2002-2003 contained harvest numbers collected from fur auction houses. The auction house records were added in cases where a trapline contained no record of reported harvests submitted by the trapper. In an attempt to limit inconsistencies between actual year of harvest and year received by the auction house, these records were also excluded. Finally, some trapline boundaries changed during the 32-year time-period I considered. Altered traplines, identified by comparing the most recent Ontario trapline base map, provided by the OMNR, with a recent digital copy, were also excluded from the analysis.

PCBs and mercury data, collected from fish samples, were provided by the Ontario Ministry of the Environment and consisted of all collected data for the yearly Guide to Eating Ontario Sport Fish (Ontario Ministry of Environment 2005). The data consists of sport fish samples collected since 1970 throughout Ontario. Collected samples were sent to the Ministry of the Environment laboratory in Toronto for contaminant analysis. A minimum sample size of 10 fish, with a representative range of lengths and weights, was preferred in the locations being tested (Ontario Ministry of Environment 2005). Numerous fish species were sampled through the program. Due to differing food preferences and habitats, each fish species may accumulate various
contaminants and toxins at different rates (Ontario Ministry of Environment 2005), which was taken into account by standardizing the contaminant levels for each species of fish. I assumed the sampled fish provide a representative sample for the relative concentration of mercury and PCBs in the lake they were sampled from; thus, determining whether every fish species was common prey for mink was unnecessary. Different contaminants and toxins were tested in each sample depending upon the geographic location (Ontario Ministry of Environment 2005). In remote areas where pollution sources are limited, mercury is the only contaminant of concern, with sources from either long-range transport or natural leeching. In more populated areas there can be many pollution sources, resulting in potentially high concentrations as well as higher variability.

Because of these factors, areas in close proximity to contaminant sources were retested every one to three years, while areas further from contaminant sources were tested at least every 5 years. The boneless, skinless, dorsal fillets provided the most accurate contaminant concentration measures, and were exclusively used for calculating PCB and mercury concentrations in this study.

The PCBs measured were a sum of all PCB congeners present in the sample. All PCB samples were summarized by species, and geographic location. The ten fish species sampled from the largest number of lakes were selected. In descending order by number of lakes sampled, these were: Walleye (*Stizostedion vitreum*) (n = 287); Smallmouth Bass (*Micropterus dolomieu*) (n = 277); Northern Pike (*Esox lucius*) (n = 252); Lake Trout (*Salvelinus namaycush*) (n = 249); White Sucker (*Catostomus commersoni*) (n = 184); Yellow Perch (*Perca flavescens*) (n = 179); Carp (*Cyprinus carpio*) (n = 167); Brown Bullhead (*Ameiurus nebulosus*) (n = 122); Whitefish (*Coregonus clupeaformis*) (n =
116); and Largemouth Bass (*Micropterus salmoides*) (n = 115). Because of the variability in contaminant levels between species, the data from each of the ten species selected were standardized using z-scores to allow the combining of data from the different species. To remove potential temporal trends in the PCB data, residuals for all the standardized data were taken from a calculated yearly provincial mean. Samples within 2 km of any trapline were included in calculating PCB levels for that trapline (an issue arising primarily in traplines along the Great Lakes). Lakes with less than 30 samples from the standardized values of the 10 species combined were discarded. For each lake that remained, a mean residual value was calculated from the standardized values of all the combined data.

Mercury samples were also summarized by species, and geographic location. As with the PCB data, the ten fish species with mercury samples from the largest number of lakes were selected. In descending order by number of lakes sampled, these were: Northern Pike (*Esox lucius*) (n = 1037); Walleye (*Stizostedion vitreum*) (n = 1002); Smallmouth Bass (*Micropterus dolomieui*) (n = 546); White Sucker (*Catostomus commersoni*) (n = 443); Lake Trout (*Salvelinus namaycush*) (n = 424); Yellow Perch (*Perca flavescens*) (n = 421); Whitefish (*Coregonus clupeaformis*) (n = 271); Largemouth Bass (*Micropterus salmoides*) (n = 263); Rock Bass (*Ambloplites rupestris*) (n = 225); and Brown Bullhead (*Ameiurus nebulosus*) (n = 215). The same steps of standardization, detrending, and summation applied to the PCB data were applied to the mercury data.

Shoreline development was my measure of habitat loss, and was estimated based upon proximity to roads. Recent digital data for roads and waterways were used (DMTI
Road network data was current to May 2005, and data accuracy was equal to that found in the National Topographic Data Base (NTDB): within 10 m in urban areas; within 25 m in rural areas; and within 125 m in isolated areas. Provincial coverage of primary road classes (expressway, principal highway, secondary highway, major road, and local road) were used in the analysis. Waterway data was current to August of 2005, and data accuracy was equal to that found in the NTDB roads data. Provincial coverage of all major and minor waterbodies and rivers were included in the analysis. To assess shoreline development, any waterbodies or rivers displayed as polygons in ArcGIS 9.1 (Environmental Systems Research Institute 2004) were transformed to outlines, and their perimeters were calculated. These outlines were then added to the existing linear river data. Using the combined lake perimeter and river length data, the length of shoreline found within 100 m of the provincial road network was calculated. The distance of 100 m is an approximate maximum distance from shoreline most commonly used by mink (Dunstone 1993). The data was then converted to a percent of total shoreline classified as ‘developed’ within each trapline.

Statistics Canada fur farm data was used to assess the potential impacts of feral mink on wild mink populations. The number of fur farms was surveyed in 1986, 1991, 1996, and 2001 by census district (Statistics Canada 1986, 1991, 1996, 2001). A mean measure of the number of fur farms over the entire time-series was calculated from these data and then converted to a density measure (number of fur farms / km²) within each census district. Each trapline was then assigned its overlaying fur farm density measure. In cases where a trapline boundary overlapped more than one fur farm density measure a proportional mean fur farm density was calculated.
Statistical Analysis

Furbearer Summary Statistics - Changes in Harvest Number Over Time

I used a nonlinear regression method to model changes in mink and muskrat harvests over time. Modelling trends in furbearer population densities can be challenging, as well as misleading, if an unsuitable method of analysis is selected. I used tested methods from the BBS to model trends in the furbearer harvest data. Following an approach used by James et al. (1996), local regression models (LOESS) (Cleveland et al. 1988) were applied to the harvest numbers for both mink and muskrat. This was deemed the most suitable method because it did not require selecting a model a priori and involved a minimal number of assumptions (James et al. 1996).

The fundamental principle of LOESS regression is that a model is fitted to the sample data using a local subset, or neighbourhood, of the data points (Cleveland et al. 1988; James et al. 1996). Within the neighbourhood of points, a distance-weighted function is applied from the centre of the points being analyzed. How smooth the fitted line becomes is influenced by the size of the neighbourhood analyzed, representing a fraction of the total data available (the f-value). The value of this fraction can range between 0 and 1. The use of LOESS regression has been criticized because the f-value can be arbitrarily selected; with small f-values providing close fitting models but with little explanatory value, and large f-values providing a simple model that may not fit the data effectively (Link and Sauer 1997). The challenge is to find a balance between models that closely fit the data and those simple enough to provide insight into the trend in the data. However, the issue of an arbitrary f-value can be avoided by selecting a value using a biased-corrected Akaike’s Information Criterion (AICc). The AICc
selection method works by selecting an f-value that possesses the optimal balance between a model that fits the data and one that is simple in its interpretation. LOESS regressions for each trapline were completed using PROC LOESS (SAS Institute 1997) (See Appendix for more detail on the LOESS method).

LOESS parameter estimates for the harvest data were used to calculate a measure of change over time. James et al. (1996) noted that parameter estimates were susceptible to greater variation at the ends of the time series, likely due to limitations in the neighbourhood analysis function. Following their approach to reduce variability, 3-year averages were calculated 4 years in from the beginning and end of each trapline regression. A value of change in furbearer harvest over the time-period was then calculated by the equation \( N_t/N_{t-1} \), equalling \( \lambda_{\text{furbearer}} \). In order to normalize \( \lambda_{\text{furbearer}} \) the natural log was taken, and the transformed values were used in all subsequent analyses.

**Spatial Analysis**

I wanted to examine the correlation between my dependent variable, \( \lambda_{\text{mink}} \), and each independent variable. However, because of the continuous geographic distribution of my samples, they likely possessed some degree of spatial autocorrelation. To overcome the issue of spatial autocorrelation I used Mantel’s tests and partial Mantel’s tests to assess the spatial relationship between \( \lambda_{\text{mink}} \) and each of the environmental variables. Euclidean distance matrices were calculated for \( \lambda_{\text{mink}} \) and all the environmental variables. For the partial Mantel’s test, geographic distance was the third matrix included in the analysis. Due to data requirements for the Mantel’s test, only sites where overlap occurred between the dependent and independent variables were used. As
a result, there was a different sample size for the analysis using each independent environmental variable. Comparison of the results from the Mantel’s test and partial Mantel’s was used to test the influence of geographic distance (Fortin and Dale 2005). Permutation tests of 9999 iterations were performed to test for significance (See Appendix for more detail on Mantel’s test).

A second set of spatial tests was done on the current distribution of mink populations. The purpose of this analysis was to assess whether the current distribution of mink harvests and environmental variables reflected the spatial pattern seen using the temporal trends. Because of limitations in the time-span covered by each variable, correlations can be weakened (e.g. Flather and Sauer 1996). As well, identification of regions presently experiencing above or below average harvests and associated environmental variables is important in providing direction for future research. Average number of mink harvested / km² was calculated for each trapline for the years 2000-2004. The same calculation was applied to muskrat harvests. Fur farm statistics from 2001 were used, rather than the previously calculated average from 1986-2001. The measure of development was already a current measure, containing all currently mapped roads. The remaining measures, PCBs and mercury, had already been made timeless by detrending the data, and were left unchanged. Mantel’s test and partial Mantel’s tests were used to measure spatial correlations between current mink harvest levels and current environmental variables.
Model Selection

I used an information theoretic approach to compare all possible environmental models to examine the relationship between $\lambda_{\text{mink}}$ and their surrounding environment (Burnham and Anderson 2002). Multivariate models were tested using Akaike’s Information Criterion for small sample sizes ($\text{AIC}_c$) to test which suite of variables best explained $\lambda_{\text{mink}}$. The function of the $\text{AIC}_c$ approach to model selection is to find the most parsimonious model by minimizing the amount of information lost from the suite of available models (Burnham and Anderson 2002).

I could find no justifiable reason to exclude any of the possible combinations of environmental models. The various combinations of environmental variables presented 31 potentially suitable models. Spatial autoregression using lagged-predictor models were used to regress each model while accounting for spatial autocorrelation in the data (Rangel et al. 2006). Correlograms of the residuals for each model were examined and no residual spatial structure was found. The difference between the lowest scoring model and each subsequent model ($\Delta_i$) was calculated, along with the Akaike weights ($w_i$). The $\Delta_i$ was used to rank the models from best to worst and $w_i$ was used to assess the probability that model $i$ was the best approximating model, within the set of candidate models (Anderson et al. 2000). The relative importance of each parameter was then calculated. I established a 95% confidence set of models for the best Kullback-Leibler model ($\text{K-L}$) by summing descending values of $w_i$ to a value greater than or equal to 0.95 (Burnham and Anderson 2002). Using this confidence set, I calculated the model-averaged parameter estimates, and their respective 95% confidence limits. For the $\text{AIC}_c$
analysis, locations where all the independent variables were present reduced the sample size to 132 (See Appendix for more detail on the $AIC_c$ analysis method).

**Results**

**Spatial Patterns**

Shoreline development, calculated from percent of shoreline within 100 m of roads, clearly showed a gradient of increasing development from north to south (Figure 3). Particularly high levels of development were seen in the southwest portion of the province around Essex county, the southwestern shores of Lake Ontario in the Halton and Peel counties, and the northern shores of Lake Erie in Niagara county (Figure 4).

Mink farm densities were found to be high in Perth and surrounding counties, and were evident in the mean density of fur farms as well as the most recent 2001 records (Figures 4, 5 & 6). Moderately high densities of farms were also found in Essex, Lanark and Niagara counties (Figure 4). Change between mean number of farms and those seen in the most recent, 2001 records, indicate a general decrease in the number of active farms, but many can still be found in clusters, most notably in Perth and surrounding counties (Figure 4).

With the exception of a few hotspots, only a small number of locations were identified as possessing higher mercury levels than average values seen throughout the province (Figures 4 & 7). Prominently elevated levels were observed in Haliburton, Halton & Peel, and Kenora counties (Figure 4).

Not surprisingly, PCBs were found to be elevated along the shores of the Great Lakes and St. Lawrence River and rapidly declined with inland progression (Figures 4 &
8). The same region of Halton county found to have higher than average mercury levels also showed above average PCB levels, particularly for an inland sampled location (Figure 4).

Areas of spatially continuous decline in muskrat harvest were most notable in the counties of Simcoe, Muskoka, and Haliburton (Figures 4 & 9). Examination of the start (1976-1979) and end (1998-2000) of the data from LOESS regression yearly parameter estimates suggests that muskrat harvest numbers have changed from a relatively even distribution in harvest numbers across a large portion of southern Ontario to the majority of muskrat coming from a few core areas (Figures 10 & 11). Muskrat harvest densities calculated using harvest records from 2001 to 2004 show spatial patterns similar to $\lambda_{\text{muskrat}}$ from the end of the LOESS regression, with the exception of patches of high density appearing less dispersed (Figure 12).

Similar to the observed changes in muskrat populations, areas of spatially continuous decline in mink populations were most notable in the counties of Muskoka and Haliburton (Figures 4 & 13). Examining maps of $\lambda_{\text{mink}}$ calculated from the LOESS regression yearly parameter estimates from the start (1976-1979) and end (1998-2000) most effectively demonstrates this change (Figures 14 & 15). There appears to also be a possible contraction in the number of mink harvested in the Grey and Wellington counties (Figure 4). However, little change has occurred in central Ontario, and modest declines can be seen in northern Ontario. The 2001 to 2004 records used to estimate the current population status of mink appears quite similar to that produced from the values found at the end of the LOESS regression (Figure 16).
**Spatial Analysis**

The Mantel’s test clearly showed $\lambda_{\text{muskrat}}$ to be the most highly correlated variable with $\lambda_{\text{mink}}$ (Table 1). The relationship between $\lambda_{\text{muskrat}}$ and $\lambda_{\text{mink}}$ may be considered weak; however, because the Mantel’s test is calculated using distance matrices, rather than raw data, it is known to result in lower measures of correlation (Fortin and Dale 2005). Analysis of the mink and muskrat relationship using a partial Mantel’s test only slightly increased the correlation. Attesting to the strength of the correlation between $\lambda_{\text{mink}}$ and $\lambda_{\text{muskrat}}$, no other environmental variables possessed any significant spatial relationship with $\lambda_{\text{mink}}$.

The strength and frequency of significant correlations changed when looking at the average mink harvest / km$^2$ from 2000 to 2004 (Table 2). While muskrat harvests remained the independent variable with the greatest degree of correlation, 2001 fur farm density was also found to be significantly correlated. Use of a partial Mantel’s test slightly increased the correlation between average mink harvest / km$^2$ and all of the independent variables, with the exception of percent shoreline development.

**Model Selection**

The most parsimonious model that explained $\lambda_{\text{mink}}$ was $\lambda_{\text{muskrat}}$ (Table 3). This selected model explained 25.5% of the variation in $\lambda_{\text{mink}}$. The evidence ratio ($w_i/w_j$) showed the $\lambda_{\text{muskrat}}$ model to be 4.7 times more likely to be the most suitable model for explaining $\lambda_{\text{mink}}$ over the second highest ranking model of $\lambda_{\text{muskrat}}$ and PCBs, and 6.3 times more likely than the third highest ranking model of $\lambda_{\text{muskrat}}$, and fur farms.
Parameter importance and estimates showed $\lambda_{\text{musk rat}}$ as, by far, the most significant variable in modelling $\lambda_{\text{mink}}$ (Table 4). Parameter importance, calculated by summing the Akaike weights from each of the candidate models, indicated that $\lambda_{\text{musk rat}}$ was 5.7 times more influential when compared to PCB concentrations.

The distinction between $\lambda_{\text{musk rat}}$ and all other environmental variables in its ability to describe $\lambda_{\text{mink}}$ was further demonstrated with the parameter estimates (Table 5). Calculated through model averaging of the confidence set of models, the estimates showed $\lambda_{\text{musk rat}}$ to be the only parameter with significant trends, with all other variables possessing 95% CL encompassing zero.

**Discussion**

The spatial structure of changes in mink harvest suggest that declines observed in total mink harvests for the province of Ontario are not solely due to the effects of trapper effort, fur price, and related variables; and that declines in mink populations are occurring. Whereas it is likely that the decline in the popularity of trapping and demand for fur are contributing to the downward trend in provincial harvest levels, these trends alone do not explain the spatial structure observed. This conclusion is based on the assumption of a spatially homogeneous decline in fur price and therefore trapper effort. Mapping of trends in mink harvest levels show certain regions of Ontario have experienced greater rates of mink harvest decline relative to the provincial average, indicating that other factors are contributing to changes in mink population levels. In particular, Muskoka and Haliburton counties appeared to be experiencing the greatest rates of decline. As well, areas of northern Ontario in the Kenora district also showed
rates of decline above the provincial average; however, data available for the region are limited.

From the suite of tested environmental parameters, $\lambda_{\text{muskrat}}$ was able to explain the greatest amount of variability in $\lambda_{\text{mink}}$ over the time-period of 1972 to 2004. Errington (1954), McDonnell and Gilbert (1980), and Soper and Payne (1997) identify muskrat as a frequently used food source, particularly important during the winter months; thus, the obvious identifiable cause for the variation in $\lambda_{\text{mink}}$ was due to the loss of muskrat as a food source. Erb et al. (2001) go on to state that in certain regions mink may be seen as a specialist predator of muskrat, and that a response to loss of muskrat can be a decline in mink population numbers. The strength of the predator-prey relationship between mink and muskrat has been compared to commonly cited lynx-hare population cycle, particularly in regions were other suitable food sources may be limited (Viljugrein et al. 2001). Nevertheless, some view mink as opportunistic hunters, targeting whatever prey is available (Eagle and Whitman 1987). If this were the case then declines in muskrat alone would not necessarily result in declines in mink. Alternatively, it has been suggested that the link with muskrat is not primarily one of predation, but rather link with habitat alterations that affect both species (Erb et al. 2001; Errington 1954). As with mink, comprehensive research on muskrat population health, harvest dynamics, and potential causes of decline are sorely lacking (Erb et al. 2000; Boutin and Birkenholz 1987). Whether the connection seen between changes in mink and muskrat populations is direct, through predation, or indirect through other environmental causes remains unclear.
The weak spatial correlation between mink and PCBs does not necessarily exclude it as a potential cause for concern. Although PCB samples from sport fish have been collected from many areas throughout southern Ontario, a large proportion were collected at distances greater than the 2 km buffer used to link them to tralines in close proximity. As a result, available PCB data were limited, mostly to the shorelines of Lake Erie and Lake St. Clair. Previous research using mink collected by trappers in Ontario has found PCB concentrations collected from tissue samples surpassing thresholds of toxic response. A study conducted in 1978-1979 found mink collected from Mersea and Dunn-Rainham townships possessed PCB concentrations at levels great enough to likely cause reproductive impairment (Proulx et al. 1987). To examine the level of PCB contamination in the Great Lakes, a second study compared mink populations residing along the shores of Lake Ontario and Lake Erie to populations residing inland (Haffner et al. 1998). Like Proulx et al. (1987), they found contaminant concentrations from mink collected in Mersea township were high enough to negatively affect mink populations. PCB concentrations capable of reducing reproductive output were also found in mink residing on Cornwall Island and in Otonabee, Dorchester, Dover, Darlington, and Monaghan townships. However, from the spatial distribution of PCB concentrations collected from mink tissue samples Haffner et al. (1998) were unable to support the hypothesis of greater PCB concentrations from populations boarding the Great Lakes. They suggest that accounting for variation due to age and increasing sample size may provide significant differences between shoreline and inland samples. In a similar study conducted by Martin et al. (2006), significantly higher concentrations of PCBs were found in mink trapped along the western basin of Lake Erie where almost 40% of the
samples exceeded concentrations linked with reproductive impairment. Mink were also
found to have intermediate levels of PCBs from eastern Lake Erie, Lake St. Clair and
Walpole Island in the St. Clair River. Martin et al. (2006) did a temporal comparison
using samples collected in 1978-1979 by Proulx et al. (1987) and recent samples. They
found the level of PCBs in mink to be declining in Lake St. Clair; however, in the
western basin of Lake Erie levels of PCB contamination in mink have remained equal to
historic levels, if not increased. In all cases, samples exceeding concentrations linked to
reproductive impairment are limited to major bodies of water, with very low levels
inland. As a result, the province-wide effect of PCBs as a major factor affecting mink
population health is questionable.

The areas of greatest concern for mercury’s negative effect on mink is in the
northern regions of southern Ontario, although its effect on mink appears to be much less
than those observed with muskrat. However, regions of extremely high levels of mercury
observed from the Ontario sport fish data appear few and far between. It should be noted
that regions identified as possessing high mercury levels using the Ontario sport fish data
appear to correlate well with known areas of high mercury pollution (e.g., Dryden; Wren
1985), providing some validation for the surrogate mercury measure. In the areas where
mink harvest records and significantly high mercury levels did overlap there did not
appear to be any consistent trend in below average harvest numbers. It is possible that
the harvest records reflected the most fit mink from the population, as those less fit were
unable to survive the local environmental stresses; thus, the remaining healthy, active
individuals were trapped (Evans et al. 2000; Osowski et al. 1995; Wobeser and Swift
1976). This is further supported by the high incidence of areas with extremely elevated
mercury levels lacking mink harvest records. Previous research assessing spatial variation in mercury concentrations in mink from the James Bay area of Quebec found mercury was significantly correlated with soil type and proximity to local industrial areas; however, less than 1% of the sampled mink exceeded the lowest threshold of mercury concentrations known to cause adverse effects (Fortin et al. 2001). Mink carcasses collected from Manitoba waters accumulated mercury concentrations about one order of magnitude higher than the local fish (Kucera 1983), further demonstrating the potential for sport fish to be an indicator of areas of concern for mink. The extent of localization effects of high mercury levels was as little as 2 km resulting in significant declines in mercury levels in resident mink (Wren et al. 1987). These localized effects may be occurring at a resolution finer than what my analysis was capable of detecting. More complete data sets would be a great benefit for further assessment of the effects of mercury on mink populations.

Contrary to the negative effects of development on mink populations reported previously (Racey and Euler 1983), no significant correlation was found between my index of shoreline development and changes in mink populations. The limited research done on the effects of habitat on the density of mink found shoreline development to be the significant factor influencing the presence of mink (Burgess and Bider 1980; Racey and Euler 1983). Racey and Euler (1983) attributed the degradation of suitable mink habitat to cottage development, and the associated clearing of land and road building. Connected with these changes in habitat they suggested mink feeding behaviour and habitat selection may be modified by changes in prey populations and vegetation. Supporting Racey and Euler’s (1983) study, an examination of mink presence found
suitable denning sites and adequate cover, the presence of mature willow trees, and shrub cover had significant influence on mink distributions (Mason and Macdonald 1983). Burgess and Bider (1980) took an inverse approach to assessing the effects of shoreline development by looking at how improving degraded habitat affects mink population density. They found that improvement of habitat through increases in food availability resulted in increases in mink activity in the area. It has also been suggested that the effect of habitat destruction may be less direct, through increasing stress and consequently increasing mink’s susceptibility to other factors of concern (Osowski et al. 1995). Differing measures between previous studies and my own could account for the differences found in the strength of the relationship between shoreline development and mink population health. Racey and Euler’s (1983) study took place in central Ontario, where shoreline development is only moderately high when compared to southern Ontario, but is an area of more recent development. All of the other aforementioned development studies possessed similar limitations in the scale of analysis. Whereas localized effects of shoreline development may result in changes in density of mink populations, the relationship breaks down when the entire range of shoreline development throughout the province is considered. Research conducted at a larger scale looking at the effects of roads on wetland biodiversity found roads could have an impact on populations up to distances of 2 km (Findlay and Bourdages 2000). As well, the total extent of effects of road construction may not be seen for decades. This may be another possibility for why development was not found to be significant in my study. The loss of biodiversity and its effect on prey availability for mink is an obvious connection. It is without a doubt that in certain situations development can affect mink populations.
Although development has been found to negatively affect mink at smaller scales, my research indicates that the extent to which development has affected the shorelines of Ontario is not sufficient to account for the majority of declines seen in mink.

The connection between fur farms and mink populations is difficult to decipher as this parameter was not highly influential in the models favoured by my model selection procedures but was spatially significant when using recent harvest records. Distribution of fur farms limits the immediate effects on wild mink populations to southern Ontario, but if the feral mink can survive then dispersal is probable (Northcott et al. 1974). The positive correlation between fur farms and recent mink harvest records suggests they may be providing a population source. Considering the positive correlation, declines in the number of fur farms resulting in a reduced number of feral mink captures, could be reflected as declines in harvest levels. Whether these could account for all the observed declines in harvest levels is questionable. Furthermore, if harvests are supplemented by ranch escapes, declines in wild populations may be more severe than suggested by the harvest data. Thus, the population source from fur farms may further complicate identifying true environmental factors of concern by obscuring these relationships with inflated population numbers. Another possible negative effect of feral mink may be to provide a source of genetic introduction, competition, or transmission of disease into the wild populations (Bowman et al. in press). The population health implications include reduced brain size, altered behaviour, reduced sexual dimorphism, and increased inbreeding (Bowman et al. in press). The spatial correlation between fur farms census data from 2001 and recent mink harvest records suggest fur farms may be having an immediate impact on mink harvest levels, and potentially population health. The change
in distribution of fur farms over the past 20 years, seen in Figures 5 and 6, is obvious. The extent of the relationship between harvest records and fur farm density could be further tested by examining the spatial correlation between harvest records coinciding with 5 year census records of fur farm densities. The possibility of fur farms having a significant effect on wild mink populations has only been recently identified, and determining the extent of its effect certainly requires further research.

There are caveats concerning data quality and assumptions that merit some attention. Limitations in available data, as well as quality of the data must always be considered when dealing with any large-scale analysis. Because traplines can differ greatly in size, furbearer harvests had to be converted to a measure / km². In doing so, the assumption had to be made that trappers were utilizing all available land within their trapline. In situations where this is not the case, the calculated density measure would be biased towards underestimating population density. However, as long as the areas that were trapped within a trapline remained the same over time, then the change in population would be accurate. A second assumption that had to be made concerned reporting of 0s in the trapping records, due to inconsistent record reporting between districts. Zeros had to be interpreted as no trapping records reported, rather than no mink captured despite an effort to do so. The result of this assumption is that traplines that exhibited a complete population loss would be excluded from the analysis, and would create the overall appearance of a healthier mink population.

The surrogate measures of PCB and mercury concentrations may not accurately reflect the possible concentrations of these toxins in mink. Toxin concentrations were calculated from the 10 most frequently sampled fish species, irrespective of whether the
species was a common prey item for mink. I assumed that the sampled fish species provides a generalized representation of toxin levels in the surrounding environment. I then inferred that since similar habitat is shared between fish and mink, these toxin levels would relate to those found in mink. However, the true relationship of the toxin levels mink were actually exposed to is unknown. The resulting implications of this assumption is that a lake may be reported as having elevated PCB or mercury concentrations, but because the fish species is not a prey item of mink, the same elevated PCB or mercury concentration may not be found in mink residing around that lake. Of course, the inverse situation may occur, where the fish species may report below average PCB or mercury levels, but because the fish species is not a prey item of mink, the mink may actually have highly elevated PCB or mercury levels.

Because of the abundance of available road data, roads within 100 m of shorelines were used as a measure of shoreline development. The inherent assumption in using this data set is that the presence of roads in close proximity to shoreline directly translates to shoreline development. It is likely that in some situations a road within 100 m of a shoreline does not equate to development along the shoreline. The consequence of this situation would be an overestimation of the degree of shoreline development in the corresponding trapline. As long as the occurrence of roads in close proximity to shorelines not equating to development does not possess a spatial structure of its own, the resulting instances of inaccuracy would be a form of measurement error; and likely overcome by the number of samples taken.

The availability of fur farm data at an adequately fine resolution is quite limited. The finest resolution of data available for Ontario was at the census district level, within
which the number of fur farms was recorded; however, the exact location of each fur
farm within the census district was unknown. As a result, I had to convert the number of
fur farms within the census district into a density measure. This calculation assumed that
the impact of the fur farm was evenly distributed throughout the census district. As well,
the census district data occurred at a larger scale than the traplines, thus multiple traplines
may have contained the same estimate of fur farm density. As a consequence of these
assumptions a degree of error is added to the fur farm variable where traplines that are
actually in close proximity to the fur farm found in that census district underestimate fur
farm impact and traplines at a greater distance from the actual fur farm location
overestimate fur farm impact.

With some adjustments to data collection methods, the quality of Ontario’s
harvest data could be greatly improved. As suggested by Wren (1991), it is important
that harvest records be maintained at a fine resolution, and collected for all traplines, to
be useful for analysis. Consistent reporting practices among districts are also essential.
Trapping records detailing number of hunting days, or some other measure of effort,
would be extremely useful in providing insight into the degree to which trapper effort is
influencing changes in harvest numbers (Chanin and Jeffries 1978). Accompanying the
current information found in trapping records, other parameters (e.g. diet, fat indices, sex
ratios, tooth samples) could be obtained from carcasses (Chilelli et al. 1996; Dixon 1981;
Whitman 2003). With this information, areas identified as declining due to decreasing
harvest numbers could be cross checked for changes in harvest pressure, recruitment
rates, or may indicate more serious environmental concerns (Chilelli et al. 1996;
Whitman 2003). With the collection of a small amount of additional information, fur managers and researchers would be able to put harvest records to much more use.

Despite dealing with large data sets for all variables involved in the analysis, limitations in their spatial overlap may have decreased the strength of analysis. A large portion of traplines found in southern Ontario did not meet the necessary data quality requirements for inclusion in the analyses. The most obvious limitation concerning distribution of data related to PCBs. Most PCB contamination was found in southern Ontario, particularly along the shorelines of Lake Erie and Lake St. Clair (Martin et al. 2006). The lack of sample overlap between these areas and mink harvest data certainly restricted my ability to successfully detect a relationship.

With so many potential factors involved, unravelling the causes of mink population decline is complicated and doing so will require further research into multiple environmental issues. Maps of changes in mink harvest levels indicate that most declines have occurred in southern Ontario. In this same region, muskrat harvest numbers have declined, and the density of fur farms has decreased. Further research is needed to determine whether the relationship between mink and muskrat is one of predator-prey or an environmental factor affecting both species. It is also yet to be determined whether mink escaping from fur farms are benign and merely provide an additional fur source for trappers, or if they are potentially integrating less adapted traits into the wild population’s gene pool or out-competing wild mink for available resources (Bowman et al. in press). Furthermore, I hope that the recommendations for the improvement in harvest data collection will be considered and thus lead to harvest data being more frequently utilized in research concerning other furbearers.
Literature Cited


Evans, R., Addison, E., Villeneuve, J., MacDonald, K. & Joachim, D. 2000. Distribution of inorganic and methylmercury among tissues in mink (Neovison vison) and otter (Lutra canadensis). Environmental research 84: 133-139.


Bay Territory, Quebec, Canada. Archives of Environmental Contamination and Toxicology 40: 121-127.


Table 1. Mantel’s and partial Mantel’s tests measuring correlation between $\lambda_{\text{mink}}$, calculated using LOESS regression of harvest data spanning 1972 to 2004, and each environmental variable. Partial Mantel’s test includes geographic distance as the third variable.

<table>
<thead>
<tr>
<th>$\lambda_{\text{mink}}$ with:</th>
<th>n</th>
<th>$r$</th>
<th>$P$</th>
<th>Partial $r$</th>
<th>$P$</th>
</tr>
</thead>
<tbody>
<tr>
<td>$\lambda_{\text{muskrat}}$</td>
<td>630</td>
<td>0.1387</td>
<td>0.0124</td>
<td>0.1393</td>
<td>0.0104</td>
</tr>
<tr>
<td>Percent Shoreline Development</td>
<td>1158</td>
<td>-0.0019</td>
<td>0.4781</td>
<td>0.0000</td>
<td>0.9976</td>
</tr>
<tr>
<td>Average Fur Farm Density / km$^2$</td>
<td>1158</td>
<td>0.0030</td>
<td>0.3790</td>
<td>0.0034</td>
<td>0.8900</td>
</tr>
<tr>
<td>Mercury Concentrations</td>
<td>482</td>
<td>0.0066</td>
<td>0.3396</td>
<td>0.0069</td>
<td>0.8418</td>
</tr>
<tr>
<td>PCB Concentrations</td>
<td>155</td>
<td>-0.0437</td>
<td>0.7385</td>
<td>-0.0395</td>
<td>0.4818</td>
</tr>
</tbody>
</table>
Table 2. Mantel’s and partial Mantel’s tests measuring correlation between average mink harvest / km$^2$ spanning 2000 to 2004 to each environmental variable. Partial Mantel’s test includes geographic distance as the third variable.

<table>
<thead>
<tr>
<th>Average Mink Harvest / km$^2$ with:</th>
<th>n</th>
<th>r</th>
<th>P</th>
<th>Partial r</th>
<th>P</th>
</tr>
</thead>
<tbody>
<tr>
<td>Average Muskrat Harvest / km$^2$</td>
<td>1128</td>
<td>0.4840</td>
<td>0.0001</td>
<td>0.4863</td>
<td>0.0001</td>
</tr>
<tr>
<td>Percent Shoreline Development</td>
<td>1466</td>
<td>-0.0032</td>
<td>0.5111</td>
<td>-0.0008</td>
<td>0.9667</td>
</tr>
<tr>
<td>2001 Fur Farms / km$^2$</td>
<td>1466</td>
<td>0.0985</td>
<td>0.0020</td>
<td>0.1007</td>
<td>0.0020</td>
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<tr>
<td>Mercury Concentration</td>
<td>571</td>
<td>-0.0319</td>
<td>0.8775</td>
<td>-0.0321</td>
<td>0.2573</td>
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<tr>
<td>PCB Concentrations</td>
<td>179</td>
<td>-0.0026</td>
<td>0.3753</td>
<td>-0.0029</td>
<td>0.9578</td>
</tr>
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</table>
Table 3. Model ranking relating $\lambda_{\text{mink}}$ with all possible combinations of environmental variables. Analysis was done using Akaike information criterion adjusted for small sample size ($\text{AIC}_c$). Models identified in bold represent the confidence set of models used for calculation of parameter estimates.

<table>
<thead>
<tr>
<th>Model Rank</th>
<th>Parameters</th>
<th>n</th>
<th>K</th>
<th>$\text{AIC}_c$</th>
<th>$\Delta\text{AIC}_c$</th>
<th>$w$</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Muskrat</td>
<td>132</td>
<td>3</td>
<td>-103.304</td>
<td>0.000</td>
<td>0.55703</td>
</tr>
<tr>
<td>2</td>
<td>PCBs + Muskrat</td>
<td>132</td>
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<td>-100.225</td>
<td>3.079</td>
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<tr>
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<td>4</td>
<td>-99.613</td>
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<td>Development + Muskrat</td>
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<td>8</td>
<td>Fur Farms + Muskrat + Mercury</td>
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<tr>
<td>23</td>
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<tr>
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<tr>
<td>25</td>
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<td>-60.699</td>
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<tr>
<td>27</td>
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<td>28</td>
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<td>45.499</td>
<td>0.00000</td>
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<tr>
<td>29</td>
<td>Fur Farms + Mercury + Development</td>
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<td>-57.346</td>
<td>45.958</td>
<td>0.00000</td>
</tr>
<tr>
<td>30</td>
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<td>-56.663</td>
<td>46.641</td>
<td>0.00000</td>
</tr>
<tr>
<td>31</td>
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<td>6</td>
<td>-53.878</td>
<td>49.426</td>
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</table>
Table 4. Parameter importance for each variable in describing $\lambda_{\text{mink}}$ using all models calculated from Akaike information criterion adjusted for small sample size ($\text{AIC}_c$).

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Importance</th>
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</thead>
<tbody>
<tr>
<td>Muskrat</td>
<td>1.00000</td>
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<tr>
<td>PCBs</td>
<td>0.17440</td>
</tr>
<tr>
<td>Mercury</td>
<td>0.13671</td>
</tr>
<tr>
<td>Fur Farms</td>
<td>0.13215</td>
</tr>
<tr>
<td>Development</td>
<td>0.09650</td>
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</table>
Table 5. Parameter estimates for describing $\lambda_{\text{mink}}$ using the identified suite of 7 candidate models calculated using model averaging from results of a Akaike information criterion analysis adjusted for small sample size ($\text{AIC}_c$).

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Estimate</th>
<th>Standard error</th>
<th>Upper 95% confidence limit</th>
<th>Lower 95% confidence limit</th>
</tr>
</thead>
<tbody>
<tr>
<td>Muskrat</td>
<td>0.49980</td>
<td>0.07500</td>
<td>0.64703</td>
<td>0.35262</td>
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<td>PCBs</td>
<td>0.08801</td>
<td>0.07672</td>
<td>0.23838</td>
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</tr>
<tr>
<td>Mercury</td>
<td>-0.07602</td>
<td>0.08710</td>
<td>0.09470</td>
<td>-0.24674</td>
</tr>
<tr>
<td>Fur Farms</td>
<td>-0.09135</td>
<td>0.11312</td>
<td>0.13036</td>
<td>-0.31307</td>
</tr>
<tr>
<td>Development</td>
<td>0.01500</td>
<td>0.11100</td>
<td>0.23256</td>
<td>-0.20256</td>
</tr>
</tbody>
</table>
Figure 1. Ontario fur harvest records reported by Statistics Canada from 1919 to 2002.
Figure 2. Correlogram of $\lambda_{\text{min}}$, within each trapline calculated using Ontario fur harvest records from 1972-1973 to 2004-2005 provided by the Ministry of Natural Resources. Solid circles representing global significance.
Figure 3. Total shoreline development (%), calculated as shoreline within 100 m of roadways, within each trapline in Ontario up to the year 2005.
Figure 4. Regions of significance, identified through the spatial patterns of mink populations and environmental factors of concern.
Figure 5. Mink farms / km² within each trapline averaged from Statistics Canada census data for the years 1986, 1991, 1996, and 2001. Data were compiled for Ontario census districts.
Figure 6. Mink farms / km$^2$ within each trapline from Statistics Canada census data for the year 2001. Data were compiled for Ontario census districts.
Figure 7. Standardized mercury concentrations found in the 10 most abundantly sampled species of sport fish collected by Environment Canada for the yearly Guide to Eating Ontario Sport Fish (Ontario Ministry of the Environment 2005).
Figure 8. Standardized PCB concentrations found in the 10 most abundantly sampled species of sport fish collected by Environment Canada for the yearly Guide to Eating Ontario Sport Fish (Ontario Ministry of the Environment 2005).
Figure 9. $\lambda_{\text{muskrat}}$ estimated using LOESS regression for the harvest seasons 1972-1973 to 2004-2005 from Ontario Ministry of Natural Resources data for Ontario fur harvest records.
Figure 10. $\lambda_{muskat}$ averaged over the years 1976-1979 using parameter estimates from LOESS regression of harvest records from 1972-1973 to 2004-2005 from Ontario Ministry of Natural Resources data for Ontario fur harvest records.
Figure 11. $\lambda_{\text{musk rat}}$ averaged over the years 1998-2000 using parameter estimates from LOESS regression of harvest records from 1972-1973 to 2004-2005 from Ontario Ministry of Natural Resources data for Ontario fur harvest records.
Figure 12. Muskrat harvested / km² estimated from harvest records spanning 2000 to 2004 from Ontario Ministry of Natural Resources data for Ontario fur harvest records.
Figure 13. $\lambda_{\text{mink}}$ estimated using LOESS regression for the harvest seasons 1972-1973 to 2004-2005 from Ontario Ministry of Natural Resources data for Ontario fur harvest records.
Figure 14. $\lambda_{\text{mink}}$ averaged over the years 1976-1979 using parameter estimates from LOESS regression of harvest records from 1972-1973 to 2004-2005 from Ontario Ministry of Natural Resources data for Ontario fur harvest records.
Figure 15. $\lambda_{\text{mink}}$ averaged over the years 1998-2000 using parameter estimates from LOESS regression of harvest records from 1972-1973 to 2004-2005 from Ontario Ministry of Natural Resources data for Ontario fur harvest records.
Figure 16.
Mink harvested / km² estimated from harvest records spanning 2000 to 2004 from Ontario Ministry of Natural Resources data for Ontario fur harvest records.
Appendix

1. Assessment of the effects of pelt price on harvest effort

In order to determine the usefulness of harvest data as a measure of population trends the influence of harvest effort on the degree correlation between harvest level and pelt price was assessed. Within each trapline, correlation between yearly harvest / km² and pelt price was calculated. This statistic was calculated for muskrat harvest vs. muskrat price, mink harvest vs. mink price, as well as mink harvest vs. muskrat price. The third variable combination was included to address the possibility of mink being an incidental capture by trappers targeting muskrat, and thus influenced by muskrat pelt price.

Any trapline identified as having a significant correlation between furbearer harvest and pelt price ($\alpha = 0.05$) were further analyzed for spatial correlation using Mantel’s test. The purpose of this test was to assess whether traplines showing significant changes in harvest effort of muskrat due to pelt price were spatially correlated with traplines showing significant changes in harvest effort of mink due to pelt price. A significant result from this test would demonstrate that the harvest data possessed a spatial structure due to pelt price and would not be suitable for analysis of population trends. The results of the analysis found no significant spatial correlation between traplines with significant muskrat harvest to muskrat price correlations and significant mink harvest to mink price ($r^2 = -0.03$ $P = 0.68$). Analysis of spatial correlation between significant muskrat harvest to muskrat price and significant mink harvest to muskrat price also yielded no significant spatial correlation ($r^2 = -0.05$ $P = 0.94$). A second set of tests were completed to examine the spatial relationship between significant harvest to price
correlations and population trends ($\lambda_\text{furbearer}$) calculated through LOESS. No significant spatial correlation was found between tralines possessing significant mink harvest to mink price and $\lambda_\text{mink}$ ($r^2 = 0.04$ $P = 0.20$). As well, no significant spatial correlation was found between tralines possessing significant muskrat harvest to muskrat price and $\lambda_\text{muskrat}$ ($r^2 = -0.05$ $P = 0.82$). These results demonstrate that trends observed in the harvest data do not spatially correlate with any changes in harvest effort that may arise from changes in pelt price.

2. **Local Regression (LOESS)**

I used local regression (Cleveland et al. 1988) to model population trends found in yearly harvest records for each traline containing sufficient data (all steps used to determine tralines with sufficient data are described in the Methods section). The LOESS method of regression was adapted from its application to the Breeding Bird Survey data by James et al. (1996). LOESS is a robust nonparametric regression method, and because of its robust nature, it is well suited to exploratory analysis of datasets containing noise due to random error. I used this regression method to model yearly harvest records of each traline as a smoothed function of year through the following equation:

$$y_{ij} = g(x_j) + \varepsilon_i$$

where $y_i$ is the furbearer harvest for traline $i$ in year $x_j$, $g$ represents a smooth function of $x$, and $\varepsilon_i$ represents the error.

The degree of smoothing is defined by a smoothing parameter, $f$, and can range from 0 to 1. The smoothing parameter determines the fraction of data, or neighborhood,
of adjacent samples to be included in the weighted linear least squares regression around each data point, using the following weighting of points:

\[ w_t(x) = W \left( \frac{|x_t - x|}{d(x)} \right) \]

where \( d \) represents the distance (measured in the same units as \( x \)) from point \( x \) being regressed. A smoothing parameter was automatically calculated for each trapline using AICc analysis. All smoothing parameters between 0 and 1 were applied to each trapline dataset and the smoothing parameter achieving the smallest AICc value (i.e. the most parsimonious between model simplicity (a small \( f \) value) and fit (small residuals)) was selected.

The difference between harvest levels at the start and end of the time-series were used to calculate \( \lambda_{furbearer} \). To calculate the start and end values, a three-year average of the LOESS parameter estimates at either end of the time-series was used. Because the local regression uses a neighborhood function, data points near the beginning and end of the time-series typically possess less accurate smoothed parameter estimates (because of the reduced number of neighborhood points that can be drawn from). To compensate for this, starting and ending harvest estimates were taken 4-years in from either end of the time series.

3. Mantel’s Test

A Mantel’s test is used to analyze the spatial relationship between two variables (Mantel 1967; Legendre and Legendre 1998). I used a Mantel’s test to analyze the spatial relationship between \( \lambda_{mink} \) and each environmental variable. As well, the Mantel’s test was used to analyze the spatial relationship of current mink harvest (averaged 2000 to
2004) and each current environmental variable. In more simplified terms, the test examines whether geographic regions showing low and high \( \lambda_{\text{mink}} \) also show corresponding, or opposing, low and high levels in an environmental variable (relative level of mercury, for example). Both variables had to contain complete data sets with samples collected from the same locations. For each data set a matrix of the difference between each pair of data points is calculated, providing two distance matrices. The Mantel statistic \( (z_M) \) is then calculated as the sum of the cross-products of two distance matrices, as follows:

\[
z_M = \sum_{i=1}^{n-1} \sum_{j=i+1}^{n} x_{ij} y_{ij}
\]

where \( i \) and \( j \) are row and column values from each matrix (\( x \) and \( y \)). This equation can also be standardized, providing a product between -1 and +1 \( (r_M) \), which can be interpreted in a similar manner to a Pearson’s correlation value. The equation for the standardized Mantel’s test is:

\[
r_M = \frac{1}{d-1} \sum_{i=1}^{n-1} \sum_{j=i+1}^{n} \left( \frac{x_{ij} - \bar{x}}{s_x} \right) \left( \frac{y_{ij} - \bar{y}}{s_y} \right)
\]

where \( i \) and \( j \) are the row and column values from each matrix (\( x \) and \( y \)), \( d = [n(n-1)/2] \) is the number of calculated distances in the upper triangle of the distance matrix.

Significance is calculated through repeated tests of random permutations of rows and columns in one of the two matrices, providing a reference distribution against which the original Mantel statistic can be compared.

A partial Mantel’s test (Smouse et al. 1986; Fortin and Dale 2005) is the extension of the two-matrix analysis to three matrices. This analysis assesses the spatial correlation between two of the matrices while controlling for the effects of the third.
Controlling the effects of the third matrix is done by detrending both the first and second matrices by the third matrix using linear regression, and the resulting residuals are then applied to the standardized Mantel’s test. In all partial Mantel’s tests used in my study, the third matrix was geographic distance. The result of these tests was to assess the effects of geographic distance on each combination of mink and each environmental variable.

4. Akaike’s Information Criterion

Akaike’s information criterion based on small sample size (AICc) was used to assess the relationship between $\lambda_{\text{mink}}$ and all additive combinations of environmental variables (31 possible combinations). AICc uses an information theoretic approach to find the most parsimonious model by minimizing the amount of information lost from the suite of available models (Burnham and Anderson 2002). Because of the inherent spatial structure in geographically distributed data, a means of model regression that accounted for this spatial structure was necessary (Legendre 1993). Spatial autoregression using a lagged-predictor model was the regression method used (Rangel 2006), which is calculated using the following equation:

$$ Y = \rho WY + X\beta + WX\gamma + \varepsilon $$

where $\rho$ is the autoregression parameter, and $W$ represents a matrix of neighborhood weights ($w_{ij}$), $\beta$ is an autoregressive process accounting for the spatial autocorrelation of the response variable, and $\gamma$ is an autoregressive process accounting for the spatial autocorrelation of the predictor variables. The neighbourhood weights were defined as a
linear, inverse power function of geographic distance \((d_{ij})\) through the following equation:

\[
    w_{ij} = d_{ij}^\alpha
\]

where \(\alpha\) was left as 1, representing a linear relationship. The result of using the lagged-predictor model is the response variable is calculated as a function of both the explanatory variable and the values of the response variable at neighboring locations. As well, the effect of the predictor variable on the response variable is calculated after taking into account the effect of neighboring values. In general terms, spatial autoregression using a lagged-predictor model works by performing a regression of your predictor and response variables while weighting the influence of geographically adjacent samples in order to account for spatial autocorrelation (Rangel 2006). Spatial structure of the residuals for each model was examined to assure that spatial autocorrelation had been accounted for using the lagged-predictor regression method.

From these regression models the residual sums of squares (RSS) was used to calculate AICc using the following equation:

\[
    AIC_c = n \ln(RSS/n) + 2K + (2K(K+1))/(n-K-1)
\]

where \(n\) represents the size of the dataset and \(K\) represents the number of parameters (Burnham and Anderson 2002).

Akaike weights were then calculated by:

\[
    w_i = \frac{\exp(-0.5\Delta_i)}{\sum_{r=1}^{p} \exp(-0.5\Delta_r)}
\]

where \(\Delta_i\) represents the difference between the lowest scoring AICc model and model \(i\) and the denominator is the sum of the relative likelihoods for all models.
Relative parameter importance for parameter $j$ was calculated by summing $w_i$ in all models where parameter $j$ occurred. This measure facilitates the relative comparison of all the parameters used in the suite of models and gives a measure of the relative overall importance of each parameter.

Parameter estimates can vary between models, thus model averaged parameter estimates can be used to provide a single, overall estimate for each parameter (Burnham and Anderson 2002). Parameter estimates were only taken from the confidence set of models because the values calculated in low weight models are more likely to be inaccurate. Model averaged parameter estimates were calculated in four steps. First, the confidence set of models was selected (summed Akaike weights until a value $\geq 0.95$ is achieved) (Burnham and Anderson 2002). Using the confidence set, the Akaike weights are recalculated (see equation above). Raw parameter estimates (those calculated in the original suite of $\text{AIC}_c$ models) are then multiplied by the new Akaike weights, producing weighted parameter estimates. The weighted parameter estimates are then summed to provide the model averaged parameter estimate. Standard error (SE) estimates are calculated in a similar manner, multiplying the raw SE estimates by the new Akaike weights. The weighted SE values are then summed to provide the model averaged SE.