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Conservation Planning for Northern Leopard Frog, Mink and Belted Kingfisher in Eastern Ontario



Technical Series 2007

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TECHNICAL SERIES**

**CONSERVATION PLANNING FOR NORTHERN LEOPARD FROG,
MINK AND BELTED KINGFISHER IN EASTERN ONTARIO**

REPORT NO. 3-23

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NOTE TO READERS

The National Agri-Environmental Standards Initiative (NAESI) is a four-year (2004-2008) project between Environment Canada (EC) and Agriculture and Agri-Food Canada (AAFC) and is one of many initiatives under AAFC's Agriculture Policy Framework (APF). The goals of the National Agri-Environmental Standards Initiative include:

- Establishing non-regulatory national environmental performance standards (with regional application) that support common EC and AAFC goals for the environment
- Evaluating standards attainable by environmentally-beneficial agricultural production and management practices; and
- Increasing understanding of relationships between agriculture and the environment.

Under NAESI, agri-environmental performance standards (i.e., outcome-based standards) will be established that identify both desired levels of environmental condition and levels considered achievable based on available technology and practice. These standards will be integrated by AAFC into beneficial agricultural management systems and practices to help reduce environmental risks. Additionally, these will provide benefits to the health and supply of water, health of soils, health of air and the atmosphere; and ensure compatibility between biodiversity and agriculture. Standards are being developed in four thematic areas: Air, Biodiversity, Pesticides, and Water. Outcomes from NAESI will contribute to the APF goals of improved stewardship by agricultural producers of land, water, air and biodiversity and increased Canadian and international confidence that food from the Canadian agriculture and food sector is being produced in a safe and environmentally sound manner.

The development of agri-environmental performance standards involves science-based assessments of relative risk and the determination of desired environmental quality. As such, the National Agri-Environmental Standards Initiative (NAESI) Technical Series is dedicated to the consolidation and dissemination of the scientific knowledge, information, and tools produced through this program that will be used by Environment Canada as the scientific basis for the development and delivery of environmental performance standards. Reports in the Technical Series are available in the language (English or French) in which they were originally prepared and represent theme-specific deliverables. As the intention of this series is to provide an easily navigable and consolidated means of reporting on NAESI's yearly activities and progress, the detailed findings summarized in this series may, in fact, be published elsewhere, for example, as scientific papers in peer-reviewed journals.

This report provides scientific information to partially fulfill deliverables under the Biodiversity Theme of NAESI. This report was written by Golder Associates. The report was edited and formatted by Denise Davy to meet the criteria of the NAESI Technical Series. The information in this document is current as of when the document was originally prepared. For additional information regarding this publication, please contact:

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NOTE À L'INTENTION DES LECTEURS

L'Initiative nationale d'élaboration de normes agroenvironnementales (INENA) est un projet de quatre ans (2004-2008) mené conjointement par Environnement Canada (EC) et Agriculture et Agroalimentaire Canada (AAC) et l'une des nombreuses initiatives qui s'inscrit dans le Cadre stratégique pour l'agriculture (CSA) d'AAC. Elle a notamment comme objectifs :

- d'établir des normes nationales de rendement environnemental non réglementaires (applicables dans les régions) qui soutiennent les objectifs communs d'EC et d'AAC en ce qui concerne l'environnement;
- d'évaluer des normes qui sont réalisables par des pratiques de production et de gestion agricoles avantageuses pour l'environnement;
- de faire mieux comprendre les liens entre l'agriculture et l'environnement.

Dans le cadre de l'INENA, des normes de rendement agroenvironnementales (c.-à-d. des normes axées sur les résultats) seront établies pour déterminer les niveaux de qualité environnementale souhaités et les niveaux considérés comme réalisables au moyen des meilleures technologies et pratiques disponibles. AAC intégrera ces normes dans des systèmes et pratiques de gestion bénéfiques en agriculture afin d'aider à réduire les risques pour l'environnement. De plus, elles amélioreront l'approvisionnement en eau et la qualité de celle-ci, la qualité des sols et celle de l'air et de l'atmosphère, et assureront la compatibilité entre la biodiversité et l'agriculture. Des normes sont en voie d'être élaborées dans quatre domaines thématiques : l'air, la biodiversité, les pesticides et l'eau. Les résultats de l'INENA contribueront aux objectifs du CSA, soit d'améliorer la gestion des terres, de l'eau, de l'air et de la biodiversité par les producteurs agricoles et d'accroître la confiance du Canada et d'autres pays dans le fait que les aliments produits par les agriculteurs et le secteur de l'alimentation du Canada le sont d'une manière sécuritaire et soucieuse de l'environnement.

L'élaboration de normes de rendement agroenvironnementales comporte des évaluations scientifiques des risques relatifs et la détermination de la qualité environnementale souhaitée. Comme telle, la Série technique de l'INENA vise à regrouper et diffuser les connaissances, les informations et les outils scientifiques qui sont produits grâce à ce programme et dont Environnement Canada se servira comme fondement scientifique afin d'élaborer et de transmettre des normes de rendement environnemental. Les rapports compris dans la Série technique sont disponibles dans la langue (français ou anglais) dans laquelle ils ont été rédigés au départ et constituent des réalisations attendues propres à un thème en particulier. Comme cette série a pour objectif de fournir un moyen intégré et facile à consulter de faire rapport sur les activités et les progrès réalisés durant l'année dans le cadre de l'INENA, les conclusions détaillées qui sont résumées dans la série peuvent, en fait, être publiées ailleurs comme sous forme d'articles scientifiques de journaux soumis à l'évaluation par les pairs.

Le présent rapport fournit des données scientifiques afin de produire en partie les réalisations attendues pour le thème la biodiversité dans le cadre de l'INENA. Ce rapport a été rédigé par Golder Associates. De plus, il a été révisé et formaté par Denise Davy selon les critères établis pour la Série technique de l'INENA. L'information contenue dans ce document était à jour au moment de sa rédaction. Pour plus de renseignements sur cette publication, veuillez communiquer avec l'organisme suivant :

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EXECUTIVE SUMMARY

The goal of this report was to contribute to the development of biodiversity performance standards under the National Agri-Environmental Standards Initiative (NAESI). Specifically, the viability of regional populations of the northern leopard frog, mink and belted kingfisher were assessed under a habitat suitability framework in eastern Ontario (i.e., Stormont, Dundas and Glengarry). First, in cooperation with the NAESI 'GIS Team', a suite of GIS-based habitat metrics were calculated using previously published information on habitat-wildlife relationships and influences of human land-use activities. Habitat metrics were scored according to a metric's capacity to support a wildlife species and integrated into a Habitat Suitability Index (HSI) model. The northern leopard frog HSI was primarily based on breeding components and included wetland type, hydroperiod, influences of canopy cover, nearby roads and agriculture, and proximity to over-wintering habitat. Features in the mink model were related to food requirements and included proximity to water, aquatic habitat type, stand type and canopy cover. HSI modelling of the kingfisher was based on proximity to suitable nesting habitat and foraging requirements, which were related to aquatic habitat type and the condition of water (i.e., healthy or degraded).

Available patch habitat and carrying capacities on the landscape were calculated using RAMAS and GIS-based raster maps constructed from HSI models. Total habitat suitability available for populations of northern leopard frog = 356 ha, mink = 116 ha, and belted kingfisher = 166 ha. Approximate carrying capacities of northern leopard frog was 4 776 000 males, mink was 1100 individuals (both sexes), and belted kingfisher was 260 male birds on the study landscape. Using these estimates in RAMAS, Population Viability Analyses (PVA) assessed the vulnerability of regional populations (or metapopulations) to local extinction, and also the impact of human

activities by comparing results of models under varying habitat and demographic scenarios using sensitivity analyses. All simulations included variability to address uncertainty in risk assessments. Interestingly, projections varied considerably among the study species.

Simulations indicated that population sizes of northern leopard frogs were viable but declining and vulnerable to local extinction (i.e., extinction risk >10% over 100 years) when using either a non-spatial PVA or 'pessimistic' estimates of vital rates in simulations. A nonviable metapopulation of the northern leopard frog was also projected if wetland quantity and quality (i.e., habitat suitability) on the current landscape decreased by approximately 30%. Mean values of stage matrix parameters were particularly sensitive to changes in the PVA. However, isolation of subpopulations through habitat fragmentation and low dispersal rates had minimal impacts on the viability of the regional metapopulation. Baseline simulations of the belted kingfisher indicated a rapidly declining metapopulation and that the median time to local extinction was 37 years. This projection should be interpreted with caution, however, given that survival rates were highly sensitive parameters in the PVA and based on reported values for a conspecific. Watershed restoration for the belted kingfisher should proceed upon verification of vital rates. Further, the protection of overwintering habitat will be critical components in the conservation of the belted kingfisher, a migratory species. Mink simulations using both non-spatial population and spatial metapopulation models forecasted very low extinction risks for management purposes, even with the addition of harvesting. Local extinction risks of the mink remain low until a harvest of 40% juveniles and 20% adults, upon which regional populations may become vulnerable to local extinction. Simulations of mink abundance in this report were most responsive to changes to the dispersal matrix. Continued protection of aquatic habitat and riparian areas on the regional landscape is recommended for mink.

Although this report should provide guidance for regional conservation planning of the northern leopard frog, belted kingfisher and mink, additional research is required before the results from our simulations are used in making management decisions. HSI models must be tested for reliability by verifying the relation between model predictions and evidence of animal occurrence in the field, followed by calibration. It is also recommended that regional databases incorporate better descriptions of aquatic habitats to reduce uncertainty in mapping the distribution and abundance of riparian and wetland-dependent species. Finally, demographic research on vital rates is recommended such that local extinction risks can be reliably estimated.

1 INTRODUCTION

Conservation planning for species of conservation concern and biodiversity values is a common goal for industry, private landowners and public agencies. As a first step in conservation planning, one of the most popular approaches is the examination of wildlife-habitat associations and responses to human land-use activities through a Habitat Suitability Index (HSI) (e.g., Schamberger et al., 1982; Prose, 1985; Allen, 1986; Larson et al., 2004). HSIs assess the quality of a species habitat using relevant habitat attributes. A particularly useful means of quantifying habitat is through use of a Geographic Information System (GIS). To evaluate the threats faced by species in terms of their risk of extinction or decline, Population Viability Analyses (PVA) are often conducted, and have become an increasingly important tool in the conservation and management of species (Akçakaya et al., 2004). PVAs produce estimates of the probability of extinction and expected population (or metapopulation) sizes into the future based on information on species' age-specific survival rates, fecundity, and dispersal rates among populations. Thus, an effective approach to planning has involved incorporating assessments of habitat quality into models of wildlife population viability (e.g., Larson et al., 2004). A clear benefit of using an HSI framework for analyzing population viability is that simulations can project the effects of the current condition of the landscape and potential management strategies on the viability of populations.

The objective of this report was to complete habitat supply and population (or metapopulation) analysis for a group of surrogate species for conservation planning in eastern Ontario (Stormont, Dundas and Glengarry). The focal species in this report are: northern leopard frog (*Rana pipiens*), mink (*Mustela vison*), and belted kingfisher (*Ceryle alcyon*). The results will support the development of biodiversity performance standards under the National Agri-Environmental

Standards Initiative (NAESI).

2 HABITAT MODELLING

Habitat suitability models can identify important resources to wildlife, and therefore be important tools in planning their conservation in managed landscapes. The Habitat Suitability Index (HSI) is an efficient and inexpensive method for determining habitat quality (Schamberger et al., 1982), particularly when linked to remote sensing. The HSI is a numerical index that represents the capacity of a given habitat to support a focal fish or wildlife species. For example, the HSI has a minimum value of 0, which represents unsuitable habitat and a maximum value of 1.0 which represents optimal habitat. In this report, the HSI assumes a direct linear relationship between the HSI values and carrying capacity (i.e., number of individuals per unit area). Further, the HSI's were designed for forested landscapes of Eastern Ontario as specified by the 'Request for Proposal' (RFP; Nov. 2, 2006). Caution should be exercised when using these models for conservation planning in other eco-regions. Verification and validation of HSI's using presence data and abundance data are outside the scope of this report.

The first step in constructing an HSI model for each study species was to identify model variables for each life requisite (e.g., food, cover, nesting) per seasonal habitat component (e.g., breeding, non-breeding). Variables met three criteria: 1) the variable was related to the capacity of the habitat to support the species; 2) there was a moderate level of understanding of the relationship of the variable to the habitat (e.g., what is the best and worst condition for the variable); and 3) the variable was practical to measure within the constraints of the model application. Specifically, variables were those that could be easily measured through remote sensing or linked to digital land cover inventories. Spatial variables were also considered in describing habitat relationships.

In general, the overall suitability of the habitat increased as life requisites occurred closer together and as the overall quantity and quality of a life requisite resource increased.

After an index relationship was defined for each variable, it was aggregated with others into an index value. In this report, three types of relationships were considered for model-building: limiting factor, compensatory, and non-compensatory (multiplicative) relationships. The limiting factor relationship was considered if the variable with the lowest suitability overrode other variables setting the suitability index equal to the lowest variable. Compensatory relationships were used when a variable with marginal or low value was offset by the high suitability of other variables. Two simple mathematical functions can describe the compensatory relationship: the arithmetic mean and geometric mean. The geometric mean typically produces a smaller score than the arithmetic mean. The arithmetic mean can also be used to describe a relationship if two or more key variables in the model were very similar. If the relationship between variables was non-compensatory, then the index was simply the product of variables, which results in the lowest score compared to limiting factor and compensatory relationships.

2.1 Northern Leopard Frog Habitat

The northern leopard frog is a pond-breeding anuran with an aquatic and terrestrial life cycle. It requires marsh habitat for reproduction and permanent water for overwinter survival. During non-breeding periods in the summer, the frog prefers abandoned fields and meadows as foraging habitat. The main factor limiting sizes of post-metamorphic populations, however, is the quantity, quality and spatial arrangement of breeding habitats (Skelly et al., 1999; Pope et al., 2002; Gibbons et al., 2006).

The suitability of variables for the northern leopard frog HSI is described below. See Figure 1 for

the associated suitability index graphs which display the relationship between the variables and the index of suitability. The habitat model is largely based on breeding habitat components but considers adjacent terrestrial habitat that can influence both breeding conditions and foraging cover life requisites during post-metamorphic stages.

Breeding Habitat and Wetland Suitability

The suitability of wetlands was directly related to the quality of environmental conditions for developing larvae and was inferred by habitat type, structure of riparian zones and adjacent human land uses. The carrying capacity of wetlands was also related to the quality of adjacent foraging habitat.

- a) Optimal wetland conditions for high rates of juvenile recruitment to metamorphosis included marshes >1.0 ha in size. This wetland type may provide vegetation as cover from predators and sites for oviposition, and hydroperiods that extend through the larval period (i.e., summer) (Wellborn et al., 1996; Stevens et al., 2002; Babbitt et al., 2003).
- b) Optimal marsh habitat was characterized by <10% riparian canopy cover (Figure 1). Open canopy wetlands are preferred habitat types because they are characterized by warm water that enhances larval development (Werner and Glennemeier, 1999; Skelly et al., 2002). For example, Skelly et al. (1999) recorded northern leopard frogs on open-canopy ponds only in forested landscapes of Michigan, US.
 - Open canopy areas also provide optimal foraging conditions for post-metamorphic leopard frogs (Dole, 1965a, b; Merrell, 1977; Kolozsvary and Swihart, 1999; Guerry and Hunter, 2002).

- c) Medium suitability habitat was characterized by marsh with approximately 50% riparian cover.
- d) Compared to marsh, lakes provided medium suitability habitats. Although lakes provide deep waters persisting through the summer, they are often cold slowing larval development and comprised of predatory fish (Wellborn et al., 1996; Babbitt et al., 2003).
- e) Marginal breeding habitat occurred when riparian canopy cover approached >90%. A closed riparian canopy intercepts solar radiation, resulting in cooler water and delayed development of larvae.
- f) Marginal breeding habitat also occurs when either roads or intense agriculture occurred in close proximity to marsh habitat (i.e., <100m). Practices associated with intense agriculture (e.g., fertilizers, pesticides), and roads in close proximity to wetlands can result in low rates of survival of developing larvae and low production of juveniles (Bishop et al., 1999; Carr and Fahrig, 2001; Houlihan and Findlay, 2003; Environment Canada, 2004), which may result in smaller population sizes of post-metamorphic amphibians. For example, Houlihan and Findlay (2003) noted that amphibian abundance was negatively correlated with water quality (e.g., phosphorous and nitrogen parameters) and road density within 100 m of Ontario wetlands.
- Intense agriculture, and roads also fail to provide adequate cover and microclimates for post-metamorphic leopard frogs (Mazerolle and Desrochers, 2005). For example, Mazerolle and Desrochers (2005) demonstrated that when presented with a choice, 72% of northern leopard frogs avoided disturbed surfaces, such as agricultural fields.

Also, traffic associated with roads can cause heavy mortality in amphibian populations (Merrell, 1970; Carr and Fahrig, 2001; Houlahan and Findlay, 2003).

- g) Unsuitable breeding habitats included peatland and forested swamp (Figure 1) because they are typically characterized by environmental conditions unsuitable for developing larvae (e.g., short hydroperiod, low pH, cold water temperatures; Werner and Glennemeier, 1999; Stevens et al., 2002; Mazerolle and Desrochers, 2005).

The suitability of breeding habitats was also related to their proximity to overwintering sites.

- a) Optimal habitats were marshes (>10 ha), lakes, and watercourse >2nd order (Emery et al., 1972; Cunjak, 1986). These habitats should provide water conditions that do not freeze at the bottom for over-wintering individuals (Figure 1) (Snodgrass et al., 2000; Environment Canada, 2004). The 10 ha threshold was largely based on Snodgrass et al. (2000). They showed that all wetlands above this size were permanent waters, whereas wetlands below this size were observed as being either temporary or permanent waters.
- b) Marginal habitats were semi-permanent marshes <10 ha in size that were at distances of 2 km or more from an overwintering site.

2.1.1 Northern Leopard Frog HSI

$$\text{WetlandSuitabilityIndex} = \sqrt{(WH \times \%RCC)} \times D2PW \times AG100 \times RD100$$

WH - Wetland habitat types were ranked. Cells within marsh ≥ 1.0 ha = 1.0, marsh < 1.0 ha = 0.8, and lakes = 0.6. For lakes, only cells within 15 of the lake shoreline were assigned a rank, which = 0.6. All other cells = 0. All other aquatic habitat types had cells = 0.

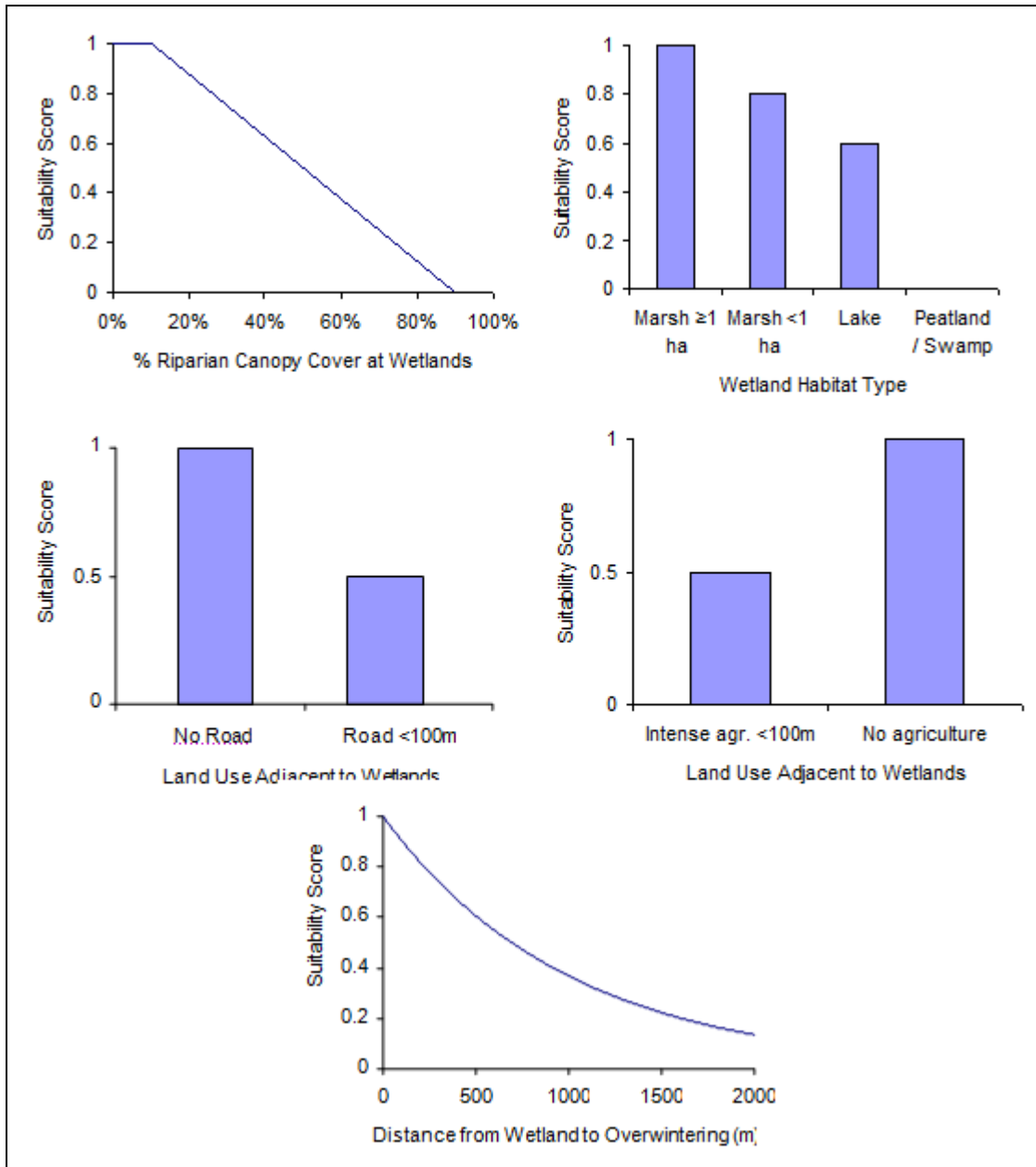
%RCC = % riparian canopy cover <50 m of site shorelines was calculated per wetland. Values were assigned to all cells within the wetland and scaled according to the following criteria: values <10 % = 1.0; and values >90% = 0.1. A linear relationship was used for between 10% and 90% (slope=-0.01125).

D2PW - Distance to permanent water was calculated per wetland. Values were assigned to all cells within the wetland. Permanent water was defined as marshes (>10 ha), lakes, streams/rivers (>2nd order). Wetland cells were ranked 1.0 at 0 km, and ranks approached 0.1 at 2 km. The relationship was based on a negative exponential model [$\text{score} = e^{(D2PW * -0.001)}$].

AG100 - Occurrence of ‘intense agriculture’ within 100 m of site shorelines was determined per wetland. Wetlands lacking nearby ‘intense agriculture’ had cells assigned a value of 1.0, and wetlands with adjacent ‘intense agriculture’ had cells assigned a value of 0.5.

RD100 - Occurrence of road within 100m of site shorelines was also determined. Wetlands without an adjacent road had cells assigned a value of 1.0, and wetlands adjacent to a road had cells assigned a value of 0.5.

Figure 1: Relationships between suitability scores and habitat for the northern leopard frog HSI.



2.2 Mink Habitat

Mink are a territorial, primarily nocturnal mammal, and are active year-round (DeGraaf and Rudis, 1986). They are opportunistic carnivores that consume small mammals (particularly

muskrats), fish, waterfowl, invertebrates, and amphibians (Melquist et al., 1981). Mink habitat preferences can be most succinctly described as an affinity for aquatic resources, such as those found in and near streams and rivers, lakes, and marshlands (Allen, 1986).

In the areas surrounding water, mink presence tends to be correlated with forest cover as well as low cover, such as brush, scrub, sedge, and even tall grasses (Marshall, 1936; Yamaguchi et al., 2003). Preferences for cover may be most pronounced in the selection of denning habitat, which tends to have higher than average shrub densities, tree densities, and coarse woody debris (Racey and Euler, 1983). However, exceptions do exist, and open fields with high microtine populations adjacent to rivers may be used by foraging mink (Loukmas and Halbrook, 2001).

In general, mink do not appear to be sensitive to the proximity of human disturbance (Allen, 1986). However, disturbance may still be a factor, as the number of resident female mink has been shown to be negatively correlated with human activity (Yamaguchi et al., 2003). In a more direct sense, human activities that reduce aquatic habitat productivity (e.g. stream channelization; Gray and Arner, 1977) or reduce cover (e.g., shoreline development; Racey and Euler, 1983) are almost certain to be detrimental to mink populations.

Many of the habitat features that would be good predictors of mink presence and abundance cannot be directly represented at the landscape scale, and correlates are likely to be unreliable. For example, the presence of river logjams, coarse woody debris on land, as well as shrub cover is likely of importance for good denning and forage opportunities (Melquist et al., 1981), but these features are certainly not available in the vegetation cover maps that are required for projecting models.

A previous habitat suitability model for mink, by Allen (1981), has recently been reported as

being ineffective (Loukmas and Halbrook, 2001). The habitat suitability model presented here attempts to represent habitat features that are likely to be directly or indirectly correlated with the carrying capacity of a habitat for mink populations, and that can be represented from available GIS landscape coverages. The features modeled include distance from water, stream or river width and speed, stand type, canopy cover, marshland area. The suitability of variables for the mink HSI model is described below. See Figure 2 for the associated suitability index graphs which display the relationship between the variables and the index of suitability.

Habitat Suitability

The suitability of mink habitat was related to cover and foraging habitat life requisites, both of which were inferred from forest composition, proximity to water, and aquatic habitat type. Proximity to water appears to be the most reliable predictor of mink presence.

- a) Optimum suitability habitats were areas adjacent to water. For example, in Michigan, mink strayed no further than about 30m from wetland (Marshall, 1936).

- b) Marginal suitability habitats were terrestrial areas approaching distances of 200 m from water. In west central Idaho, no mink were found further than 200m from streams (Melquist et al., 1981), and no further than about 180m in east-central Minnesota (Schladweiler and Storm, 1969).

It appears that the wider the stream or river, the better the associated habitat for mink. Sidorovich et al. (1996) found that there were an average of 1.9 mink per 10km of very small rivers (1-5m width), 3.8 per 10km of small rivers (5-11m width), and 6.3 per km of medium-sized rivers (11-15m width). There also seems to be a connection between increasing river speed and decreasing

mink preference. Sidorovich (2000) found that mink averaged 2.9 individuals per 10km of fast rivers, 6.3 individuals per 10km of moderately flowing rivers, and 10.7 individuals per 10km of slow rivers.

- a) Optimum suitability habitats were streams wider than 11m.
- b) Optimal habitats were fast rivers having slopes <2% (Rosgen, 1994, 1996).
- c) Marginal suitability habitats were streams 1-5m in width.
- d) Marginal suitability habitats had slopes >4% (Rosgen, 1994, 1996).

As mink strongly prefer foraging near water, it makes sense that ephemeral sources of water provide less appealing habitat. Indeed, Arnold and Fritzell (1990) found that mink avoided dry to nearly dry wetlands. In addition, it seems clear that the size of wetlands affects not only permanence, but also the diversity of habitats necessary to be productive wildlife habitat (Environment Canada, 2004).

- a) Optimal suitability wetlands were marsh >10ha. Snodgrass et al. (2000) found that marshes larger than 10ha in area were consistently permanent, while those less could be either permanent or semi-permanent.
- b) Optimal suitability wetlands were swamps >30ha. This threshold was based on Snodgrass et al. (2000) and the fact that swamps typically have shorter hydroperiods than marsh (Keddy, 2002).

- c) Marginal suitability wetlands were marsh <1ha. Todd Arnold (personal communication) found that the majority of mink use in southern Manitoba fell within wetlands greater than 1ha in size, which would seem to be a reasonable bottom threshold for permanence and preference.

- d) Marginal suitability wetlands were swamps <4ha.

Racey and Euler (1983) found that mink preferred coniferous forest cover, were neutral to mixed stands, and avoided deciduous stands. There may also be preference by mink for thick forest cover (Allen, 1986).

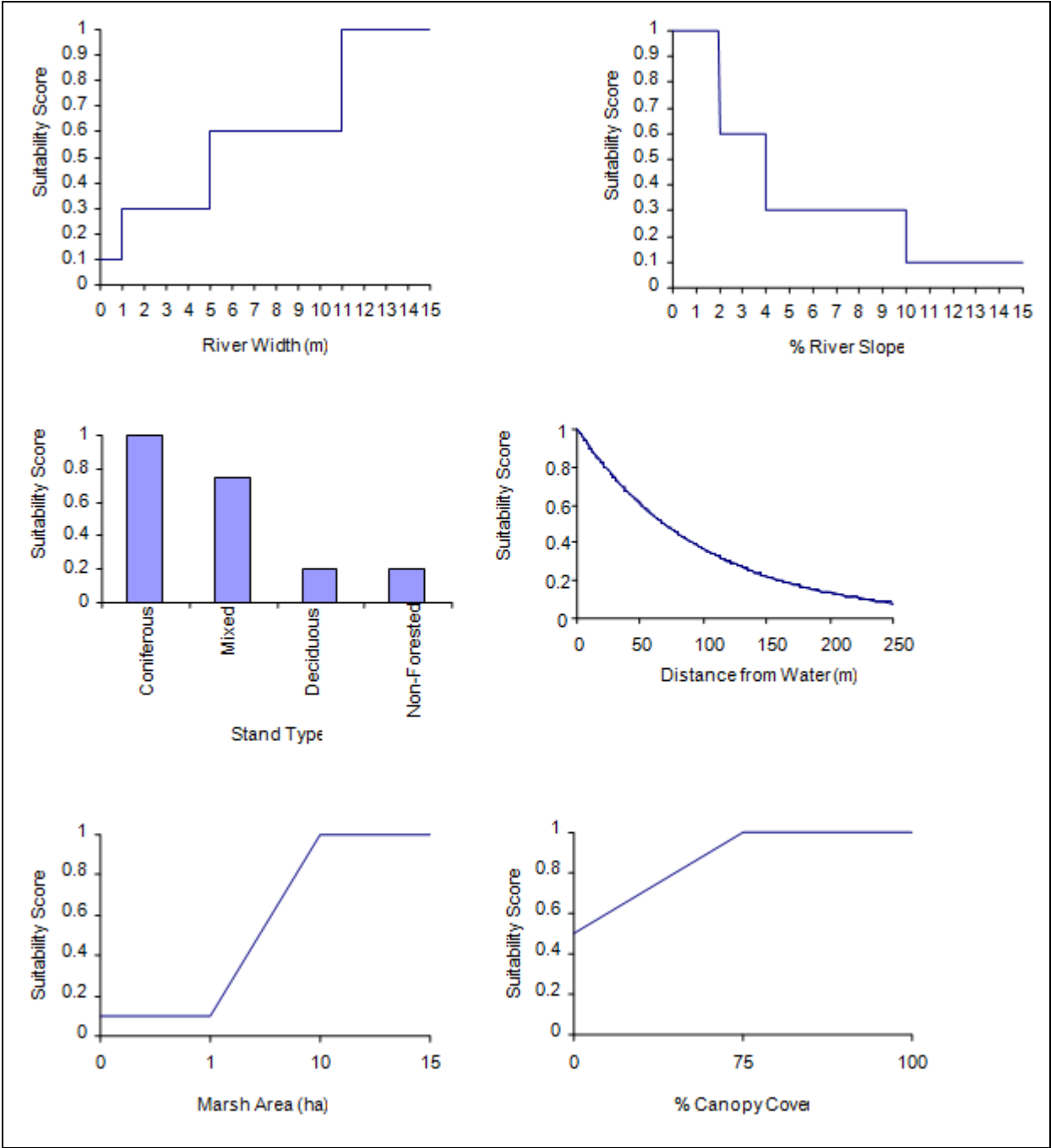
- a) Optimum suitability habitats were coniferous forest stands.

- b) Optimum suitability habitats were also areas with >75% canopy cover.

- c) Medium suitability habitats were areas with 0% canopy cover.

- d) Marginal suitability habitats were deciduous stands and non-forested areas.

Figure 2: Relationships between suitability scores and mink habitat for calculating the HSI.



Note that the 'Swamp Area' curve is not shown but is comparable to the 'Marsh Area' curve with the exception that thresholds are at 4 and 30 ha.

2.2.1 Mink HSI Formula

$$HSI = MAX \left(D2S * \sqrt{\left(\frac{\left(\frac{SW + SS}{2} \right) + STND}{2} \right)} * CC, D2W * \sqrt{\left(\frac{WET + STND}{2} \right)} * CC, D2L * \sqrt{STND * CC} \right)$$

D2S = Modifier related to distance to nearest stream/ river. Modifier is calculated as $e^{(distance * -0.01)}$. As this model is to be run at a 30m cell size, and distances from water per cell will be measured from a cell's centre, distance as measured must be adjusted. For example, a 30x30m cell immediately adjacent to water should have a distance of zero, and not 15m. This is solved by subtracting half a cell width from distance as measured.

D2W= Modifier related to distance to nearest wetland (i.e. marsh, swamp). Modifier is calculated as $e^{(distance * -0.01)}$.

D2L= Modifier related to distance to nearest lake. Modifier is calculated as $e^{(distance * -0.01)}$.

SW = Stream/ river width modifier. Less than 1m in width = 0.1. Very small (1-5m wide) = 0.3, small (>5-11m) = 0.6, medium to large (>11m) = 1. Stream width calculated at 2m cell size.

SS = Stream / river speed modifier, calculated on the basis of slope. Slow (<2% slope) = 1, moderate (2-4% slope) = 0.6, fast (>4-10% slope) = 0.3, steep (>10%) = 0.1.

STD = Stand type modifier. If the stand is coniferous, STD = 1. If the stand is deciduous, STD = 0.2. If the stand is mixed, STD = 0.75. If the stand is non-forested (not including human development), STD = 0.1. A coniferous stand is defined as one where the total percentage (FRI fields PERC1 through PERC10) of coniferous species (HE, L, LE, LJ, PJ, PR, PS, PW, SB, SN, SP, SW in FRI fields SP1 through SP10) is >75%. A deciduous stand is one where the total

percentage of coniferous species is <25%, while a mixed stand is one with 25-75% coniferous species.

CC = Cover modification. From 0.5 at 0 cover to 1 at 75% cover and above. Change between 0 cover to 75% cover defined as a straight line with equation

$$y = 0.5 + (0.0067 * \% \text{ canopy closure}).$$

WET = Wetland size modification. For marshes; <1ha = 0.1, 1ha = 0.1, to 1 at 10 ha and above. Change in modifier between 1 ha and 10 ha defined as a straight line with a slope of 0.1. For swamps; <4ha = 0.1, 4ha = 0.1, to 1 at 30ha and above. Change in modifier between 4 ha and 30 ha defined as a straight line with a slope of 0.0436.

2.3 Belted Kingfisher Habitat

The belted kingfisher is a water-obligate species of streams, rivers, lake and pond edges (Davis, 1982; Brooks and Davis, 1987; Sullivan et al., 2006), and large wetlands (Stevens, 2000). It is also a migratory species throughout much of Canada, including eastern Ontario. The HSI in this report identifies critical habitats during the breeding season only. Important life requisites for breeding populations of belted kingfishers include nesting habitat and food availability. The belted kingfisher uses vertical nesting banks with sandy soils for underground nests (Davis, 1982; Prose, 1985; Sayler and Lagler, 1949; Hamas, 1994). Interestingly, nest burrows are often excavated in areas disturbed by human activities that expose soil surfaces (Hamas, 1974). Some researchers suggest that its breeding distribution and densities may be more closely linked to suitable foraging versus nesting areas (Sayler and Lagler, 1949; Prose, 1985; Hamas, 1994; Sullivan et al., 2006). The primary prey of belted kingfisher is small fish (Davis, 1982). Davis

(1982) reports that the size of feeding territories is inversely proportional to both fish density, and that fitness is positively correlated with food density. The nature of kingfisher-habitat associations suggests that kingfishers rely on the condition of the in-stream and riparian habitat, and that birds may be particularly susceptible to impaired conditions (i.e., unhealthy streams). Indeed, the belted kingfisher is noticeably absent from turbid and eutrophic waters overgrown with vegetation (Reviewed in Hamas, 1994; Sullivan et al., 2006).

The suitability of variables for the belted kingfisher HSI is described below. See Figure 3 for the associated suitability index graphs which display the relationship between the variables and the index of suitability. The habitat model comprised a nesting habitat component and a foraging habitat (i.e., water quality and cover) component.

Nesting Habitat Suitability

The suitability of shoreline for reproduction was inferred by surrounding soil conditions. Optimum shoreline conditions were those within 3km of 'non-urban' soils comprised of >70% sand, such as sandy loam, loamy sand, and sand soil types at shoreline locations (Figure 3) (Brooks and Davis, 1987). Unsuitable shorelines were those >3km from sandy soils.

Foraging Habitat Suitability

First, the suitability of shorelines for foraging was inferred through reports of belted kingfisher densities on various aquatic habitat types. Territory size may be inversely related to food abundance (Davis, 1982). Smaller territories may contain richer food sources.

- Optimum shorelines were those associated with lakes and large rivers (≥ 5 order). In a study conducted in Michigan, lakeside territories averaged approximately 1.0 km of shoreline (Figure 3) (Salyer and Lagler, 1949). In the Oregon Coast Range, Loegering

(1998) reported that Belted Kingfisher densities were highest on 5th-6th order streams and approached one pair per km on these stream sizes.

- Medium suitability shorelines were those on large marshes. Although densities of territories for marsh habitat are not available in the literature, pairs can occur on marsh habitat, particularly on permanent wetlands (Stevens, 2000).
- Minimal suitability shorelines are those on small marshes and low-order streams. Both Sullivan et al. (2006) and Loegering (1998) reported no kingfishers on 1st-order stream. However, low-order streams obstructed by beaver damming can provide suitable habitat for belted kingfishers (Salyer and Lagler, 1947).

The suitability of foraging habitat was also related to fish habitat and water conditions through measures of agricultural land uses within the surrounding valley at multiple spatial scales (Allan, 2004). Agricultural land use degrades streams by increasing nonpoint inputs of pollutants, impacting riparian and stream channel habitat, and altering flows. Higher inputs of sediments, nutrients, and pesticides accompany increased agricultural use of sub-basins (Allan, 2004).

- a) Optimal water conditions occurred when <30% of the sub-basin was in agriculture (or, when >70% of the sub-basin was in forest cover) (Figure 3). For example, a study of agricultural streams in Wisconsin found indications of a decline in the biological integrity of streams when catchments comprised >30% of agriculture (Fitzpatrick et al., 2001).
- b) Optimal water conditions also occurred when riparian areas were comprised of >90% canopy cover. Research has shown that below this level, stream degradation usually occurs

(Allan, 2004). Similarly, Steedman (1987) reported that when riparian vegetation is <75% cover along 1st-3rd order streams, stream degradation occurs.

- c) Unsuitable water conditions occurred when >80% of the sub-basin was in agriculture (or, when <20% of the sub-basin was in forest cover). Allan (2004) suggests that agricultural land use has strong effects when it exceeds 50% of catchment area, and that the biological integrity of streams is almost always impaired when it exceeds 80% of catchment area.
- d) Unsuitable water conditions also occurred when there was <40% riparian cover.

The suitability of foraging habitat was also related to the presence of streamside trees.

- a) Optimal streamside habitats were those with riparian trees present within 10 m of the water edge (Figure 3). Loegering (1998) noted that belted kingfisher was 4.2-24 times more likely to use an area with trees immediately adjacent to the stream than reaches without streamside trees.
- b) Marginal streamside habitats were those lacking shoreline trees within 10 m of the water edge.

2.3.1 Belted Kingfisher HSI Formula

The belted kingfisher HSI comprised a food and spatial component. Only shoreline habitats within 3km of ‘non-urban’ sandy soils were considered for mapping.

$$\text{Foraging Habitat Suitability Index} = \left(\frac{SBA + SBF + RCC}{3} \right) \times AHT \times OST$$

SBA = % Sub-basin in agriculture was calculated per waterbody and stream section. Values were

then assigned to shorelines. Catchments with 0-30% agriculture = 1.0 and with >80% agriculture = 0. Catchments with between 30-80% agriculture were linearly scaled (slope = -0.02).

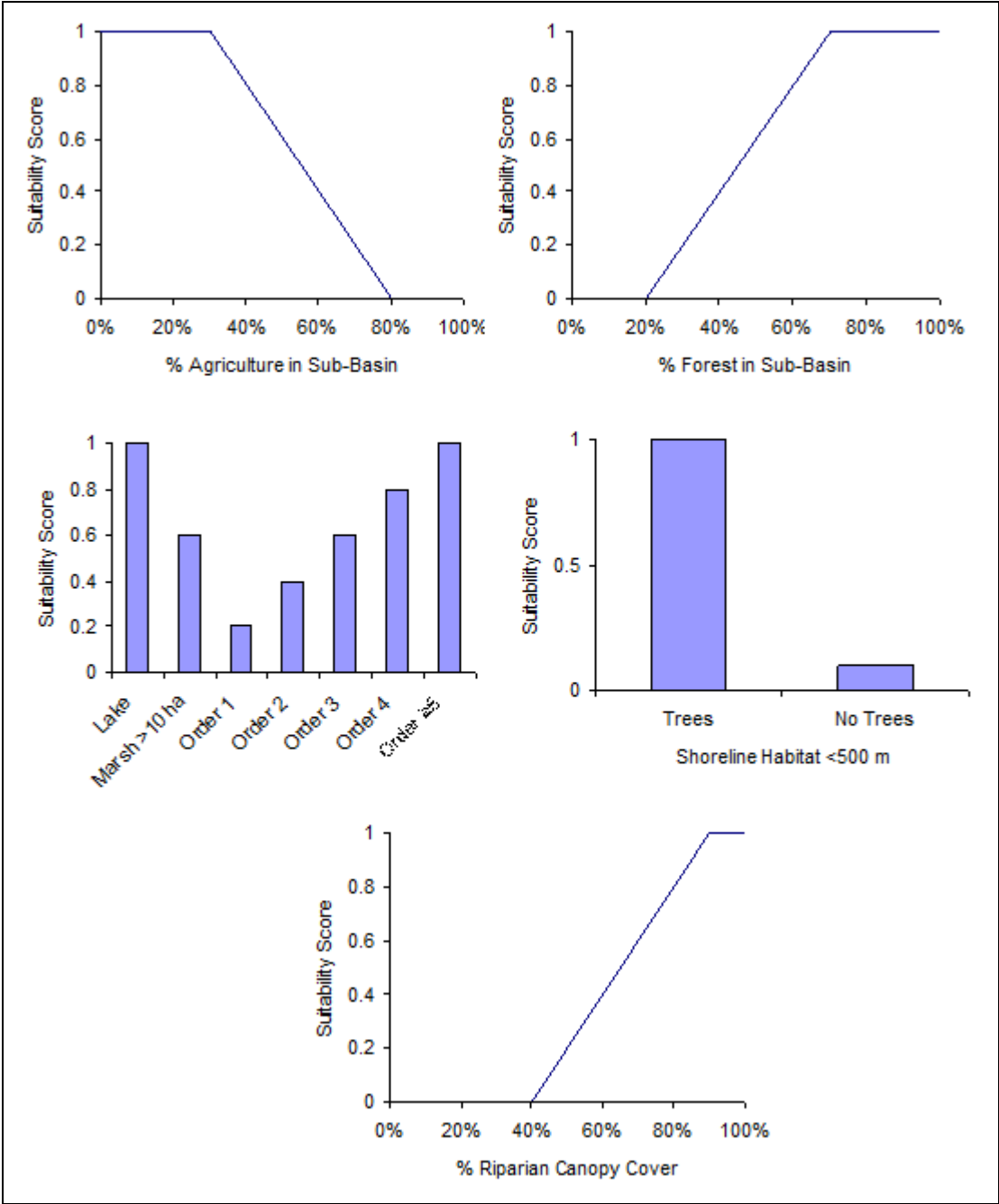
SBF = % Sub-basin in forest was also calculated per waterbody and stream section. Values were then assigned to shorelines. Catchments with >70% forest cover = 1.0, and with <20% forest cover = 0. Catchments with between 70% and 20% forest cover were linearly scaled (slope = 0.02).

RCC = % Riparian canopy cover (30 m from water's edge) in the sub-basin was also calculated per waterbody and stream. Values were then assigned to shorelines. Suitability scores were based on a linear relationship (slope = 0.02); >90% forest cover = 1.0 and <40% forest cover = 0.

AHT = Aquatic habitat types were ranked based on their classification, and ranks were assigned to shorelines. Lake shorelines = 1.0. Marsh and streams shoreline scores were based on their size. Marshes >10 ha = 0.6. Stream scores were based on a linear relationship (line slope = 0.2); stream order 1 = 0.2, and stream order 5 or higher = 1.

OST = Occurrence of shoreline trees (<10 m of waters edge) was determined per shoreline location. Locations with no trees 500m upstream and 500m downstream were assigned a score of 0.1; whereas locations with shoreline trees within 500m were assigned a score of 1.0.

Figure 3: Relationships between suitability scores and habitat feature for the belted kingfisher HSI.



3 POPULATION VIABILITY ANALYSES

Population Viability Analyses (PVA) is an increasingly important tool in the conservation and

management of species (Akçakaya et al., 2004). PVA makes use of estimates of a species' age-specific survival rates, fecundity, and dispersal rates among populations to produce estimates of the probability of extinction and expected population (or metapopulation) size into the future. In addition to assessing the vulnerability of focal species to local extinction in this report, the impact of human activities was evaluated by comparing results of models under varying habitat and demographic scenarios using sensitivity analyses. However, the accuracy and precision of estimates produced from the PVAs will depend on the accuracy and precision of estimates of vital rates, as well as on the assumption that the demographic models chosen were appropriate for the focal species. Although the general approach to modelling population viability was consistent among study species, model structures varied given inherent differences in the natural history and the mechanisms underlying population fluctuation of the various taxonomic groups involved (i.e., mammals, amphibians and birds). Thus, details of methods and assumptions for the PVAs are described under the species sub-headings further below. General assumptions of the PVAs include:

- Density was uniform throughout the study area, and only suitable habitat was included in estimates of total habitat suitability and carrying capacity.
- Vital rates used in the stage matrix reflected actual rates in eastern Ontario; these rates did not change in the future.
- When projecting local extinction risks, there was no change in the amount, quality or configuration of habitat during the simulation (50 or 100 simulation years).
- When modelling multiple sub-populations, each functioned as a discrete population loosely connected through dispersal, forming a metapopulation.

- Dispersal was considered as permanent movement of a proportion of individuals from one subpopulation (or habitat patch) to another in a single year.

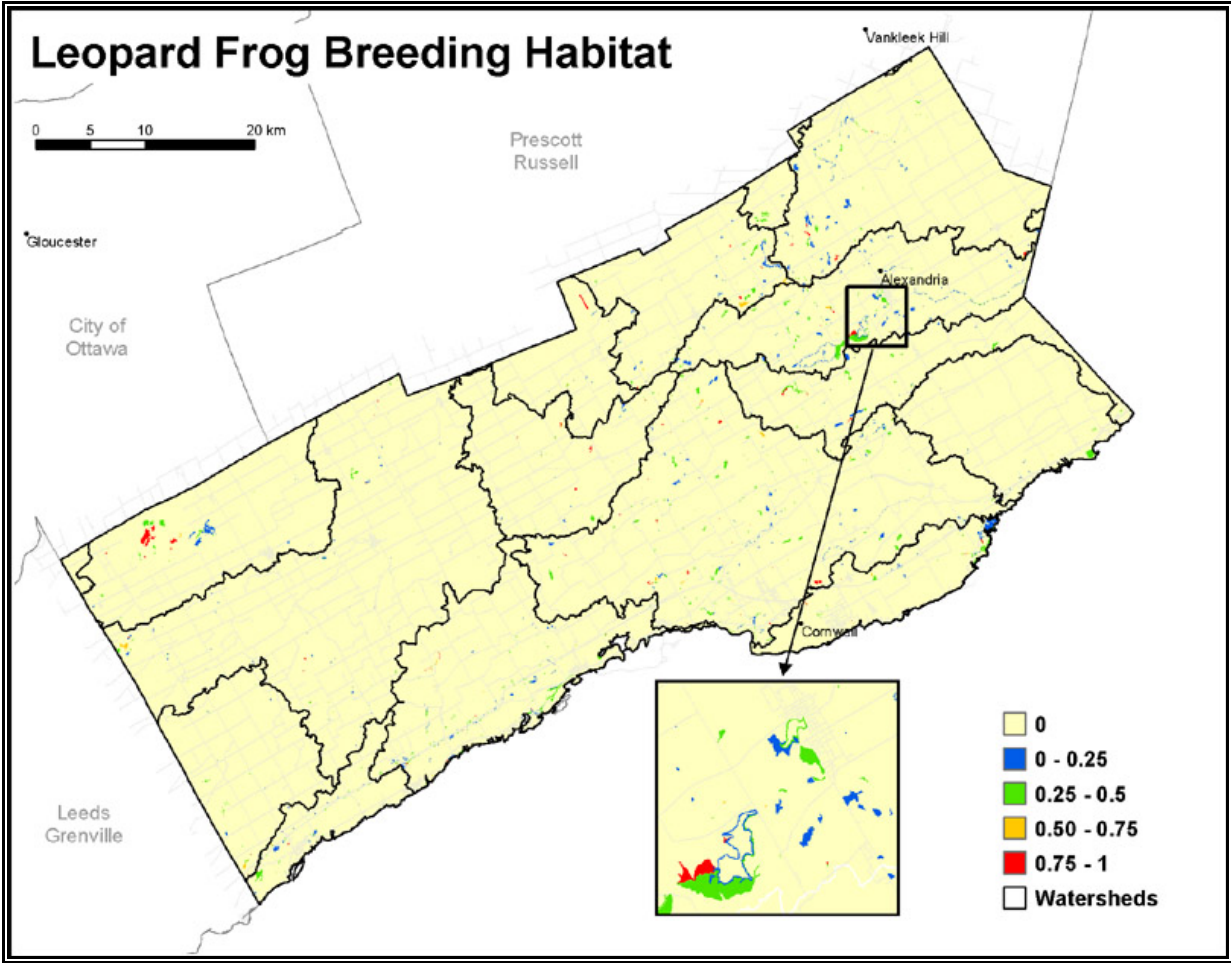
3.1 Northern Leopard Frog Populations

3.1.1 Methods

Habitat-Population Relationships:

Using the Spatial Data command in RAMAS, a habitat suitability map of the northern leopard frog (Figure 4) identified patches of breeding habitat, their carrying capacities and initial abundances and the relative fecundity of breeding frogs within patches. The map was comprised of 0.02 x 0.02 km cells scored according to the habitat suitability index (0-1) described earlier in this report. A patch was defined as a cluster of cells within 0.5 km from another cell (edge to edge); only cells having a habitat suitability score > 0.50 were considered. A 0.5 km distance was chosen as it may reflect the maximum foraging distance traveled by a post-metamorphic frog from a breeding pond during the summer months (Dole, 1965a; Merrell, 1970). To calculate carrying capacities per patch, total habitat suitability was multiplied by the number of post-metamorphic males estimated to occupy 400 m² of breeding habitat representing ideal environmental conditions (i.e., 536 male frogs). These estimates were based on Gibbons et al. (2006), which reported a density of approximately 984 juvenile (male and female) southern leopard frogs per 400 m² of water in a wetland in an agricultural landscape of South Carolina, USA. Initial abundances of patches were calculated as two-third the estimated carrying capacities. The local threshold at which a patch (i.e., subpopulation) was considered occupied was 100 male frogs. The mean habitat suitability scores of cells within patches were related to the relative fecundity of frogs breeding at that location.

Figure 4: Leopard frog breeding habitat identified from a Habitat Suitability Index (0-1; HSI) for eastern Ontario.



Stage Structure:

A Leslie matrix was used to model age-structured populations of post-metamorphic male northern leopard frogs: age-0 (juvenile or young-of-year), age-1 (sub-adult or immature), age-2 (mature adult) and age-3 (mature adult). It was assumed that male and female frogs had identical age structures and survival, and that sex ratios were one male to one female. There is no reason to suspect that sex ratios are strongly imbalanced (Merrel, 1968; Hine et al., 1981; LeClair and Castanet, 1987). A ‘birth-pulse’ population was assumed, in which all breeding takes place in a

short period of time. The Leslie matrix was based on a 'post-breeding' census of juvenile, age-1 and age-2 frogs and the assumption that no mortality took place between the breeding and the census (Table 1). Individuals were considered juveniles until maturity at age 2 (Leclair and Castanet, 1987; Gilbert et al., 1994). It was also assumed that individuals breed twice (Corn and Livo, 1989), reproduce only on their 2nd and 3rd birthday, and died after reaching their 3rd birthday.

Annual survival rates for adults and sub-adults were estimated at 40% (Merrell and Rodell, 1968; Merrell, 1977; Hine et al., 1982). Survival rates of newly metamorphosed juveniles were based on estimates from populations in Minnesota, Wisconsin and Alberta. The ratio of YOY frogs to sexually mature frogs ranged from 15:1-20:1 in Minnesota (Merrell, 1977). Similar values were reported in a breeding population in Wisconsin (Hine et al., 1981). These ratios indicate a 5-7% survival rate between metamorphs and sexually mature frogs. In Alberta, Seburn et al. (1997) observed ratios of approximately 120 juveniles to 1 sexually mature frog in a breeding population in the Cypress Hills suggesting 1% survival rate for juveniles to a sexually mature stage. Based on reported ratios of juveniles to breeding adults, the assumption that the total population also included 1-year old (non-breeding) sub-adults in the above-mentioned studies, and annual survival rates of sub-adults and adults, annual survival rates of juvenile northern leopard frogs were estimated at 9% in this report. Note that survival estimates used in this report are within the range used in other amphibian PVA studies (see Griffiths, 2004; Hatfield et al., 2004).

Fecundity rates were calculated by multiplying survival rate (0.4) by the average number of newly-metamorphic juveniles produced per individuals within each age class (Table 1). Models were based on post-metamorphic male age classes only. The number of new juvenile recruits per

clutch was estimated at 191 (range 7-376), which was based on the average number and range of eggs in a clutch and reported survival rates of egg to newly metamorphosed juvenile. Specifically, Corn and Livo (1989) recorded, on average, 3045 eggs per clutch (range 645-6272 eggs), whereas Merrell (1977) noted that individual egg clutches contain 2000-5000 eggs. Merrell (1977) also noted that survival rates during the tadpole stage are at least 6%, and possibly as low as 3%. Similarly, survivorship from total egg to metamorphosed young was 1-6% in Minnesota (Hine et al., 1981).

Table 1: Stage matrix comprised of fecundity (first row) and survival rates (± 1 SD) of young-of-year (YOY), sub-adult and breeding northern leopard frogs.

	YOY male	Sub-adult male	Adult male
YOY male	0	38.0 (± 37)	38.0 (± 37)
Sub-adult male	0.09 (0.07)	0	0
Adult male	0	0.4 (0.15)	0.4 (0.15)

Stochasticity:

Stochasticity was modelled by drawing values randomly from lognormal distributions described by fecundity and survival values and their associated standard deviations (Table 1). Standard deviation estimates for number of offspring produced and juvenile survival rates were calculated as approximately half the range reported for each parameter (also see methods in Griffiths, 2004). Because no variance estimates were available for adult survival rates, it was assumed that the coefficient of variation of adult survival rates was half that of juvenile survival rates (i.e., 39%; also see methods in Griffiths, 2004). The effects of stochasticity on fecundity, survival, and carry capacity were assumed to be correlated with a population. As additional elements of uncertainty, modelling incorporated two regional catastrophe scenarios. Drought as a 20% probability of a catastrophe was used to reduce the abundance of newly metamorphosed individuals by 90%

(Fowler, 1935; Merrell, 1977; Shriver and Gibbs, 2004). Similar probabilities were used for PVAs of birds on wetlands in northeastern United States (Shriver and Gibbs, 2004). The other scenario was of disease, such as fungal pathogens and viruses that can have drastic impacts on amphibian populations (Crawshaw, 1997; Kiesecker et al., 2001). Disease was modelled as a 5% probability catastrophe that resulted in the removal of all young-of-year, and 90% reduction in sub-adult and breeding adult abundance.

Spatial Structure and Dispersal:

The baseline models were simulated as a metapopulation comprising spatially-distinct subpopulations linked by dispersal (using the Metapopulation Model command in RAMAS). Although adults occasionally move between ponds, the main dispersal phase of amphibians, such as northern leopard frog, was assumed to be the juvenile phase (Dole, 1971; Smith and Green, 2005). For example, Dole (1968) noted that 98% of adults returned to their home pond after a 1 km displacement. It was also assumed that YOY were moderately-to-highly philopatric. Although it is not known what proportion of juvenile northern leopard frogs disperse, it was assumed that 15% of juveniles disperse from their natal pond to new (sub)populations. This estimate is based on a study of wood frogs (*Rana sylvatica*), which noted a similar dispersal rate of juveniles to new ponds to breed (Berven and Grudzien, 1990). The wood frog is a species of anuran with a comparable life history strategy and distribution in North America to the northern leopard frog (Russell and Bauer, 2000). With regard to dispersal distances, Seburn et al. (1997) reported that northern leopard frogs successfully dispersed to downstream ponds 2.1 km from the source site. Interestingly, one frog was recaptured 8 km from its natal pond. Smith and Green (2005) suggest that anurans, in general, have an average maximum dispersal movement of 2 km. Smith and Green (2005) also state that the frequency distribution of maximum movements for anurans is

well described by the inverse power law and that this relationship predicts that distances beneath 11-13 km are in a range that they may receive one emigrating individual. Thus, on the basis of these data, it was assumed that the average dispersal distance of northern leopard frogs was 2 km, and that the maximum dispersal distance was up to 10 km.

Density Dependence:

Density-dependent effects are poorly understood in post-metamorphic stage amphibians (Griffiths, 2004; Hatfield et al., 2004). A simple ceiling model was used that affected all vital rates and that was based on the abundance of all stages.

Simulations and Sensitivity Analyses:

The models projected population sizes with either non-spatial structure (1 group) or spatial structure with many (sub)populations linked by dispersal (i.e., metapopulation). Projections from the metapopulation model were also compared to those from a multi-population model without dispersal to examine for influences of habitat fragmentation and population isolation (through no dispersal movements) on population viability (e.g., Kolozsvary and Swihart, 1999). Sensitivity analyses were conducted on the metapopulation model only, which may be a more accurate description of the abundance and distribution of northern leopard frogs (Pope et al., 2000). All simulations were run over a 100-yr period (n=1000). At each time step, the number of juveniles, sub-adults and adults were projected, using a set of vital rates drawn from a random normal distribution with mean values taken from the stage matrix and standard deviations taken from the standard deviation matrix. A local extinction risk of <10% over 100 years was regarded as acceptably low for management purposes (Akçakaya et al., 2004; www.cosewic.gc.ca). To identify future vulnerable populations (e.g., to approximately 1000 breeding individuals; www.cosewic.gc.ca), quasi-extinction risks to 11,000 males was simulated over a 100 year

period.

To aid management decisions and determine the influences of model parameters on metapopulation viability, we conducted sensitivity analyses on catastrophe scenarios, carrying capacity, initial abundance, stage matrix values, and average and maximum dispersal distances. To conduct a sensitivity analysis, we varied each model parameter by 10% while holding others constant. Specifically, the relative influences of model parameters on terminal extinction risk of the regional metapopulation were evaluated. Risk curves were also plotted together and the location of the maximum difference (the threshold value at which the difference was maximum) between the base model and the test model was noted. Next, the Kolmogorov-Smirnov test statistic D was reported for the maximum difference value, and was based on a two-sample test. Upon identification of the most sensitive parameter, and if baseline simulations projected a viable metapopulation, the parameter value was adjusted to a ‘pessimistic’ level according to the range of values reported in the literature, followed by calculation of a new extinction risk for the regional metapopulation.

Two additional habitat-related scenarios were conducted. First, the effect of the addition of riparian buffers from roads and intense agriculture on the number of habitat patches and total suitable habitat for the northern leopard frog was evaluated. Specifically, the northern leopard frog HSI formula was revised by removing the agricultural and road components [i.e., $(WH \times \%RCC)^{1/2} \times D2PW$; see Section 2.1.1]. Another scenario was executed to simulate the potential impacts of lakeshore development and wetland destruction. Snell (1987) estimated that from 1967 to 1982, 5.2% of wetlands were lost in southern Ontario, with a maximum of 30% lost from any one county (Kent County). Thus, the effect of a 30% reduction in habitat quality and quantity

(i.e., habitat suitability) on the current landscape on the viability of the northern leopard frog metapopulation was evaluated.

3.1.2 Results

RAMAS identified 215 patches or sub-populations of breeding habitat in eastern Ontario that had a total habitat suitability of 8910 (mean habitat suitability = 0.69) (see Appendix A). Carrying capacity of the landscape was 4 775 931 male frogs, whereas starting abundance of male frogs was 3 180 987 individuals.

Both non-spatial and spatial models were used to project abundance 100 years into the future. The non-spatial structure assumed one population (i.e., one habitat patch) and no dispersal, whereas spatially-structured models assumed 215 sub-populations with either dispersing individuals or no movement between habitat patches. The spatially-structured model with dispersal among breeding patches was considered the baseline model. Non-spatial simulations indicated that the terminal population size was reduced 84% over a 100 year period. The final population size (+1SD) was 509 734 males (+1 732 445), and the terminal extinction risk was estimated at 0.36 (0.332-0.388, 95% CI).

Projections of metapopulation sizes based on the baseline model (215 subpopulations linked by dispersal) also indicated that the number of male frogs was lower than the starting population (55% lower). The final population size (+SD) was 1 424 813 males (+2 517 024). The probability of local extinction (100% decline) at year 100 was estimated as 0.012 (0-0.04, 95% CI), which is acceptably low for management purposes (i.e., <10%). However, the cumulative probability of quasi-extinction (to below 11,000 male frogs) was estimated at 0.3450 (0.317-0.373, 95% CI). The trajectory summary of the spatial PVA without movement between habitat patches (i.e.,

isolation scenario) was comparable to projections from the baseline model. For example, the probability of local extinction remained low at 0.018 (0-0.046, 95% CI). Also, the confidence interval associated with the cumulative probability of quasi-extinction (to below 11,000 male frogs) overlapped with that of the baseline model. For the isolation scenario, the cumulative probability of quasi-extinction was 0.385 (0.357-0.413, 95% CI).

Sensitivity analyses of the metapopulation model (based on 10% changes to parameters) indicated that of all parameters tested, the stage matrix was the most sensitive parameter. It was the only parameter to significantly increase terminal extinction risk (i.e., probability of 100% decline over 100 years) based on non-overlapping confidence intervals between new and baseline projected outcomes (Table 2). The new extinction risk of the regional metapopulation was 0.085 (0.057-0.113, 95% CI). Of stage matrix parameters, adult fecundity and juvenile survival were almost two times more sensitive than adult survival (Table 2). Sensitivity analysis of adult fecundity and juvenile survival showed that both parameters were similarly sensitive to changes in their mean values, and decreased final metapopulation sizes an additional 28%, when compared to the terminal projection from the baseline model (Table 2). It should be noted that times to local extinction and quasi-extinction were >100 years for all scenarios.

Table 2: Sensitivity analyses showing further decreases in regional metapopulation projections at year 100, and new probabilities of terminal extinction risk (100% decline). Confidence intervals were based on the Kolmogorov-Smirnov test statistic

	Terminal population size	% decrease in population size	Extinction Risk (100% decline)	95% CI
Baseline	1424813	--	0.012	0, 0.04
Model Parameter				
Initial Abundance	1379875	3.2	0.013	0, 0.041
Average Dispersal Dist.	1346043	5.5	0.013	0, 0.041

Table 2: Sensitivity analyses showing further decreases in regional metapopulation projections at year 100, and new probabilities of terminal extinction risk (100% decline). Confidence intervals were based on the Kolmogorov-Smirnov test statistic

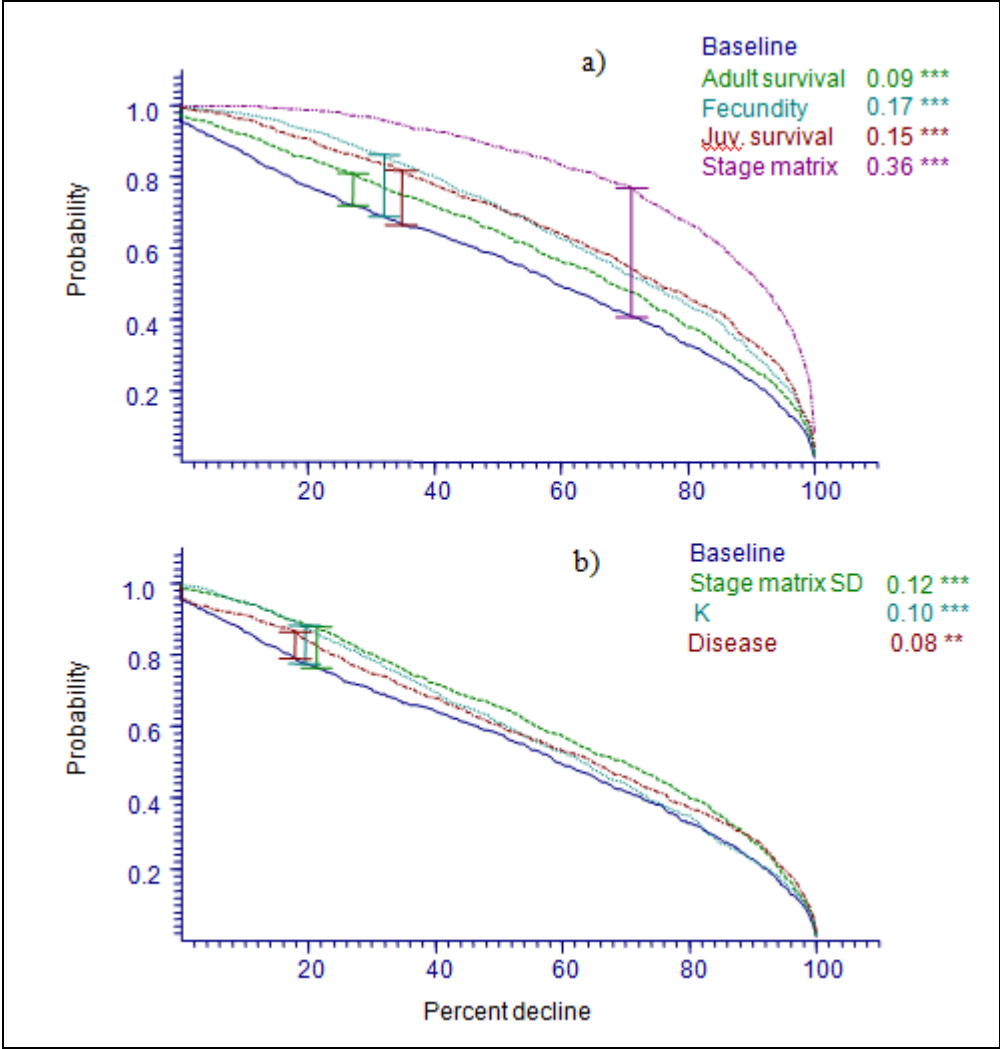
	Terminal population size	% decrease in population size	Extinction Risk (100% decline)	95% CI
Drought	1339028	6.0	0.023	0, 0.051
Max. Dispersal Dist.	1327889	6.8	0.02	0, 0.048
Disease*	1292382	9.3	0.027	0, 0.055
K ^{*a}	1276381	10.4	0.017	0, 0.045
Stage Matrix SD*	1177734	17.3	0.017	0, 0.045
Stage Matrix Means ^{*a}	566674	60.2	0.085	0.057, 0.113
Stage Matrix Parameter				
Adult survival*	1225727	14.0	0.021	0, 0.049
Juvenile survival*	1026629	27.9	0.041	0.013, 0.069
Adult fecundity*	1024072	28.1	0.028	0, 0.056
* <i>Maximum difference between model's risk curve and the baseline risk curve was significant at alpha = 0.05; a non-overlapping 95% confidence interval with those of baseline model.</i>				
^{a)} <i>Indicator of available suitable habitat.</i>				

In addition to the stage matrix and associated parameters, stage matrix SD, carrying capacity (as an indicator of total available habitat) and catastrophe (disease) were classified as being sensitive parameters based on comparisons of decline risk curves and the Kolmogorov-Smirnov test statistic. When compared to the curve constructed from the original (i.e., baseline) model, the modified models produced curves that were significantly different at locations of maximum difference ($P < 0.05$). Sensitive parameters and their decline risk curves are shown in Figure 5.

Based on sensitivity analyses, a worst-case scenario was modelled using reduced survival rates of juvenile northern leopard frogs (i.e., by 50%). The juvenile survival parameter was chosen for re-calculating local extinction risks because the rate used in the baseline model (i.e., 9%) may be too

high compared to other interpretations (e.g., Alberta SRD, 2003) of ratios of young-of-the year to breeding adults reported by Merrell (1977) and Seburn et al. (1997). Simulations indicated that the pessimistic extinction risk was 0.357 (0.329-0.385, 95% CI), and that current metapopulation of northern leopard frogs in eastern Ontario is vulnerable to local extinction.

Figure 5: Sensitivity analyses based on decline risk curves simulated from baseline models, and models with 10% changes in a) means of stage matrix parameter and b) values associated with other sensitive parameters. Maximum differences between baseline and new risk curves, and their statistical significance are noted (*p<0.001, **p<0.01).**



Using the spatial data command, the addition of riparian buffers from roads and intense agriculture had a significant impact on the number of habitat patches and total suitable habitat for the northern leopard frog. The number of habitat patches (or subpopulations) increased in eastern Ontario by over 250% (i.e., >500 patches). Unfortunately, a PVA based on this landscape configuration could not be conducted due to computational demands of evaluating over 500 patches.

The second habitat scenario of a 30% reduction in the quality and quantity of breeding habitat resulted in the identification of 129 habitat patches with a total habitat suitability of 4355. The carrying capacity of this modified landscape was 2 334 205 male individuals. Based on this spatial and population information, a PVA indicated that the probability of local extinction was 0.117 (0.089-0.145, 95% CI). Further, the median time to quasi-extinction was calculated as 57 yrs.

3.1.3 Conclusion

Assuming that habitat quality and quantity remains constant over the next 100 years, and that actual demographic rates are closer to those used in the optimistic versus pessimistic PVA, the northern leopard frog metapopulation in eastern Ontario is viable and should persist over the next 100 years (i.e., probability of local extinction <10% over 100 years). This result is consistent with status reports on population trends of northern leopard frog in eastern Canada ([Report3-22\(B9C\).doc](#)). However, an environmentally conservative approach to landscape planning is recommended given that stage matrix parameters were highly sensitive parameters, and that ‘pessimistic’ simulations using reduced juvenile survival rates (by 50%) resulted in non-viable metapopulation projections. Further, there is a 35% probability that the

leopard frog metapopulation will be reduced to a level that is susceptible to local extinction (i.e., 1000 breeding adults; www.cosewic.gc.ca) in 100 years. Thus, at the very least, conservation planning should aim to protect the quantity and quality of wetlands on the current landscape. Interestingly, isolation of subpopulations through reduced dispersal movement had minimal impacts on the viability of the regional metapopulation.

If demographic rates are closer to those used in the pessimistic PVA, habitat enhancements and restoration strategies are recommended. Specifically, protection of riparian zones from human activities, such as pesticide use, may improve water conditions and promote both higher rates of juvenile recruitment and larger body sizes of newly metamorphosed juveniles (Relyea, 2003). Preliminary analyses indicate that the addition of riparian buffers from roads and intense agriculture may have a significant impact on the number of habitat patches and total suitable habitat for the northern leopard frog. For example, revising the northern leopard frog HSI formula by removing the agricultural and road components [i.e., $(WH \times \%RCC)^{1/2} \times D2PW$; see Section 2.1.1], increased the number of habitat patches (or subpopulations) in eastern Ontario by over 250% (i.e., >500 patches). In contrast, a 30% reduction in the quality and quantity of breeding habitat on the study landscape resulted in the northern leopard frog metapopulation being vulnerable to local extinction. Wetland mitigation strategies that replace wetland area that would otherwise be lost on the landscape (Stevens et al., 2002), and management strategies that prevent disease transmission through fish stocking programs (Kiesecker et al., 1999) will be critical to the persistence of the northern leopard frog metapopulation.

Finally, given the paucity of data on survival and fecundity rates, and their variability, research based on understanding year-to-year and among population variability in demographic parameters

will have an important role in improving the reliability in estimating local extinction risks for the northern leopard frog. The accuracy of estimation of juvenile survival rates used for this report requires further investigation as highlighted by the sensitivity analyses and trajectory summaries under a pessimistic scenario. Validation of the HSI used to identify breeding habitat will also be critical in resolving uncertainty in PVA projections. In the meantime, it is recommended that land managers use an environmentally conservative approach for the northern leopard frog given the current level of uncertainty and the recent decline of amphibians around the globe.

3.2 Mink Populations

3.2.1 Methods

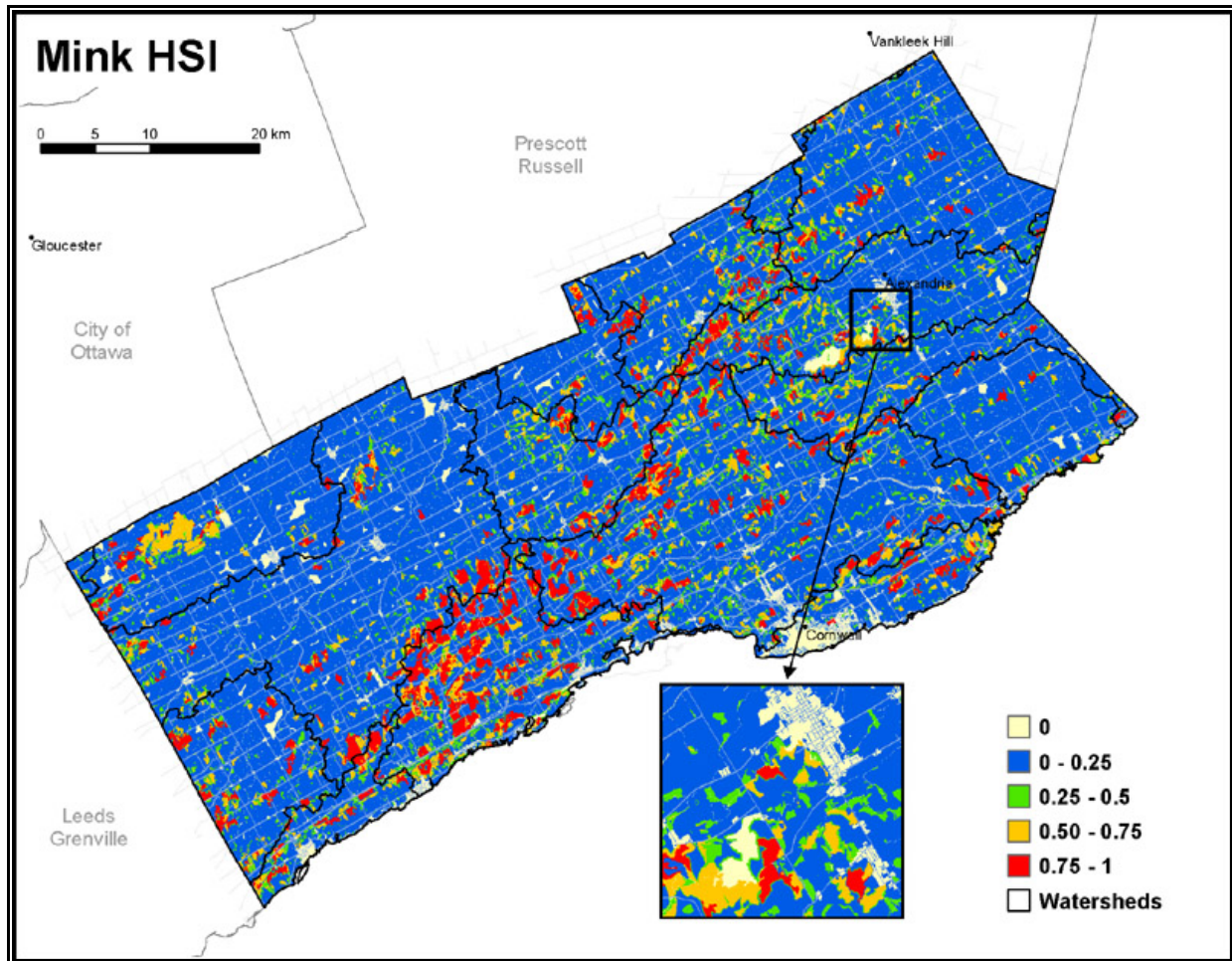
Habitat-Population Relationships:

Maps of mink habitat suitability (Figure 6) were input into the Spatial Data component of RAMAS. Minimum habitat suitability necessary for breeding was set to 0.4. Initial abundance and carrying capacity per population were calculated in the Spatial Data subprogram of RAMAS by summing the total of all habitat suitability scores across the landscape and multiplying by the maximum mink density estimated from literature. Sidorovich et al. (2000) and Allen (1986) suggest an optimum population density of about 4 individuals per km². This translates to 0.004096 individuals per 32m x 32m landscape pixel, and is calculated in RAMAS by multiplying 0.004096 by the total habitat suitability (i.e. the sum of pixel HSI values in a population). A threshold function was included in the calculation of patch carrying capacity such that only patches with 10 or more individuals would be considered a population.

Two spatial layers were produced for the Metapopulation module of RAMAS. In the first, a neighbourhood distance of 50 cells was used to approximate a home range estimate of 1.5 km (Gerell, 1970). In the second, a neighbourhood distance of 200 cells was applied based on the

average home range of 6.5 km found by Arnold and Fritzell (1989).

Figure 6: Mink habitat identified from a Habitat Suitability Index (0-1; HSI) for eastern Ontario.



Stage Structure:

Demographic information for the PVA was either estimated, or taken directly from available literature. Table 3 comprises the matrix of fecundity and survival rates at age (Table 3). Both sexes were modelled, as some inter-gender territoriality appears to be present (Gerell, 1970). Survival estimates were based on Bonesi et al. (2006). Pre-breeding data from the previously non-

culled Hiiumaa population was used for estimates of adult mortality in a stable age distribution. A full year of census data collected from the South Harris population was used for a combined estimate of juvenile and sub-adult mortality. The authors suggest that estimates of juvenile mortality as reported may be exaggerated, and the potential impacts of this will be investigated in sensitivity analysis.

Initial fecundity rates were calculated by multiplying four kits per female (Gerrell, 1971) by survival rates of juveniles and subadults. However, litter sizes have been found to range between three (3-4; DeGraaf and Rudis, 1986, Smal, 1991) to five (4-5; Nichol, 2002, Eco-West Environmental Services, 2003) and even six kits (Gerrell, 1971). It was assumed that female mink reproduce in the first breeding season after their birth (Hatler and Beal, 2003), and that no mink survive to their sixth birthday. This translates to five age classes in the stage matrix (i.e., juveniles and years 1 through 4), as pre-breeding sourced survival estimates hold implicit that five year old individuals are present, but do not survive to the following breeding season (after which they would be six years old). In fact, mink may live to six years old and perhaps beyond, however the available evidence suggests that the proportion of the population at that age is likely negligible (e.g. Bonesi et al., 2006).

A standard deviation matrix was produced using combined estimates of variation in birth rates and juveniles mortality for the standard deviation of fecundity. In the absence of data, standard deviations of adult survival rates was set at 0.05.

Table 3: Stage matrix comprised of fecundity (first row) and survival rates (± 1 SD) of juvenile through 4 year-old mink.

	Juveniles	1 Year	2 years	3 years	4 years old
Juveniles	0.6 (0.24)	0.6 (0.24)	0.6 (0.24)	0.6 (0.24)	0.6 (0.24)
1 year old	0.86 (0.05)	0.0	0.0	0.0	0.0

Table 3: Stage matrix comprised of fecundity (first row) and survival rates (± 1 SD) of juvenile through 4 year-old mink.

	Juveniles	1 Year	2 years	3 years	4 years old
2 years old	0.0	0.9 (0.05)	0.0	0.0	0.0
3 years old	0.0	0.0	0.5 (0.05)	0.0	0.0
4 years old	0.0	0.0	0.0	0.33 (0.05)	0.0

Density Dependence:

A ceiling model of density dependence was selected to avoid unnecessary complexity, as well as to remain consistent with territorial populations (Akçakaya, 2005). Territoriality is implicit in the maximum population density estimated from literature.

Dispersal:

Mink dispersal occurs primarily in the juvenile stage (Gerell, 1970). However, it seems likely that adults, primarily males, may travel considerable distances in search of mates, or may be forced to disperse if ousted from a territory. Relative dispersal was therefore set such that adult mink dispersed at 10% of the rate for juvenile mink. The maximum dispersal distance observed is 45 km for a dispersing juvenile (Mitchell, 1961; Gerell, 1970). This number is fairly consistent with the output of the allometric scaling equation Sutherland et al. (2000) developed to calculate maximum dispersal distances for carnivores by body weight (approximately 39 km for females, 42 km for males). Results from mark-recapture studies by Gerell (1970) suggest that approximately 70% of juveniles will disperse outside of a 10,000 ha area (10km x 10km). This implies that 70% of juveniles will disperse over 5km on average. To model this, a dispersal function was produced in RAMAS Metapop, with coefficient b set to 16.95, D_{max} set to 45, and the remaining coefficients (a and c) to 1. The dispersal-distance function of RAMAS represents an exponential decay curve according to;

$$m = a \times \exp\left(\frac{-D^c}{b}\right)$$

where m is the dispersal rate, and D is the distance between two populations (Akçakaya, 2005). If the distance between two populations is greater than D_{max} , the dispersal rate is automatically set to 0. The baseline scenario was also run without dispersal to evaluate the impacts of metapopulations dynamics on extinction risk.

Density dependence was incorporated into dispersal by including a density dependent slope for each population such that the rate of dispersal would decrease linearly from that represented in the dispersal function at carrying capacity, down to 0 at a population size of 0.

To calculate dispersal rates among subpopulations, the dispersal matrix was first filled automatically using the dispersal-distance function. However, based on preliminary analyses these values had to be further adjusted because smaller populations were acting as catastrophic demographic ‘sinks’. The territorial nature and dispersal characteristics of mink (i.e., 100% juvenile dispersal over short distances), and the fact that two of the three population were only 1.5% and 6.9% the size of the larger population meant that values calculated by the dispersal-distance function were only appropriate for populations dispersing into equal or larger sized populations. Therefore, in the dispersal matrix, dispersal rates were decreased by a proportion equal to the proportional difference in population sizes at carrying capacity wherever there was dispersal into a smaller population.

Stochasticity:

Stochasticity was modelled by drawing values randomly from lognormal distributions described by the fecundity and survival values, and their associated standard deviations. The effects of

stochasticity on fecundity, survival, and carrying capacity were assumed to be correlated with a population, and the local extinction threshold. A 10% probability of a catastrophic regional drought was introduced to represent the possibility of a dramatic drop in carrying capacity due to a reduction in aquatic prey. The periodicity of drought in south-eastern Ontario has been found to range between 0.05 and 0.1 (10 to 20 years per cycle; Girardin et al., 2004). All simulations performed in RAMAS Metapop were conducted over 50 year time frames with 500 replications.

Wetland Development:

A scenario was enacted to simulate the potential impacts of lakeshore development and wetland destruction. Snell (1987) estimated that from 1967 to 1982, 5.2% of wetlands were lost in southern Ontario, with a maximum of 30% lost from any one county (Kent County). Some wetlands were created as well, leading to a net loss of only 1.8%. However, the relative quality of newly created wetland is questionable. Estimates may be underestimating the negative impacts on mink, which are sensitive to not only losses of wetlands, but also development that simplifies vegetation structure at the cost of low cover (Racey and Euler, 1983). Four development scenarios were investigated by reducing total habitat suitability across the landscape. The scenarios involved 5%, 30%, and 50% reductions in total habitat suitability.

Harvest Scenario:

A harvest scenario was simulated to estimate the impacts of various levels of harvest on population projections. Scenario one involved a 40% harvest of juveniles and a 20% harvest of adults from each population. Harvest occurred every year prior to dispersal to represent the fact that individuals are most susceptible during dispersal (i.e. they never reach their potential destination). Scenario two was a 10% harvest of juveniles and a 5% harvest of adults from each population. The greater proportion of juveniles harvested is meant to convey the greater

susceptibility of juveniles to trapping, particularly during dispersal (Bonesi et al., 2006). Harvest was set to occur before dispersal, representing the concept that individuals are most susceptible during dispersal, and likely will not reach their target population.

3.2.2 Results

For the 50 cell neighbourhood (1.5 km average home range), RAMAS identified three populations (with greater than or equal to 10 individuals), while for the 200 cell neighbourhood (6.5 km average home range) only one population was identified in eastern Ontario (see Appendix A). Carrying capacities based on both neighbourhood sizes along with reductions in total habitat suitability associated with the ‘development’ scenario are shown in Table 4. Modelling without dispersal for the three population (i.e. the 50 cell neighborhood) scenario did not show any variation in results from scenarios where dispersal was occurring.

Table 4: Carrying capacities of mink per neighborhood size used for identifying habitat patches, and reductions in habitat suitability simulated for the ‘development’ scenario.

Total Habitat Suitability	Carrying Capacity Per Neighbourhood Size	
	1.5 km (50 cell) ^a	6.5 km (200 cell) ^b
100%	1138	1155
95%	1080	1097
70%	796	809
50%	561	578

a) resulted in three population model (i.e., spatial model)

b) resulted in one population model (i.e., non-spatial model)

Sensitivity Analysis:

Results of sensitivity analysis are shown in Table 5. Decreases in the slope of density dependent dispersal, and decreases in the dispersal matrix had no considerable effect on population means over 50 years. Likewise, the impacts of reducing initial abundances and the standard deviations of

the survival matrix also had a minimal impact on projected population sizes. Adjustments to the dispersal coefficients ‘a’ and ‘b’ showed small negative impacts. The impacts of reductions in carrying capacity on the 50-year population mean were noteworthy (i.e., 10% and 20%; Table 5). Reductions made to the survival matrix had a considerable impact, particularly after 20% survivability decrease. This simulation reduced the final populations size an additional 30% compared to the final population size projected from the baseline model. The most sensitive parameter was, by far, the variable ‘c’ of the dispersal distance function, showing an almost 80% decrease in the population trend over time. This highlights the extreme sensitivity of the simulated mink populations presented here to dispersal parameters in RAMAS.

The impacts of the three harvest regimes proposed on the population are shown in Figure 7. Results are virtually identical for projections based on the non-spatial (one population) and spatial (three population) model. Simulations suggest that harvesting reduced total abundance, but does not affect local extinction risk until a 40% juveniles and 20% adult harvest. At that level of harvest, RAMAS estimates a 1% local extinction risk for three population model, and a slightly higher 3% local extinction risk for the one population model.

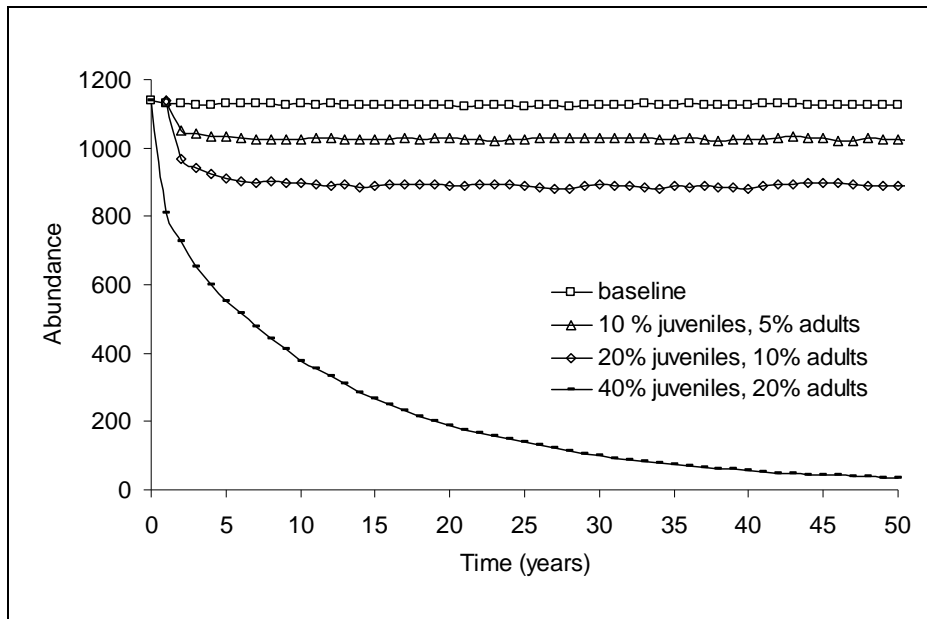
Table 5: Sensitivity analysis results showing changes in the mean of the population trajectory of mink over 50 years for 10% and 20% parameter alterations.

Parameter Adjusted	Sensitivity Analyses				
	Baseline Mean	10% decrease		20% decrease	
		Mean	% Change	Mean	% Change
Density dependent dispersal	1126.7	1126.9	0.0	1126.6	0.0
Dispersal coefficient 'a'	1126.7	1065.2	-5.5	1081.2	-4.0
Dispersal coefficient 'b'	1126.7	1075.4	-4.6	1095.7	-2.7
Dispersal coefficient 'c'	1126.7	527.2	-53.2	248.1	-78.0
Dispersal matrix	1126.7	1126.9	0.0	1126.6	0.0
Initial abundance	1126.7	1124.1	-0.2	1120.9	-0.5
Carrying capacity	1126.7	1017.0	-9.7	906.3	-19.6

Table 5: Sensitivity analysis results showing changes in the mean of the population trajectory of mink over 50 years for 10% and 20% parameter alterations.

Parameter Adjusted	Sensitivity Analyses				
	Baseline Mean	10% decrease		20% decrease	
		Mean	% Change	Mean	% Change
Survival matrix means	1126.7	1079.8	-4.2	803.7	-28.7
Survival matrix standard deviations	1126.7	1131.1	0.4	1133.9	0.6

Figure 7: Impacts of three harvest regimes on the population trajectory of mink (three-population models).



3.2.3 Conclusion

In the scenarios investigated, mink populations appear to be fairly resilient and are unlikely to experience catastrophic declines. At the optimal population density taken from the literature, and assuming a habitat suitability model that reasonably represents total habitat suitability on the landscape, populations appear to be resilient to moderate levels of harvest. Further, exploratory analyses have shown that even when 20% of the population is impacted by moderate levels of

PCB toxicity, and is then harvested of 20% juveniles and 10% adults per year, the mink populations still remains stable, although at a reduced mean population size over time (for more detail see Appendix B).

The sensitivity of mink to dispersal parameters highlighted the challenges of modelling a species with a high rate of dispersal in RAMAS. The reductions in population mean trends with increases in dispersal rate (e.g. reducing 'c' from the dispersal distance function in sensitivity analyses: see Table 5) in effect are creating sink patches that can not provide enough habitat for a large numbers of immigrants forced to leave the large population. Perhaps individuals of some species disperse despite a hostile matrix environment; however, it seems likely that individuals would return to natal patches if suitable habitat could not be found.

The three population baseline scenario (i.e. neighbourhood size of 50 cells) without dispersal produced comparable extinction risks to the 200 cell neighbourhood, and showed no meaningful difference in extinction risk in the scenarios investigated here. This suggests that an overall focus on preserving and enhancing wetland habitat would have a more pronounced positive impact on mink populations than the creation or maintenance of dispersal corridors. That being said however, metapopulations (i.e. multiple interacting populations, as represented with the 50 cell neighbourhood scenarios) do have theoretical and observed resistance to regional extinction that should not be completely disregarded. For example, if one population is harvested (or reduced in some other manner) sufficiently for local extinction, dispersal from a neighbouring population will have the potential to naturally re-establish it.

Results from sensitivity analyses show that even with 20% reductions in the estimates of most parameters, mink populations remain stable. However, sensitivity analyses have also shown that

if carrying capacity and survival matrix means have been overestimated, population means may be substantially lower than estimated. This would of course increase the risk of mink extirpation beyond that estimated here, due to negative impacts such as harvest or wetland loss. The possible combination of interacting scenarios is nearly limitless, and leaves room for further and important research opportunities.

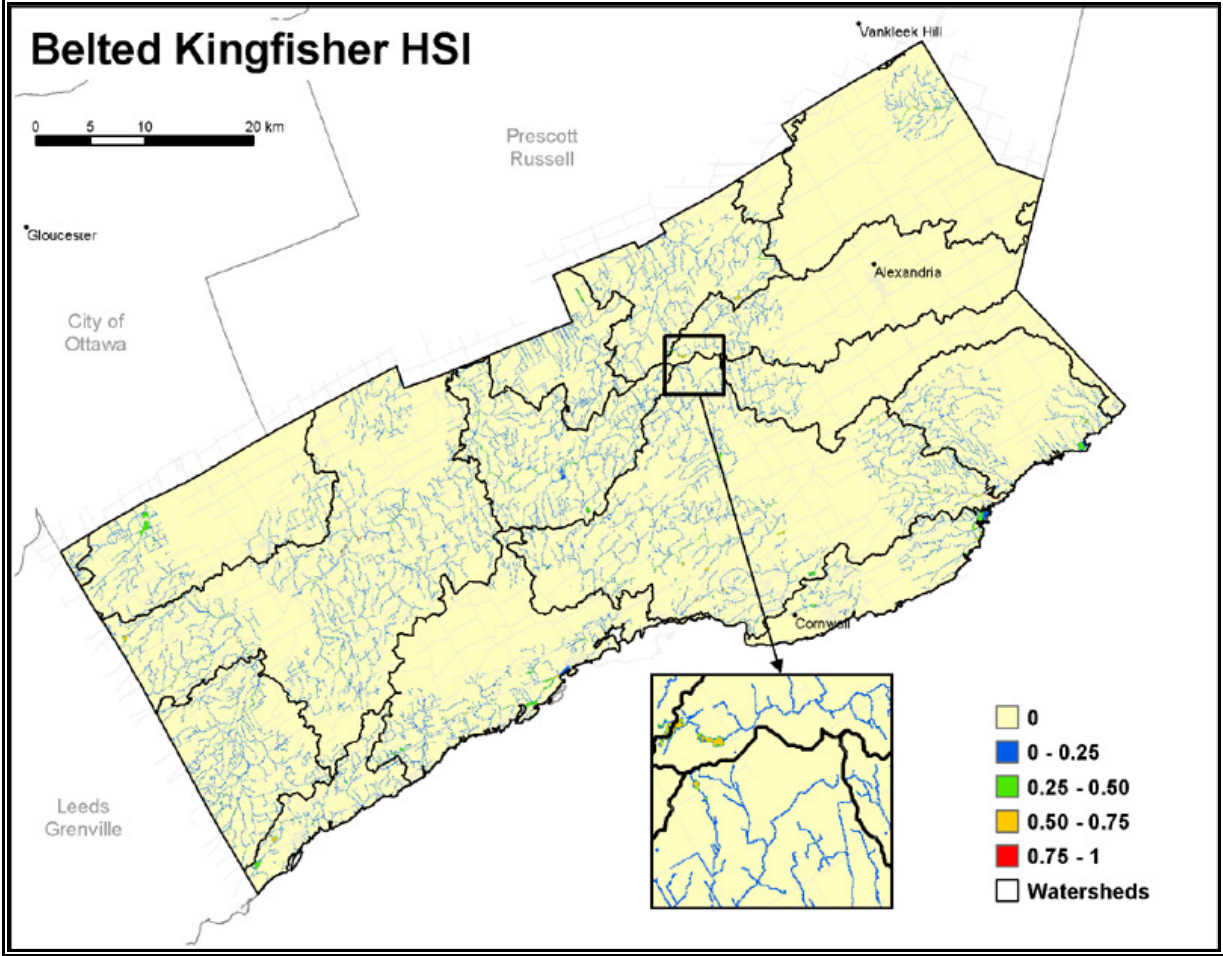
3.3 Belted Kingfisher Populations

3.3.1 Methods

Habitat-Population Relationships:

Using the Spatial Data command in RAMAS, a habitat suitability map of the belted kingfisher (Figure 8) identified patches of breeding habitat, their carrying capacities and initial abundances. Unfortunately there was insufficient information in the literature to make linkages of habitat suitability scores to relative fecundity of breeding birds within patches. The map was comprised of 0.032 x 0.032 km cells scored according to the habitat suitability index (0-1) described earlier in this report. A patch was defined as a cluster of cells within 8.0 km from another cell (edge to edge); only cells having a habitat suitability score > 0.50 were considered. The 8 km distance may reflect the maximum daily range of nesting adults (Prose, 1985). To calculate carrying capacities per patch, total habitat suitability was multiplied by the number of males to occupy a raster cell of breeding habitat representing ideal environmental conditions. It was assumed that under ideal conditions, approximately one pair of breeding adults and 6 juvenile birds occupied per 0.8 km of shoreline (or one male bird per 200 m) (Sayler and Lageler, 1949; Hamas, 1994). Initial abundances of patches were calculated as two-third the estimated carrying capacities. The local threshold at which a patch (i.e., subpopulation) was considered occupied was 10 male birds.

Figure 8: Belted kingfisher habitat identified from a Habitat Suitability Index (0-1; HSI) for eastern Ontario.



Stage Structure:

A stage-structured model with two life stages was developed: age-0 (juveniles), age-1 (adult) and age-2 (adult). A bird was considered an adult after surviving its first winter (White, 1953). Male birds were modelled only and a sex ratio of 1 male: 1 female was assumed (Kelly, 1998). It was also assumed that all breeding took place during a defined breeding season (birth pulse model; Caswell, 1989). The Leslie matrix was based on a ‘post-breeding’ census of juvenile and age-1 birds and the assumption that no mortality took place between the breeding and the census. It was

also assumed that individuals breed twice (Hamas, 1994), and reproduce only on their 1st and 2nd birthday, and died after reaching their 2nd birthday.

It was assumed that survival rates were very low for both juvenile and adult belted kingfishers based on observed low rates of return of juveniles and adults to natal sites (White, 1953; Davis, 1980). Adults appear to have higher survival rates than first-year birds, but researchers have noted it being difficult to distinguish between juvenile mortality and juvenile dispersal. Few if any survival estimates are available. For this study, estimates were based on those published for the American dipper, a similar species occurring in riparian areas in western North America (Table 6). In an Alberta study on American dippers, juvenile survival was estimated at 13% and annual adult survival was estimated at 56% (Ealey, 1977). Although variations in vital rates for birds are poorly known, it was assumed that the coefficient of variation (CV) of juvenile survival rates was 15%, which is a standard level of variation reported in previously published PVA studies on bird populations (Larson et al., 2004).

Fecundity rates were calculated by multiplying survival rate (0.13 for juvenile, 0.56 for adult) by average number of male offspring produced (Table 6). The number of new recruits per clutch was estimated at 5.48 (approximate range = 4.2-6.8), which was based on the average number of eggs in a clutch and reported survival rates of egg to newly fledged juveniles. Specifically, clutch sizes range 5–8 eggs, but most commonly 6 or 7 (Bent, 1940). Typically there is only one clutch per breeding season. In Minnesota, Hamas (1994) reported that hatching success was nearly 87% and that fledging success was 97% over 4 breeding seasons. It was assumed that environmental stochasticity was twice as great for fledging rates compared to survival rates (CV = 30%; Larson et al., 2004).

Table 6: Stage matrix comprised of fecundity and survival rates (± 1 SD) of belted kingfisher.

	YOY male	Adult male
YOY male	0.36 (± 0.11)	1.53 (± 0.46)
Adult male	0.13 (± 0.02)	0.56 (± 0.08)

Stochasticity:

Stochasticity was modelled by drawing values randomly from lognormal distributions described by the fecundity and survival values, and their associated standard deviations. The effects of stochasticity on fecundity, survival, and carry capacity were assumed to be correlated with a population, and the local extinction threshold. Belted kingfishers nests can be susceptible to flooding (Hamas, 1994); therefore we incorporated environmental fluctuations in the simulations. The impact of rare storm flooding events was simulated for all scenarios by including a 10% probability of a catastrophe that would reduce juvenile abundance by 25%.

Spatial Structure and Dispersal:

Adults may return to breed where previous nesting was successful (Hamas, 1994, Davis, 1980). White (1953) documented the return of three kingfishers to their general natal locality, but none of 46 fledglings banded in n. Minnesota returned. Low rates of return suggest high mortality or weak fidelity to breeding sites (Davis, 1980). In this study, it was assumed that the majority of juveniles disperse away from natal territories after fledging (95%; Hamas, 1994), and that juveniles were more likely to disperse as adults (also see Shriver and Gibbs, 2004). Adults were assumed to be 100% philopatric. Dispersal distances were calculated using an empirical model for calculating probabilities that animals disperse particular distances based on the mass of species of interest (Sutherland et al., 2000). Using a mass of 0.155 kg (Hamas, 1994), the model

predicted a maximum dispersal = 66 km and median dispersal = 12 km for the belted kingfisher.

Density Dependence:

A ceiling type of density-dependence (for adults only) was employed in models to avoid unnecessary complexity and to ensure that properties of territorial animals are maintained for projecting population sizes into the future (Akçakaya et al., 2004).

Simulations:

The models comprised either non-spatial structure (1 group) or spatial structure with multiple subpopulations linked by dispersal. The latter model type was considered the baseline model. At each time step, the number of juveniles and adults were projected, using a set of vital rates drawn from a random normal distribution with mean values taken from the stage matrix and standard deviations taken from the standard deviation matrix. A local extinction risk of <10% over 100 years was regarded as acceptably low for management purposes (Akçakaya et al., 2004; www.cosewic.gc.ca).

To aid management decisions and determine the influences of model parameters on metapopulation viability, we conducted sensitivity analyses on catastrophe scenarios, carrying capacity, initial abundance, stage matrix values, and average and maximum dispersal distances. To conduct a sensitivity analysis, we varied each model parameter by 10% while holding others constant. Specifically, the relative influences of model parameters on local extinction risk of the metapopulation were evaluated using confidence intervals and the Kolmogorov-Smirnov test statistic. If possible, further assessments of the sensitivity of parameters were explored by plotting risk curves together and by comparing the values at the point of maximum difference between the baseline curve and modified curve. Differences were assessed using a two-sample Kolmogorov-

Smirnov test statistic.

An additional scenario simulating habitat restoration was conducted given that preliminary analyses of HSI maps indicated that total habitat suitability in the study area may be low. Simulations were conducted using the baseline model described above but with 50% increases in total habitat suitability of shoreline cells within 3km of non-urban sandy soils.

3.3.2 Results

RAMAS identified three patches or sub-populations of breeding habitat in eastern Ontario that had a total habitat suitability of 1622 (mean habitat suitability = 0.59) (see Appendix A). Carrying capacity of the landscape was 260 male birds, and abundance at the start of simulations was 178 male birds.

Both non-spatial and spatial models were used to project abundance 100 years into the future. The difference between the two model structures was that the non-spatial structure assumed one population (i.e., one habitat patch) and no dispersal, versus three sub-populations and dispersing individuals between habitat patches for the spatially-structured model. Non-spatial simulations indicated that the regional population size was reduced 100% by year 100 of the simulation: terminal extinction risk = 1.0 (0.972-1.0, 95% CI). The median time to local extinction was reported as being 32.4 years. At year 20, the local extinction risk was 0.043 (0.08-0.1360, 95% CI), and the population size (+1SD) was 26.2 individuals (+62). At year 50, the local extinction risk was 0.861 (0.833-0.889, 95% CI), and the population size was 1.27 individuals (+5.87).

Projections of metapopulation sizes based on the spatial model (three subpopulations) indicated near certain probability of local extinction at year 100: terminal extinction risk = 1.0 (0.972-1.0, 95% CI). The median time to local extinction was 37.2 years (Figure 9). At year 20, the local

extinction risk was 0.052 (0.024, 0.08, 95% CI), and the metapopulation size (+1SD) was 26.9 individuals (+50.5). At year 50, the local extinction risk was 0.808, (0.78-0.836, 95% CI), and the metapopulation size was 1.68 individuals (+6.83).

Sensitivity analyses of the metapopulation model indicated that of all parameters tested, stage matrix parameters were the most sensitive parameters. They were also the only parameters to show significant increases in terminal extinction risk based on non-overlapping confidence intervals between risks projected from new and baseline models (Table 7) (Figure 9). Comparisons of decline (or local extinction) risk curves between the baseline curve and those from modified models failed to identify locations of maximum difference, based on Kolmogorov-Smirnov tests.

A watershed restoration simulation was also conducted based on a 50% increase in habitat suitability for shorelines within 3 km of non-urban sandy soils. Under this new scenario, RAMAS identified two patches or sub-populations of breeding habitat in eastern Ontario that had a total habitat suitability of 5589. Carrying capacity of the landscape was 894 male birds, and abundance at the start of simulations was 615 male birds. As with previous projections of metapopulation sizes of the belted kingfisher, results indicated near certain probability of local extinction at year 100: terminal extinction risk = 0.998 (0.97-1.0, 95% CI). However, median time to local extinction was slightly greater under the restoration scenario (45 years) compared to the baseline model (37.2 years).

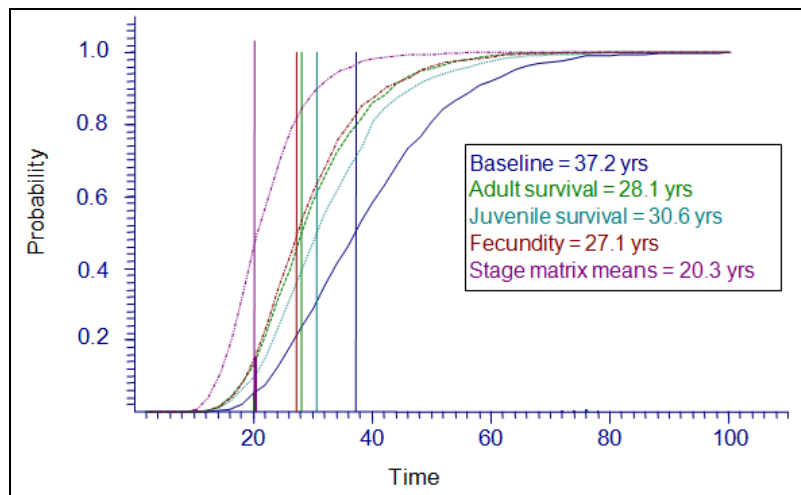
Table 7: Sensitivity analyses showing further decreases in belted kingfisher metapopulation projections at year 50, and new probabilities of local extinction risk (100% decline). Confidence intervals were based on the Kolmogorov-Smirnov test statistic.

	50-year population size	% decrease in population size	Extinction risk (100% decline)	95% CI
Baseline	1.7	--	0.808	0.78, 0.836
Model Parameter				
Stage matrix SD	1.68	0.0	0.804	0.776, 0.832
Average dispersal dist.	1.63	3.0	0.833	0.805, 0.861
Max. dispersal dist.	1.48	11.9	0.846	0.818, 0.874
Flooding	1.41	16.1	0.837	0.809, 0.865
K ^b	1.41	16.1	0.83	0.802, 0.858
Initial abundance	1.3	22.6	0.856	0.828, 0.884
Stage matrix means ^a	0.01	99.4	0.995	0.967, 1.0
Stage Matrix Parameter				
Adult survival ^a	0.44	73.8	0.931	0.903, 0.959
Juvenile survival ^a	0.22	86.9	0.957	0.929, 0.985
Adult fecundity ^a	0.2	88.1	0.964	0.936, 0.992

^{a)} non-overlapping 95% confidence interval with those of baseline model

^{b)} an indicator of total suitable habitat

Figure 9: Sensitivity analyses based on times to local extinction simulated from baseline models, and models with 10% changes in mean values of stage matrix parameter. Median times to local extinction are also identified.



3.3.3 Conclusion

The belted kingfisher in eastern Ontario is vulnerable of being extirpated within 100 years based on the simulations in this report. Both non-spatial and spatial models projected 100% declines over this time period. These results are consistent with recent observations (1982-1992) that suggest that sizes of continental populations are declining (see Hamas, 1994). Immediate management actions aimed at restoring habitat to pre-disturbance conditions may be required to maintain the persistence of the belted kingfisher metapopulation. Based on sensitivity analyses, a cost-effective conservation plan is one with an underlying goal of enhancing juvenile and adult survival rates, and fecundity rates. For example, restoration of degraded streams to healthy conditions (i.e., clear water with abundant forage fish) would enhance survival rates, particularly during migrations to winter habitat.

However, conservation planning that is constrained to eastern Ontario regions only may have marginal benefits for regional populations (or metapopulation) given that the kingfisher overwinters in the United States where mortality rates may be high (Hamas, 1994). Further, PVA projections indicated that the belted kingfishers remains vulnerable to local extirpation (i.e., probability of extinction of metapopulation > 10% over 100 year simulation) even with a 50% increase in the quantity and quality of suitable habitat. This suggests that to ensure the persistence of the metapopulation, watershed restoration must occur at a very large but potentially non-feasible scale to be successful.

It must also be noted that risks of local extinction and projections of population sizes from the PVAs may be inaccurate given that very little information could be obtained on actual vital rates of belted kingfisher. For the models in this report, survival rates were based on those of the

American dipper. Thus, an important step in the conservation of the belted kingfisher will be demographic research and the collection of information on annual survival rates and rates during breeding, migratory and non-breeding stages.

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5 APPENDICES

APPENDIX A: Spatial Data Results and Patch Information

Table A1: Spatial data results for northern leopard frog based on the HSI, 500 m neighborhoods and a 0.5 cell threshold for identification of patches.

Patch	Total HS	Average HS	K	Initial Abundance	R max.	Relative Fecundity	Relative Survival	X coord.	Y coord.
1	1	0.5	269	179	1	0.501	1	74.040001	4.18
2	1	0.64	342	228	1	0.638	1	78.160004	6.56
3	14	0.85	7759	5168	1	0.852	1	74.335999	7.798
4	1	0.56	298	199	1	0.557	1	70.919998	10.76
5	2	0.58	934	622	1	0.581	1	71.519997	11.493
6	1	0.77	415	277	1	0.775	1	66.879997	14.5
7	165	0.83	88554	58981	1	0.83	1	68.445	17.011
8	3	0.68	1452	967	1	0.677	1	61.595001	16.690001
9	7	0.51	3560	2371	1	0.511	1	87.117996	17.393999
10	3	0.53	1428	951	1	0.533	1	72.848	17.792
11	42	0.75	22420	14933	1	0.747	1	67.041	18.204
12	6	0.58	3395	2262	1	0.576	1	62.634998	18.209
13	33	0.83	17806	11860	1	0.83	1	61.410999	18.333
14	8	0.69	4041	2691	1	0.685	1	68.463997	18.360001
15	125	0.73	66898	44557	1	0.726	1	88.351997	19.101999
16	195	0.86	104615	69678	1	0.864	1	71.105003	19.370001
17	1	0.67	358	239	1	0.668	1	64.720001	19.4
18	24	0.86	12941	8619	1	0.862	1	64.250999	20.250999
19	193	0.57	103685	69059	1	0.572	1	66.346001	21.643999

Table A1: Spatial data results for northern leopard frog based on the HSI, 500 m neighborhoods and a 0.5 cell threshold for identification of patches.

Patch	Total HS	Average HS	K	Initial Abundance	R max.	Relative Fecundity	Relative Survival	X coord.	Y coord.
20	1	0.67	358	239	1	0.668	1	78.18	21.940001
21	22	0.52	11728	7811	1	0.521	1	68.432999	22.174
22	2	0.71	1142	760	1	0.71	1	68.892998	22.587
23	338	0.93	181005	120558	1	0.928	1	48.002998	23.482
24	452	0.69	242461	161490	1	0.687	1	62.534	23.531
25	215	0.5	114972	76576	1	0.5	1	75.528999	23.315001
26	1	0.62	333	222	1	0.622	1	66.540001	23.540001
27	135	0.67	72484	48277	1	0.669	1	70.515999	24.094999
28	1	0.67	710	473	1	0.663	1	73.230003	24.51
29	1	0.73	782	521	1	0.73	1	74.349998	24.6
30	3	0.59	1587	1057	1	0.592	1	67.875999	25.143999
31	1	0.75	401	267	1	0.748	1	64.900002	25.26
32	12	0.71	6513	4338	1	0.715	1	73.258003	25.514
33	572	0.91	306798	204341	1	0.909	1	72.502998	26.358999
34	2	0.76	1230	819	1	0.765	1	75.992996	26.127001
35	1	0.63	677	451	1	0.632	1	68.620003	26.23
36	35	0.62	18527	12340	1	0.617	1	71.099998	26.381001
37	8	0.7	4146	2761	1	0.703	1	70.004997	28.135
38	1	0.77	415	277	1	0.775	1	65.099998	28.219999
39	1	0.51	276	184	1	0.514	1	90.5	28.639999
40	2	0.72	1155	770	1	0.719	1	52.639999	29.58
41	16	0.58	8423	5610	1	0.582	1	53.407001	29.641001
42	1	0.58	311	207	1	0.581	1	74.760002	29.719999

Table A1: Spatial data results for northern leopard frog based on the HSI, 500 m neighborhoods and a 0.5 cell threshold for identification of patches.

Patch	Total HS	Average HS	K	Initial Abundance	R max.	Relative Fecundity	Relative Survival	X coord.	Y coord.
43	1	0.67	719	479	1	0.671	1	48.689999	30.540001
44	1	0.77	415	277	1	0.775	1	65.720001	30.879999
45	1	0.56	600	400	1	0.56	1	69.790001	30.93
46	4	0.63	2360	1572	1	0.629	1	30.277	31.006001
47	15	0.61	8171	5442	1	0.61	1	46.502998	31.056
48	26	0.85	14192	9453	1	0.854	1	52.918999	31.313999
49	12	0.63	6430	4283	1	0.631	1	54.859001	31.834
50	156	0.79	83638	55707	1	0.792	1	66.315002	31.709
51	66	0.85	35630	23731	1	0.852	1	68.445	31.871
52	1	0.77	415	277	1	0.775	1	60.240002	31.84
53	10	0.71	5296	3527	1	0.706	1	53.637001	32.469002
54	78	0.69	41842	27868	1	0.691	1	58.757999	32.618
55	7	0.7	3745	2495	1	0.699	1	29.128	32.563999
56	2	0.65	1038	691	1	0.645	1	94.800003	33.02
57	1	0.58	311	207	1	0.581	1	46.639999	33.16
58	1	0.62	331	220	1	0.617	1	75.059998	33.400002
59	20	0.66	10921	7274	1	0.657	1	76.254997	33.976002
60	1	0.63	672	447	1	0.627	1	71.489998	33.529999
61	1	0.72	384	256	1	0.716	1	29.52	33.540001
62	1	0.77	415	277	1	0.775	1	28.84	33.84
63	1	0.58	311	207	1	0.581	1	60.919998	33.919998
64	104	0.92	55594	37028	1	0.918	1	72.349998	34.266998
65	160	0.98	85551	56981	1	0.979	1	60.448002	34.466

Table A1: Spatial data results for northern leopard frog based on the HSI, 500 m neighborhoods and a 0.5 cell threshold for identification of patches.

Patch	Total HS	Average HS	K	Initial Abundance	R max.	Relative Fecundity	Relative Survival	X coord.	Y coord.
66	93	0.76	50024	33318	1	0.759	1	45.895	34.709
67	39	0.94	20642	13749	1	0.939	1	55.185001	34.714001
68	1	0.77	415	277	1	0.775	1	46.779999	34.799999
69	27	0.73	14499	9657	1	0.731	1	74.832001	35.131001
70	2	0.77	830	553	1	0.775	1	74.07	34.939999
71	4	0.63	2017	1343	1	0.627	1	48.362999	35.067001
72	125	0.52	67245	44788	1	0.516	1	64.287003	35.567001
73	1	0.67	358	239	1	0.668	1	76.080002	35.32
74	1	0.77	415	277	1	0.775	1	79.440002	35.380001
75	2	0.58	932	621	1	0.58	1	55.893002	35.459999
76	1	0.77	415	277	1	0.775	1	57.560001	35.720001
77	1	0.64	341	227	1	0.637	1	60.119999	35.720001
78	8	0.74	4371	2911	1	0.741	1	41.875999	35.896
79	1	0.6	646	430	1	0.602	1	62.849998	36
80	1	0.52	558	371	1	0.52	1	28.15	36.049999
81	3	0.6	1609	1071	1	0.6	1	29.052	36.34
82	1	0.58	309	206	1	0.577	1	46.139999	36.18
83	1	0.55	294	196	1	0.549	1	71.419998	36.240002
84	8	0.64	4116	2742	1	0.64	1	94.707001	36.556999
85	1	0.57	305	203	1	0.569	1	68.040001	36.580002
86	111	0.85	59573	39678	1	0.855	1	48.620998	36.900002
87	2	0.77	1246	830	1	0.775	1	52.487	36.973
88	1	0.63	339	226	1	0.633	1	63.66	37.380001

Table A1: Spatial data results for northern leopard frog based on the HSI, 500 m neighborhoods and a 0.5 cell threshold for identification of patches.

Patch	Total HS	Average HS	K	Initial Abundance	R max.	Relative Fecundity	Relative Survival	X coord.	Y coord.
89	14	0.57	7273	4844	1	0.565	1	90.837997	37.528
90	1	0.5	269	179	1	0.502	1	44.080002	37.540001
91	1	0.51	276	184	1	0.515	1	64.599998	37.619999
92	41	0.67	21861	14560	1	0.669	1	72.356003	37.793999
93	1	0.71	381	254	1	0.71	1	53.439999	37.66
94	135	0.5	72534	48311	1	0.503	1	60.967999	38.332001
95	1	0.53	283	189	1	0.529	1	55	38.060001
96	13	0.76	6893	4591	1	0.756	1	91.434998	38.888
97	2	0.65	1038	691	1	0.645	1	54.126999	39.193001
98	1	0.58	311	207	1	0.581	1	48.720001	39.639999
99	7	0.75	3633	2420	1	0.753	1	53.867001	39.740002
100	47	0.78	24932	16606	1	0.775	1	28.039	40.063999
101	66	0.95	35487	23636	1	0.946	1	47.240002	40.633999
102	1	0.58	623	415	1	0.581	1	53.34	40.57
103	2	0.77	830	553	1	0.775	1	73.110001	41.259998
104	2	0.65	1038	691	1	0.645	1	52.632999	41.727001
105	4	0.6	2260	1506	1	0.602	1	54.48	42.194
106	1	0.51	550	366	1	0.513	1	13.85	42.59
107	27	0.88	14610	9731	1	0.879	1	51.268002	42.974998
108	4	0.77	2076	1383	1	0.775	1	84.472	42.976002
109	1	0.57	618	412	1	0.576	1	60.279999	43.23
110	3	0.73	1557	1037	1	0.726	1	39.950001	43.349998
111	1	0.73	785	523	1	0.732	1	47.029999	43.34

Table A1: Spatial data results for northern leopard frog based on the HSI, 500 m neighborhoods and a 0.5 cell threshold for identification of patches.

Patch	Total HS	Average HS	K	Initial Abundance	R max.	Relative Fecundity	Relative Survival	X coord.	Y coord.
112	59	0.87	31600	21047	1	0.867	1	54.655998	43.546001
113	2	0.65	1038	691	1	0.645	1	81.313004	43.567001
114	5	0.68	2906	1936	1	0.678	1	63.16	43.827
115	2147	0.96	1150876	766535	1	0.956	1	7.963	44.993
116	9	0.89	4775	3180	1	0.891	1	85.480003	44.349998
117	1	0.77	415	277	1	0.775	1	15.04	44.639999
118	417	0.9	223765	149038	1	0.902	1	10.472	45.361
119	3	0.7	1510	1005	1	0.704	1	45.875	44.884998
120	9	0.83	4901	3264	1	0.831	1	85.189003	44.919998
121	1	0.58	311	207	1	0.581	1	40.560001	44.98
122	2	0.71	1146	763	1	0.712	1	70.720001	45.073002
123	76	0.97	40928	27260	1	0.967	1	84.453003	45.348
124	3	0.79	1687	1124	1	0.787	1	76.745003	45.220001
125	2	0.75	810	539	1	0.755	1	61.560001	45.529999
126	1	0.63	338	225	1	0.63	1	44.84	45.740002
127	5	0.66	2491	1659	1	0.664	1	39.959999	45.900002
128	1	0.58	311	207	1	0.581	1	16.98	46.16
129	1	0.57	306	204	1	0.571	1	59.52	46.220001
130	1	0.63	675	450	1	0.63	1	68.209999	46.380001
131	22	0.74	11937	7950	1	0.742	1	4.229	46.506001
132	40	0.68	21226	14137	1	0.683	1	83.019997	46.574001
133	33	0.87	17721	11803	1	0.87	1	46.213001	46.646999
134	1	0.56	300	200	1	0.56	1	39.099998	46.560001

Table A1: Spatial data results for northern leopard frog based on the HSI, 500 m neighborhoods and a 0.5 cell threshold for identification of patches.

Patch	Total HS	Average HS	K	Initial Abundance	R max.	Relative Fecundity	Relative Survival	X coord.	Y coord.
135	65	0.77	34608	23050	1	0.769	1	7.755	46.84
136	6	0.67	3240	2158	1	0.672	1	24.311001	46.988998
137	17	0.73	9030	6015	1	0.732	1	66.717003	47.169998
138	1	0.77	415	277	1	0.775	1	9.74	47.459999
139	1	0.58	311	207	1	0.581	1	39.900002	47.619999
140	1	0.77	415	277	1	0.775	1	65.260002	47.84
141	60	0.77	32094	21376	1	0.768	1	57.506001	48.275002
142	40	0.64	21497	14318	1	0.637	1	74.346001	48.470001
143	1	0.58	311	207	1	0.581	1	69.080002	48.380001
144	160	0.68	85543	56975	1	0.685	1	54.360001	49.056999
145	1	0.77	415	277	1	0.775	1	71.559998	48.880001
146	285	0.74	152869	101818	1	0.741	1	69.468002	49.154999
147	29	0.62	15378	10242	1	0.624	1	57.049	49.179001
148	5	0.52	2509	1671	1	0.52	1	47.653	49.296001
149	1	0.72	384	256	1	0.717	1	37.700001	49.360001
150	1	0.6	323	215	1	0.603	1	62.799999	49.48
151	2	0.77	830	553	1	0.775	1	8.04	49.57
152	1	0.68	367	244	1	0.685	1	67.360001	49.560001
153	2	0.67	1076	717	1	0.669	1	69.946999	49.907001
154	2	0.69	1102	734	1	0.686	1	71.133003	49.932999
155	1	0.68	727	484	1	0.678	1	68.669998	50.18
156	3	0.63	1339	892	1	0.624	1	3.87	50.334999
157	1	0.58	311	207	1	0.581	1	77.599998	50.759998

Table A1: Spatial data results for northern leopard frog based on the HSI, 500 m neighborhoods and a 0.5 cell threshold for identification of patches.

Patch	Total HS	Average HS	K	Initial Abundance	R max.	Relative Fecundity	Relative Survival	X coord.	Y coord.
158	10	0.6	5473	3645	1	0.601	1	61.563999	50.875999
159	51	0.72	27081	18037	1	0.722	1	68.320999	51.115002
160	2	0.76	816	543	1	0.761	1	76.160004	51.349998
161	1	0.62	333	222	1	0.621	1	36.299999	51.619999
162	1	0.52	276	184	1	0.515	1	3.84	51.66
163	1	0.68	362	241	1	0.675	1	46	52
164	4	0.68	2180	1452	1	0.678	1	60.587002	52.272999
165	1	0.58	311	207	1	0.581	1	62.34	52.18
166	1	0.77	415	277	1	0.775	1	74.419998	52.580002
167	6	0.71	3425	2281	1	0.71	1	72.624001	52.844002
168	1	0.77	415	277	1	0.775	1	37.099998	52.84
169	20	0.72	10483	6982	1	0.724	1	65.016998	52.965
170	2	0.71	1142	760	1	0.71	1	57.473	52.946999
171	1	0.77	415	277	1	0.775	1	35.5	53
172	7	0.82	3506	2335	1	0.818	1	59.591999	53.025002
173	2	0.75	811	540	1	0.756	1	58.330002	53.220001
174	1	0.68	727	484	1	0.678	1	31.629999	53.639999
175	2	0.77	1246	830	1	0.775	1	5.54	53.907001
176	3	0.73	1557	1037	1	0.726	1	50.005001	54.009998
177	314	0.62	168254	112065	1	0.624	1	5.931	54.987999
178	1	0.58	311	207	1	0.581	1	48.880001	54.599998
179	3	0.82	1753	1168	1	0.818	1	50.665001	54.895
180	14	0.7	7519	5008	1	0.701	1	47.280998	54.931999

Table A1: Spatial data results for northern leopard frog based on the HSI, 500 m neighborhoods and a 0.5 cell threshold for identification of patches.

Patch	Total HS	Average HS	K	Initial Abundance	R max.	Relative Fecundity	Relative Survival	X coord.	Y coord.
181	1	0.63	681	453	1	0.635	1	10.83	55.23
182	175	0.52	94043	62637	1	0.516	1	49.479	55.924999
183	71	0.8	38176	25427	1	0.8	1	34.375999	56.431
184	14	0.66	7469	4975	1	0.664	1	51.519001	55.742001
185	56	0.87	29962	19956	1	0.873	1	22.466	56.208
186	1	0.63	337	224	1	0.628	1	27.719999	57.18
187	15	0.78	7900	5261	1	0.776	1	22.288	57.327999
188	3	0.68	1453	968	1	0.678	1	34.255001	58.665001
189	1	0.58	311	207	1	0.581	1	29.040001	58.700001
190	34	0.83	18287	12180	1	0.832	1	46.292999	59.199001
191	7	0.5	3492	2326	1	0.501	1	38.749001	59.214001
192	8	0.65	4505	3000	1	0.646	1	35.075001	59.682999
193	1	0.67	358	239	1	0.668	1	33.360001	60.200001
194	30	0.91	16096	10721	1	0.91	1	43.203999	60.855999
195	3	0.73	1557	1037	1	0.726	1	27.465	61.400002
196	16	0.74	8719	5807	1	0.739	1	38.047001	61.449001
197	2	0.65	1044	695	1	0.649	1	33.599998	61.426998
198	4	0.84	2250	1499	1	0.84	1	43.736	61.571999
199	13	0.67	7212	4803	1	0.673	1	37.049999	62.723999
200	1	0.64	694	462	1	0.647	1	38.509998	63.279999
201	1	0.68	727	484	1	0.678	1	22.860001	63.43
202	14	0.63	7736	5152	1	0.628	1	35.150002	64.030998
203	1	0.58	311	207	1	0.581	1	38.060001	64.019997

Table A1: Spatial data results for northern leopard frog based on the HSI, 500 m neighborhoods and a 0.5 cell threshold for identification of patches.

Patch	Total HS	Average HS	K	Initial Abundance	R max.	Relative Fecundity	Relative Survival	X coord.	Y coord.
204	106	0.9	56609	37704	1	0.895	1	35.332001	65.834
205	34	0.77	18162	12097	1	0.77	1	31.839001	65.856003
206	2	0.76	815	543	1	0.761	1	30.809999	65.82
207	1	0.66	354	236	1	0.661	1	25.959999	65.839996
208	1	0.73	778	518	1	0.725	1	27.07	66.610001
209	14	0.62	7699	5128	1	0.625	1	34.080002	66.871002
210	5	0.72	2699	1797	1	0.719	1	15.034	70.628998
211	40	0.88	21667	14431	1	0.879	1	20.841999	73.138
212	19	0.58	9985	6650	1	0.582	1	19.409	73.272003
213	5	0.71	2672	1780	1	0.712	1	17.451	74.116997
214	112	0.57	59825	39846	1	0.569	1	16.934999	75.166
215	14	0.75	7266	4839	1	0.753	1	17.257999	76.032997

Table A2: Spatial data results for mink based on the HSI, 1.5 km and 6.5 km (*) neighborhoods, and a 0.4 cell threshold for identification of patches.

Patch	Total HS	Average HS	K	Initial Abundance	R max	Relative Fecundity	Relative Survival	X coord.	Y coord.
1	3881	0.62	16	16	1	1	1	27.41	39.145
2	17539	0.6	72	72	1	1	1	8.305	47.974
3	256347	0.65	1050	1050	1	1	1	52.48	42.885
1*	282045	0.64	1155	1155	1	1	1	48.76	43.222

**only 1 patch identified using 6.5 km threshold.*

Appendix A3: Spatial data results for belted kingfisher based on the HSI, 8 km neighborhoods and a 0.5 cell threshold for identification of patches.

Patch	Total HS	Average HS	K	Initial Abundance	R max	Relative Fecundity	Relative Survival	X coord.	Y coord.
1	72	0.56	12	8	1	1	1	79.634	6.582
2	1081	0.61	173	119	1	1	1	37.772	50.862
3	439	0.64	70	48	1	1	1	82.43	41.97

APPENDIX B: PCB-Mink Scenarios

Polychlorinated hydrocarbon toxicity:

It has now been well established in the literature that mink are sensitive to the presence of polychlorinated hydrocarbons (e.g. PCB's) in their diet (Bursian et al., 2006). Negative effects can range from jaw lesions that can potentially lead to a loss of teeth (Bursian et al., 2006), to decreased kit growth and survivability (Restum et al., 1998). Martin (2006) found that mink in certain parts of southern Ontario are consuming sufficient PCB's in their diet to negatively impact kit growth and survivability. Over the period of 1998 to 2003, sampled regions ranged from 0 to 37.5% of mink exceeding PCB concentrations necessary to limit kit growth, and 0 to 19% of mink exceeding PCB concentrations necessary to limit kit survivability. Although chlorinated hydrocarbon concentrations in mink have shown a general decrease in southern Ontario, western Lake Erie mink have shown an increase.

Restum et al. (1998) reported statistically significant impacts on kit survivability at 0.5 ppm of polychlorinated hydrocarbons in the diet. Although it is reasonable to assume that it occurs, impacts on adult survivability were not reported, so only changes to juvenile survivability were applied. Martin (2006) only reports that certain percentages of minks sampled contained more than 0.5 ppm polychlorinated hydrocarbons in their diet, so some mink may have had much more than that, and therefore exhibited greater impacts on kit survivability. Furthermore, impacts of reduced growth have not been quantified in terms of survivability, but negative impacts would be expected. Despite this lack of desirable detail in quantifying impacts of polychlorinated hydrocarbons toxicity, a number of scenarios were conducted. Restum et al. (1998) reported an average of 68% decreased kit survivability over three generations. In keeping with Martin's (2006) reported percentages of mink exceeding the 0.5 ppm threshold, three scenarios were

conducted with juvenile survival rates decreased by 5% of 68% (i.e. 5% of the population exceeds the threshold of 0.5 ppm), 10% of 68%, and 20% of 68%, respectively. For the sake of interest, a fourth scenario was conducted investigating the effect of a full 68% reduction in kit survivability.

Results:

The impacts of the 4 polychlorinated hydrocarbon toxicity scenarios are shown in Figure 1. Results were virtually identical for both models (one and three population models). Minimal impact on populations was observable until 100% of the population was subjected to a 68% decrease in survivability. At that point, RAMAS estimates a 98% extinction risk for the three-population model, and a 99% extinction risk for when only one large population is present.

Figure B1: Impacts of various polychlorinated hydrocarbon toxicity scenarios on the population trajectory for 3 subpopulations of mink. Scenarios represent the percent of each population experiencing a 68% decrease in kit survivability.

