Home range, population density, habitat preference, and survival of fishers (*Martes pennanti*) in eastern Ontario

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Abstract

By the 1940s, fishers (Mustelidae, *Martes pennanti)* were extirpated in Ontario south of the French and Mattawa Rivers, probably as a result of overharvesting and habitat loss. However, during the last several decades fishers have recolonized much of their former range in Ontario. This recolonization, combined with (for the most part) conservative harvest management, has led to increases in abundance. Perhaps inevitably, these increases have resulted in requests by fur harvesters to increase fisher quotas. The question then arises as to what the effect of the current quota system is on fisher populations in eastern Ontario. Unfortunately, very little is known about fisher demographics in eastern Ontario; as a result, the current management system is based almost exclusively on information and data on well-studied fisher populations from other regions, notably Algonquin Park. The extent to which these data – and the inferences regarding effective management therefrom - reflect fisher population characteristics in eastern Ontario is unknown.

To fill in important information gaps, I examined home range, population density, habitat preference, and survival of a fisher population in Leeds and Grenville County in eastern Ontario. Sixty-one fishers were fitted with radio-collars and tracked using ground and aerial telemetry from February 2003 until January 2005. Home ranges were consistently smaller than those reported in the literature, with some overlap of adjacent intrasexual home ranges. As such, population density was estimated to be relatively high in this area. Habitat selection was assessed at two spatial scales: fishers prefer coniferous forest over fields and deciduous forest at a coarse scale, but use habitat in proportion to its availability at a fine scale. Non-harvest mortality was high compared to reported rates in the literature. These results, in conjunction with ancillary harvest data, suggest that while current fisher population density may be relatively high in the study area, abundance is very likely declining. If one subscribed to the principle of conservative wildlife management, this implies that current harvest quotas should not be increased.

Résumé

Dans les années 1940, les pékans (Mustelidae, *Martes pennanti*) ont été éliminés au sud des rivières French et Mattawa (Ontario). Cela fut probablement le résultat de la trappe excessive et de la perte d'habitat. Durant les dernière décennies, les pékans ont recolonisé une bonne partie de leur aire de répartition originale dans le sud de l'Ontario. Cette recolonisation, combinée (pour la plus grande part) avec les mesures de conservation et de gestion de la trappe conservatives, a mené à une augmentation de l'abondance. Inévitablement, cette plus grande abondance a eu pour effet de faire multiplier les demandes d'augmentation de quotas pour la trappe du pékan. L'effet du système de quotas sur les populations de pékans dans l'est de l'Ontario est inconnu. Malheureusement, très peu d'information est disponible sur la démographie des pékans dans l'est de l'Ontario. Le présent système de gestion des populations est donc basé presque uniquement sur des données des populations mieux étudiées, soient celles d'Algonquin Park. L'on ne sait à quel point ces données et les suppositions qui en découlent concernant la gestion efficace des populations de pékans reflètent la réalité de l'est de l'Ontario.

Pour compléter l'information manquante, j'ai examiné les domaines vitaux, la densité de population, la préférence d'habitats et la survie chez les populations de pékans du compté de Leeds et Grenville, dans le sud-est de l'Ontario. Des colliers émetteurs ont été installés sur 61 pékans et ces derniers ont été suivis par télémétrie aérienne et au sol de février 2003 à janvier 2005. L'étendue des domaines vitaux était constamment plus petite que ce qui a été rapporté dans la littérature, avec un léger chevauchement de domaines vitaux juxtaposés appartenant à des individus de même sexe. Il a donc été estimé que la densité de population est relativement élevée dans cette région. Les pékans ont démontré une sélectivité de l’habitat à deux échelles spatiales. À l’échelle globale, il y avait préférence pour la forêt de conifères par rapport aux champs et à la forêt décidue. À une échelle plus fine, il y avait une utilisation proportionnelle des habitats selon leur disponibilité. La mortalité qui n’a pas été causée par la trappe était élevée comparativement aux valeurs rapportées dans la littérature. Ces résultats, en combinaison avec les données auxiliaires de la trappe, suggèrent que même si la densité de population est élevée dans la région étudiée, l’abondance est fort probablement en diminution. Si l’on applique les principes de conservation et de gestion de la faune, il est suggéré de ne pas augmenter les quotas de trappe du pékan.

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General Introduction

The value of the fisher

Fishers (*Martes pennanti)*, of the family Mustelidae, are mid-sized carnivores endemic to North America. As solitary predators in northern forest communities, fishers are highly regarded by some for their aesthetic value (Powell 1993), but as a nuisance by others, especially where densities are relatively high (Brander and Books 1973). Fishers are a major predator of porcupines (*Erethizon dorsatum*) where the 2 species co-exist, and there is evidence that fishers can limit porcupine populations (Earle and Kramm 1982). This is important, both ecologically and economically, in areas where high densities of porcupines have damaged trees (Cook and Hamilton 1957, Brander and Books 1973). Fishers have thick fur that is sought after by fur trappers, and their high pelt value makes them a lucrative resource for trappers (Powell 1993, Obbard et al. 1987). The species is particularly sensitive to over-trapping and habitat destruction because of its low density (Powell 1993, Douglas and Strickland 1987) and limited dispersal (Arthur et al. 1993). The distribution of fishers has fluctuated greatly in North America since human settlement and some western populations are in danger of extinction (Powell 1993, Thompson 2000).

Historical and current ranges of the fisher

The oldest known remnants of the fisher are from Virginia and are dated at 29,870 years bp (Anderson 1994). Historically, fishers likely inhabited most of the forested regions of Canada and the northeastern United States (Hagmeier 1956, Hall and Kelson 1959, Powell 1993, Gibilisco 1994, Graham and Graham 1994). By the 1930s and

1940s, the range and density of fishers had decreased in North America, likely due to over-exploitation and habitat loss (Rand 1944, Strickland and Douglas 1981, Powell 1993, Gibilisco 1994), particularly south of the Great Lakes region (Seton 1953, deVos 1964, Gibilisco 1994). In 1944, there were very few, if any, fishers south of the French and Mattawa Rivers in Ontario (Rand 1944, Hagmeier 1956). Since then, restricted harvesting has resulted in an increase in fisher density and recolonoziation of much of its former range (Hamilton and Cook 1958, deVos 1964, Powell 1993, Gibilisco 1994). Moreover, abandoned farmland has reverted to second growth forests, increasing the abundance of suitable habitat (Coulter 1960, Arthur et al. 1989, Powell 1993, Lancaster et al., unpublished data). By 1960 there were reports of increasing fisher populations in Maine (Coulter 1960), New York (Hamilton and Cook 1958), New Brunswick (Dilworth 1974), Nova Scotia (Dodds and Martell 1971) and Ontario (deVos 1964). Currently, fishers are distributed throughout most of Ontario (Gibilisco 1994, Thompson 2000).

A problem for fur managers in Ontario

In the 1980s, more fishers were harvested in North America than ever before (Obbard et al. 1987), and trappers in Ontario harvest more fisher pelts than in any other province or state (Strickland and Douglas 1981). In 2001, fur harvesters in eastern Ontario alone trapped 1,300 fishers (FUR Management Information System (FURMIS)). The critical question for fur managers in eastern Ontario (and indeed, elsewhere) is: given current habitat availability, what level of fisher harvest is sustainable? Unfortunately, answering this question is made more difficult by the fact that prior to the fisher decline of the 20th century, fisher trapping was not regulated (Strickland and Douglas 1981). Thus, there is no previous management regime for which to base current management on.

In recent years, researchers have proposed methods for monitoring fisher populations (Strickland 1994, Zielinski and Stauffer 1996, Zielinski et al. 1997) and the intensity of the harvest (Strickland and Douglas 1981). However, problems can arise when management plans are developed for one population and then applied to another. For example, fisher managers in Minnesota found that management guidelines developed for a sustainable fisher harvest in Ontario (Strickland and Douglas 1981) resulted in an over-harvest in Minnesota (Berg and Kuehn 1994). This underscores the importance of population-specific demographic information in the setting of quotas and trapping regulations.

Current management in eastern Ontario

In eastern Ontario, fishers are managed using trapper licensing, restricted seasons, registered traplines and a quota system (Novak 1987). Fisher quotas are allotted such that each trapper can harvest 1 fisher, and additional fishers may be trapped at one fisher per 1.62 km^2 (400 acres) of registered trapline or signed up land. Managers attempt to monitor changes in fisher abundance from year to year and adjust quotas accordingly. Currently, fisher managers use guidelines developed by Strickland and Douglas (1981) to assess the effects of the harvest on the fisher population. This method uses the ratio of juvenile to adult ($\geq$ 1.5 years old) female fishers in the harvested population as an index of the level of harvest. Strickland and Douglas (1981) found that a ratio of $>$ 4 juveniles per adult female corresponded to a growing population, while a ratio of $<$ 4 juveniles per adult female indicated a declining population. Managers in eastern Ontario estimate this ratio annually for local fisher populations from the canine tooth extracted from the skulls of trapped fishers that are voluntarily submitted by trappers.

The guidelines suggested by Strickland and Douglas (1981) were developed for fishers in the Algonquin region of Ontario, not for eastern Ontario. Fisher demographics will vary between populations, and along with habitat differences between the Algonquin region in the Boreal Shield ecozone, and eastern Ontario in the Mixedwood Plains ecozone, begs the question of whether current management practices are appropriate for eastern Ontario.

Objectives

The main objective of this thesis is to provide fur managers in eastern Ontario with demographic information on the local fisher population such that the species can be managed in a sustainable manner. Specifically, estimates of fisher population density, derived from home ranges, give an indication of the local population size prior to the harvest, from which I can estimate the proportion of the population that is harvested. Estimates of cause-specific mortality provide insight into the major causes of mortality in the trapped fisher population. Furthermore, estimates of survival rates indicate whether recruitment can sustain mortality rates. To measure these parameters, fishers were monitored using radio-telemetry in eastern Ontario between February 2003 and January 2005.

The first chapter of this thesis describes the methods used to collect radio-telemetry data, and also describes fisher home range size, intrasexual territoriality, and population density. Chapter 2 builds on Chapter 1 by using fisher home ranges to determine habitat preferences both within the study area and within the home range. In Chapter 3, triangulation error is estimated for both stationary and moving radio-collars; this information is used to assess the accuracy of future calculations (such as home range and habitat use) based on triangulation data. Finally, Chapter 4 uses radio-telemetry data over 2 years to estimate survivorship and cause-specific mortality of fishers in eastern Ontario.

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Chapter 1: Home range characteristics and population density of fishers in eastern Ontario

Introduction

The fisher (*Martes pennanti*) is a North American furbearer with a history of extirpation from over trapping and habitat loss (Powell 1993a). Population characteristics such as home range size, spacing among individuals, and population density give managers basic information on which to base future sustainable management regimes. Unfortunately, fishers are not easy to study due to their low population densities (Powell 1993a). Methods for assessing population demographics, such as capture-recapture, are difficult to employ with a relatively low density species because sample sizes tend to be small (Douglas and Strickland 1987, Powell and Zielinski 1994). Techniques using harvest records and catch per unit effort are useful indices of population change over time (Douglas and Strickland 1987), and estimates of sex and age ratios in the harvested population can be used to monitor the effects of the harvest on the population (Strickland and Douglas 1981). However, these methods do not provide estimates of home range size, spacing patterns, population density, habitat use, or survival rates – all of which can, at least in principle, be obtained from radio-telemetry studies (White and Garrott 1990, Millspaugh and Marzluff 2001).

This chapter will investigate home range size, spacing patterns, and population density of adult fishers in eastern Ontario from radio-telemetry data. This will be preceded by an introduction to home range estimators as this is the basis of all future calculations, and an introduction to the current knowledge of fisher spacing patterns.

Home range estimators

Home ranges are often measured from a series of radiolocations of a radio-tagged animal. There are several methods of estimating home range from location data, and each produces a home range of a different size and shape for a given individual. The following section will outline some of the available home range estimators and their advantages and disadvantages.

Burt (1943) defined an animal's home range as the area that it occupies while performing its normal routine. This definition has become more specific as methods for measuring home range size have advanced. The minimum convex polygon (MCP; Mohr 1947) is the most intuitive method of home range estimation. From a set of radiolocations, the most peripheral locations are connected to create a polygon such that there are no interior angles >180°. This method is commonly used to estimate home range size in radio-telemetry studies (Harris et al. 1990, Seaman et al. 1999) and as such, provides a useful tool for comparison between studies. However, the MCP does not provide an indication of the intensity of home range use. Moreover, the estimated home range size increases with the number of radiolocations and is highly dependent upon peripheral locations (Anderson 1982, Boulanger and White 1990, Burgman and Fox 2003).

More recently, home range has been defined in terms of the area in which an animal has some probability of being located during a specified period of time (Kernohan et al. 2001). This is the basis of *probabilistic* methods of home range estimation. The distribution of an animal's position in a plane has been termed the utilization distribution (UD) (Van Winkle 1975). It is a bivariate frequency distribution (Van Winkle 1975), with the x-y axes representing the animal's position in 2-dimensional space and the z-axis representing the frequency of occurrence in that space (Anderson 1982, Worton 1987). By defining the UD of an animal, one can then identify core areas of use within the home range, such as the area with a 50% occurrence probability (Anderson 1982).

Parametric home range estimators- Bivariate normal models

Parametric methods of home range estimation usually define the home range as a series of probabilistic ellipses about a center of activity. These ellipses represent the contours of a bivariate normal distribution for which a defined percentage of the volume is contained (Anderson 1982). The methods rely on the assumption that an animal uses its home range in a normal distribution about a mean (Harris et al. 1990). Jenrich and Turner (1969) and Koeppl et al. (1975) used bivariate normal distributions to describe home ranges, and Smith (1983) presented a method to test whether animal movements fit the bivariate normal distribution.

Disadvantages of the bivariate normal methods include the inability to accurately represent home ranges that do not fit the bivariate normal distribution, such as home ranges with uniformly distributed locations or that contain multiple centers of activity (Boulanger and White 1990, White and Garrott 1990). Probability ellipse estimators tend to overestimate home range size when the distribution of the data is not bivariate normal (Boulanger and White 1990). Samuel and Garton (1985) presented a home range estimator robust to extreme locations by applying more weight to the locations closer to the mean of the ellipse. Anderson (1982) and Boulanger and White (1990) tested the effectiveness of home range estimators to represent a variety of simulated home range shapes and concluded that non-parametric estimators, such as the Fourier transform and harmonic mean methods, respectively, should be used for home range estimation when the underlying distribution is not bivariate normal.

Non-parametric home range estimators

Non-parametric methods do not assume a specific underlying distribution for radiolocation data. The more widely used of the non-parametric methods are the harmonic mean (Dixon and Chapman 1980), Fourier transform (Anderson 1982) and kernel methods (Worton 1989).

Harmonic mean

Dixon and Chapman (1980) defined the center of activity of animal movements as the area within an animal's home range with the greatest amount of activity. They developed a method to estimate these centers of activity using the harmonic mean of observed animal locations, rather than the previously used arithmetic mean (Dixon and Chapman 1980). A grid is placed over the area, and the distance between a grid node and each radiolocation is calculated. Each grid node is then labeled with a value corresponding to the harmonic mean of the set of distances. For example, a grid node that is close to a cluster of locations will receive a relatively low value. Isopleths are then drawn, connecting grid nodes of similar values. These isopleths can be selected to represent areas that contain a specified proportion of the locations. This method also allows the calculation of multiple areas of activity (Dixon and Chapman 1980).

The harmonic mean method for home range estimation has been criticized on several fronts. Location distributions that are highly skewed or leptokurtic will give home ranges that include areas not used by the animal (Harris et al. 1990). Furthermore, the choice of grid size and grid placement influences the estimation of the home range, making it difficult to compare home ranges between studies when using this technique (Worton 1987, Harris et al. 1990, White and Garrott 1990). If an observation is close to a node of the grid, that observation will contribute disproportionately more to the harmonic mean, or if the location falls on the node, the value is undefined (Worton 1987, 1989, White and Garrott 1990). These problems notwithstanding, Boulanger and White's (1990) simulation experiment revealed that the harmonic mean was less biased than the MCP or probability ellipses when estimating home ranges of varying shapes.

Fourier transform method

Anderson (1982) proposed the Fourier transformation, a series of sines and cosines, to smooth the bivariate frequency distribution of animal locations. A plane perpendicular to the z-axis is used to slice through the distribution at the point at which 95% of the volume under the distribution is above the plane. The contour created by the distribution at this point represents the home range. The drawback to this method is that the distribution at the 95% contour is not precise. Small errors in the UD influence the 95% contour greatly (Anderson 1982, Boulanger and White 1990). Because of this, Anderson (1982) recommended using the 50% contour to obtain more accurate results. As there is no good *a priori* reason for using either the 50% or 95% home range, this suggestion appears warranted given the improved accuracy of the 50% home range estimate (White and Garrott 1990). Anderson (1982) found that the Fourier transform method at the 50% contour performed as well as the bivariate ellipse, even when the underlying distribution was bivariate normal.

Kernel estimators

The kernel method for estimating home ranges was proposed by Worton (1989). This method is similar to the Fourier transform method (Anderson 1982) in that it smooths the UD. For the kernel estimator, a unimodal, bivariate probability density function, called a kernel, is placed over each location. The overlap of kernels when locations are close together results in areas with higher densities. A grid is placed over the area and the average of the kernel densities at the grid nodes is calculated (Worton 1989, Seaman and Powell 1996). The size of the grid has little effect on the estimated UD; rather, it is the width of the kernel that has a large effect on the estimate (Worton 1989, Kernohan et al. 2001).

A user-defined smoothing parameter, or bandwidth, determines the width of the kernels. Smaller bandwidths result in narrower kernel densities, and the resulting distribution shows well-defined structures. Larger bandwidths produce wider kernels and overall distributions that show only the general shape of the distribution (Silverman 1986, Seaman and Powell 1996). The optimal bandwidth can be estimated by the reference method, which assumes that the distribution of the data is bivariate normal (Silverman 1986). It is an *ad hoc* method that utilizes the variance from the data. However, when the data are multimodal, the reference method for bandwidth estimation tends to oversmooth the data (Silverman 1986, Worton 1995, Seaman and Powell 1996).

Alternatively, the method of least squares cross validation (LSCV) can be used to calculate the optimal bandwidth and is described in detail by Silverman (1986). This method finds the value of the bandwidth that minimizes the difference between the estimated and true density functions. However, the true density function is unknown in practice, so an approximation to the true density function is used (Silverman 1986). The LSCV method of estimating the optimal bandwidth appears less biased than the reference bandwidth method (Worton 1995, Seaman and Powell 1996, Seaman et al. 1999, Gitzen and Millspaugh 2003). Gitzen and Millspaugh (2003) compared variations of the LSCV method in different home range software and found little difference between the various LSCV methods; the variations used most commonly in home range software performed satisfactorily in terms of bias of the kernel estimate.

There are 2 types of kernel estimators: fixed and adaptive. Fixed kernel estimators have a constant bandwidth for each kernel, whereas the adaptive kernel estimators have variable bandwidths, such that areas with a lower concentration of points have wider bandwidths. Thus, the adaptive kernel method smooths more in the tails of the density distribution and less near the centers of activity (Silverman 1986, Worton 1989). Studies using simulated data to compare home range estimators have found that the fixed kernel method provides more accurate home ranges than the adaptive kernel method (Worton 1995, Seaman and Powell 1996, Seaman et al. 1999, Getz and Wilmers 2004).

Kernel estimates of home range perform well compared to other methods. Worton (1995) extended the simulation study of Boulanger and White (1990) to include kernel estimators in the comparison of home range estimators. He found the kernel estimator to be less biased than the harmonic mean, which was favoured by Boulanger and White (1990). Critics of the kernel estimator have shown that as sample size increases, the accuracy of the home range estimate does not improve (Getz and Wilmers 2004). Additionally, kernel estimators perform poorly when the data are clumped (Getz and Wilmers 2004).

α-hull and k nearest neighbor convex hull

Burgman and Fox (2003) used the α-hull method to construct home ranges. This method uses the Delaunay triangulation to connect locations with lines such that no lines intersect. Any line longer than a multiple (α) of the average line length is eliminated. The home range is thus the sum of the area of the triangles created by the remaining lines. They found that α = 3 created home range estimates that were the least biased.

Getz and Wilmers (2004) compared the kernel and α-hull estimators to the *k* nearest neighbor convex hull (k-NNCH) method. The *k*-NNCH method connects points to their (*k*-1) nearest neighbors (those points that are closest in proximity). The convex hulls created by these connections are summed to create the home range area. This method has the advantage of allowing the measurement of areas of high activity by ordering the hulls by size and adding them to the estimate from smallest to largest, until x% of points are included in the estimate (Getz and Wilmers 2004). The method of determining the value of *k* that produces the best estimate is still uncertain. Getz and Wilmers (2004) showed that the *k*-NNCH method produced more accurate home ranges than both the kernel and α-hull methods when the data was aggregated. Although the α-hull and *k*-NNCH methods are promising, they await thorough examination with simulation, especially when the choice of important values such as α for the α-hull method and *k* for the *k*-NNCH method is still unclear.

Kernohan et al. (2001) evaluated 12 commonly used home range estimators based on criteria such as required sample size, robustness to autocorrelated data, ability to identify intensity of use and sensitivity to outliers. Kernel home range estimators ranked the highest using these criteria; as such, kernel estimators will be used in this study as estimates of home range size. Minimum convex polygons will also be used for comparability with other studies (Harris et al. 1990).

Sample size and autocorrelation

Home ranges are based on a sample of locations for individual animals: in general, the larger the number of radiolocations, the more accurate the home range estimate (Boulanger and White 1990, Worton 1995, Seaman and Powell 1996, Seaman et al. 1999, Gitzen and Millspaugh 2003). One way to estimate whether a sufficient number of locations has been collected is to plot the estimated home range area against the number of locations (Harris et al. 1990, Otis and White 1999). The home range has been adequately sampled when the size of the home range does not increase as more locations are added (Harris et al. 1990). This exercise is particularly important when using MCPs because their size tends to increase with the number of radiolocations (Anderson 1982, Boulanger and White 1990, Burgman and Fox 2003). The recommended number of locations necessary for accurate MCP home range estimates varies from 23 up to 200 (see Kernohan et al. 2001 for a review). Seaman et al. (1999) found that ≥ 50 observations were required for the kernel estimator to perform optimally (i.e. to be less biased with a better surface fit).

Confounded with sample size is sampling interval. Since the length of a radio-telemetry study is usually predetermined, increasing the number of locations per individual will decrease the time elapsed between locations. Locations taken too closely together in time can be spatially autocorrelated, meaning that the present location of an animal is influenced by its previous location (Dunn and Gipson 1977). This can influence the utilization distribution such that areas that appear to be used greatly are, in fact, an artifact of positive spatial autocorrelation due to locations having been collected too closely together in time (De Solla et al. 1999). Swihart and Slade (1985) found that autocorrelated observations resulted in an underestimation of home range size.

De Solla et al. (1999) showed that kernel home range estimators are more accurate and precise with larger sample sizes, despite the associated increase in autocorrelation. In fact, De Solla et al. (1999) suggested that perceived autocorrelation may not be due to short sampling periods, but rather to an animal's inherent use of its home range, such as when an animal periodically returns to a particular section of its home range. However, if autocorrelation is to be disregarded, radiolocations must be separated by relatively constant time periods since bursts of locations close in time will affect the utilization distribution (De Solla et al. 1999, Otis and White 1999). If radiolocations are taken randomly or systematically, autocorrelation of locations can be disregarded (White and Garrott 1990, De Solla et al. 1999, Otis and White 1999).

Fisher spacing patterns and population density

Fishers typically exhibit intrasexual territoriality (Arthur et al. 1989, Powell 1993a, 1994, Garant and Crête 1997, Fuller et al. 2001), where exclusionary home ranges are maintained relative to members of the same sex. Home ranges of males will overlap those of females, with large male home ranges overlapping several females' (Powell 1994). Powell (1993a, 1993b, 1994) proposed that intrasexual territoriality in fishers reduces competition for patchily distributed food while allowing males access to females. Assuming that fishers exhibit intrasexual territoriality, the density of fisher populations can be estimated by territory mapping (Arthur et al. 1989, Garant and Crête 1997, Fuller et al. 2001), such that unoccupied areas as large as the mean territory size with suitable habitat are assumed to support an uncollared fisher of the same sex (Fuller et al. 2001).

Methods

Study area

The area used as the boundary for live trapping fishers encompassed the townships of Edwardsburgh, Augusta, and North Grenville (approximately 45°50' N, 75°30' W) in Leeds and Grenville County, Ontario, Canada; an area of 975 km^2 (Figure 1.1). For population density analysis, the study area was delineated by a 100% MCP around the locations of all of the fishers used in the density estimate; an area of 894 km^2 (Figure 1.2). For the purposes of habitat analysis, the study area was defined as the area within the 100% MCP of all fisher locations used in the habitat analysis (see chapter 2); an area of 576 km^2 (Figure 1.2).

In 2001, 96,606 people lived in Leeds and Grenville County; 58,454 of those lived in rural areas (Statistics Canada 2001). The total area of Leeds and Grenville County is 3,390 km^2, thus human population density in rural areas is roughly 17 people/ km^2. In 2000, there was just over 1,800 km of roads in eastern Ontario (4,600 km^2 area; excluding roads in towns or cities) (Lancaster et al. unpublished data)

The elevation of the study area ranged from 80-130 m above sea level. The landscape was approximately 40% agriculture, 40% forest and 20% swamp (Landsat TM data; see Chapter 2). The forested region was composed mainly of cedar (*Thuja occidentalis*), larch (*Larix laricina*), hard maple (*Acer saccharum*), and soft maple (*Acer rubrum*) (Forest Resource Inventory (FRI) data; see chapter 2).

Temperatures in the study area during 2003 and 2004 ranged from a January average of -14.5°C to an August average of 20.7°C. The mean annual temperature for both 2003 and 2004 was 6.0°C. In 2003 and 2004, Kemptville received a total of 851 and 772 mm of rain and 194 and 168 cm of snow, respectively. These values are comparable to normal for this area: between 1971 and 2000, the mean annual temperature was 5.9°C, mean annual rainfall was 763 mm, and mean annual snowfall was 199 cm (Environment Canada, 2005).

Trapping procedure

Tomahawk live traps (Tomahawk Live Trap Co., Tomahawk, WI; model 106, 66 x 23 x 23 cm or model 108, 81 x 25 x 30 cm) were set between January 15 and March 2, 2003 in the study area in Grenville County. Traps were borrowed from the Rabies Research and Development Unit of the Ontario Ministry of Natural Resources (OMNR).

Ten local fur trappers set a total of 170 live traps throughout the townships of Edwardsburgh, Augusta, and North Grenville during this period. The locations of the set traps were at the discretion of the fur trapper and, thus, were not set randomly or systematically. The fur trappers moved traps repeatedly to different locations. Not all 170 traps were open for the entire 47-day period, thus catch per trap night cannot be calculated. Baits such as sardines, beaver, and muskrat were supplied by the trappers and used based on the availability of these baits to the trappers. The trappers checked each trap every 24 hours and were paid $50 for each live fisher trapped (see Appendix 1 for successful trap locations). Fifty fishers were caught during this period.

Additionally, the Rabies Research and Development Unit trapped fishers for this study between June 3 and mid July, 2003, during their trap-vaccinate-release program in parts of Grenville County within the study area. Seven fishers were trapped and outfitted with radio-collars during this period. The Rabies Research and Development Unit also trapped 4 fishers for this study between October 2 and October 29, 2003, during their rabies vaccination program. Areas trapped and the intensity of the trapping depended upon the incidence of rabies in the area. Fishers trapped during these periods were incidental catches as trappers were primarily targeting raccoons (*Procyon lotor*) and skunks (*Mephitis mephitis*).

Radio-collaring protocol

Drug administration

Fishers were weighed in a live trap using a linear spring scale to the nearest 1/4 of a kilogram. A 10:1 mixture of Ketamine : Xylazine was administered at 20 mg/kg using a sterile, disposable syringe (3 cc/ml 22G x 3.81 cm, latex free) as per recommendation by the OMNR Wildlife Animal Care Committee. To inject the drug, fishers were first pinned to the bottom of the trap (Tomahawk 106) using a smaller trap (Tomahawk 104.5). This prevented the fisher from moving while the drug was administered intramuscularly through the mesh of the trap. The drug usually took effect within 5 minutes. When the fisher showed signs of sedation, such as the inability to hold its head up and non-responsiveness to touch, the fisher was removed from the trap.

Tooth extraction, blood/hair sampling, ear tag, and collar application

The upper first premolar was extracted from live-trapped fishers for aging using a dental elevator to cut the gum away from the tooth, and pliers to remove it (Strickland et al. 1982). A small sample of blood was taken from the tooth socket and applied to blotting paper. An antibiotic ointment (Hibitane) was applied to the tooth socket to prevent infection. One ear tag (1005-3 National Band and Tag Co., Newport, KY) was put in each ear. Hair, including the root, was pulled from the base of the tail. The hair and blood samples were used in a different study on fishers (Carr et al. in prep.). Radio-collars (see below) were fitted as loose as possible, but not so loose that fisher could slide it over its head.

Release

Xylazine was reversed with 0.1 mg/kg Yohimbine. The trap containing the fisher was covered as the fisher recovered from the drugs (~2 hrs). Once fully conscious, the fisher was transported and released at the location where it was trapped.

Radio-collar specifications

Radio-collars (SMRC-3, Lotek Wireless Inc., Newmarket, ON) were in the 155 MHz frequency range with a 28 cm adjustable leather belt and a protruding whip antenna. The transmitters had a pulse rate of 35 bpm and were equipped with a mortality sensor at 70 bpm. The minimum battery life was 18 months. The radio-collars weighed 45-50 g each, and, thus, were less than 5% of the body weight of an adult fisher. Three of the radio-collars were made by Holohil (MI-2, Holohil, Carp, ON) and were also in the 155 MHz frequency range, with a pulse rate of 35 bpm and a mortality sensor. These radio-collars weighed 37 g and had an enclosed whip antenna.

Collar removal

Tomahawk live-traps were set in the home ranges of radio-collared fishers in July 2004 in an attempt to capture fishers to remove the radio-collars. In 1,240 trap nights, 1 radio-collared fisher was captured (155.820).

Aging

Sagittal crest

Fishers were aged as juveniles (<1 year old) or adults (≥1 year old) by feeling the sagittal crest of the sedated animal. The sagittal crest is enlarged in adult male fishers compared to juveniles (Kuehn and Berg 1981, Douglas and Strickland 1987). In female fishers, a relatively large sagittal crest is indicative of an adult, but the absence of a discernable sagittal crest does not distinguish an adult female from a juvenile (Douglas and Strickland 1987).

Tooth analysis

The upper first premolar was used to age fishers as juvenile (<1 year old) or adult (≥1 year old) by OMNR personnel using a combination of cementum annuli counts and pulp cavity size (Poole et al. 1994). When possible, the exact age was determined.

Triangulation and data collection

Equipment

Signals were located approximately from a vehicle with an omni direction whip antenna. Bearings were taken using a hand-held, 2-element, directional H-antenna. Signals were pinpointed from the air using 2 wing mounted H-antennas. Signals were received with one of 3 receivers: the Communications Specialists R1000 (Communication Specialists Inc., Orange, CA), the Lotek Suretrack STR_1000-W1 and the Lotek SRX_400 (Lotek Wireless Inc., Newmarket, ON). Bearings for triangulations were taken with a compass using a sighting mirror at the manufacturer's set declination of 0. Magnetic declination was accounted for as bearings were entered into Location of a Signal (LOAS; Ecological Software Solutions 3.0.2) computer software. The appropriate magnetic declination correction factor was calculated for the study area using web-based software (Natural Resources Canada) as 14°W in 2003 and 2004. Universal Transverse Mercator (UTM) coordinates were recorded with the Garmin GPS12 (Garmin International, Inc.) in NAD 83, or the Magellan GPS 12 (Magellan Systems Corporation), accurate to within 15 m or better, in NAD 27. Locations of the moving collar in the moving error tests were recorded using the Garmin GPSmap 76s in NAD 83.

Coordinates in NAD 27 were later converted to NAD 83 by adding 222 m to the NAD 27 northing and 31 m to the NAD 27 easting (conversion values obtained from Natural Resources Canada topographic maps).

Fisher locations

Radio-collared fishers were located using 4 methods from February 2003 until July 2004:

Triangulation

The majority of the locations were determined by triangulation from the ground. The observer used a hand-held 2-element directional H-antenna to determine the direction in which the signal was coming from. The direction was further pinpointed by reducing the gain of the receiver until the signal was barely audible. The signal could often be heard as an arc (usually about 100 degrees) rather than a point. The bearing was taken at the midpoint of this arc, either by recording the direction of the loudest signal or by recording the signal null on either side of the arc and calculating the midpoint mathematically (Springer 1979). The observer recorded the bearing, the coordinates of the location from which the bearing was taken, and the time. Observers took as many accurate bearings as possible per individual fisher in a 20-minute period (usually between 3-7 bearings) in an attempt to minimize the error if the fisher was moving. However, if the observer felt that after 20 minutes the triangulation was not adequate, more bearings were taken. The number of bearings was usually a function of the ease with which the bearings were taken: interference from hydro wires, background noise from vehicles, and lack of roads from which to take bearings increased the time between bearings. The observer attempted to take bearings such that the angle between the bearings was approximately 90-100 degrees (Zimmerman and Powell 1995).

Walking up to the animal

Attempts were made at approaching female fishers while they were attending natal dens during May and June 2004 by following the radio-signal. The location of the female was recorded after visual contact was made or the observer felt confident that the female was in a particular location even if she was not seen.

Re-trapping

Between April and November 2003, the Rabies Research and Development Unit set live traps in parts of the study area targeting raccoons and skunks. Occasionally, radio-collared fishers were trapped incidentally. When this occurred, the trapper recorded the identity of the tagged fisher and the coordinates of the trap before releasing the fisher.

Aerial telemetry

Fishers were located from the air from a Cessna 172. In July and October 2003, and May and June 2004, transects were flown across the study area and extended outside of the study area (ranging 20-40 km on the western, northern and eastern edges; the southern edge boardered the United States; see Appendix 2). Transects were flown at 750-950 m, and at 1,500 m along the perimeter of the transect grid to increase the coverage by the receiving system along the edge of the grid. Transects were 10 km wide since the receiving system picked up signals at least 6 km from either side of the airplane. These flights were intended to locate fishers that could not be located from the ground. Any fisher not found on these flights was considered lost to the study.

In January, March, and May 2004, fishers were located by flying towards the last known location for each fisher at 950-1,500 m. If the signal was not received when in the proximity of the last known location, the observer flew in an outward spiral in an attempt to locate the fisher.

To pinpoint the signal, the observer flew directly at the signal until the strength of the signal peaked. A control box was used to listen to the signal coming from each wing antenna separately. With the receiver gain low, the observer was able to circle the signal by making sure that the signal was always the strongest on the inside wing of the aircraft. When the observer felt confident that the fisher was within the circle, the coordinates were recorded with a GPS. Location error in this method varies with the radius of the circle flown around the animal since the observer is unlikely to be directly above the signal. The average location error for aerial telemetry was 729 m (± 422 SD) (see Chapter 3).

Volunteers

Seventy-eight percent of the triangulations were performed by myself, the primary observer, while the remainder of the triangulation data was obtained by 9 different individuals. These individuals spent at least 1 day with the primary observer learning the triangulation technique before collecting data alone. One observer with experience in aerial telemetry (Greg Borne) collected all of the data from the airplane.

Sampling interval and autocorrelation

Fishers were usually located at least once per week by triangulation, walking up to the animal, re-trapping, or by aerial telemetry. Exceptions occurred when the signal could not be located from the ground and the interval between the last location and the next flight to locate the lost signal was greater than one week. Time intervals between locations for individual fishers were always greater than 16 hours to reduce spatial autocorrelation, as recommended for fishers by Arthur et al. (1989). The sampling intervals were determined to maximize efficiency in data collection while making sure that individuals were not always sampled at the same time of the day for consecutive days.

Fishers are primarily crepuscular (Arthur and Krohn 1991) but also active during the day and night (Powell 1979, Powell 1993a). When animals are active at night, diurnal sampling alone will fail to identify areas used by the animals at night, resulting in an underestimation of home range size and habitat use if, in fact, space use is different at night (Arthur et al. 1989, Beyer and Haufler 1994). A mean of 9.7% (SD = 6.3, range 0 - 23.1%, n = 40 fishers) of the locations for fishers were taken at night. Night locations were included in the home range estimates, although not enough night data were collected to assess differences in nocturnal compared to diurnal behaviour (movements and/or habitat use).

Analysis

Point location estimation

Bearings were analysed with LOAS software. Point locations were calculated from triangulations using the maximum likelihood estimate (Lenth 1981) after eliminating potentially erroneous bearings. Triangulation error was estimated using the location error method (Zimmerman and Powell 1995) on a set of test data (n = 68). The average location error for stationary transmitters was 251 m (± 242 SD) and the 95% confidence distance was 600 m (see Chapter 3).

Number of locations

To determine the minimum number of locations necessary to adequately measure the 100% MCP home ranges of fishers, home range size was plotted against the number of observations used to make up the home range (Harris et al. 1990, Otis and White 1999). Five fishers of each sex with ≥ 30 locations between June 2003 and February 2004 were selected for this analysis. Only locations during this interval were used so as not to include locations obtained during the fisher mating season, when males tend to leave their home ranges (Leonard 1986, Arthur et al. 1989, Arthur and Krohn 1991, Powell 1993a) and females tend to spend a lot of time in natal dens (Paragi et al. 1994). The observations were selected randomly without replacement from the set of locations using a random numbers generator (Microsoft Excel). One hundred percent MCP home ranges were calculated for each animal, beginning with 10 randomly selected locations and adding 2 randomly selected locations at a time until all locations were used. This was repeated 3 times for each animal, giving a total of 3 estimated 100% MCPs per animal.

Home range estimation

Home ranges were estimated using 3 methods: 100% MCPs, 95% MCPs and fixed kernels with the bandwidth estimated using the LSCV method. The 100% MCP and 95% kernel estimates were calculated using the Animal Movement Analysis ArcView extension (Hooge and Eichenlaub 1997) in ArcView GIS 3.2 (Environmental Systems Research Institute Inc.). The algorithm for the fixed kernel estimate is based on Worton (1989) and uses a bivariate normal density kernel. The LSCV algorithm is based on Silverman (1986) (Hooge and Eichenlaub 1997). The 95% MCPs were calculated using the Home Range Extension (Rodgers and Carr 2001) in ArcView GIS 3.2. The outlying locations were identified by calculating the arithmetic mean of the locations and determining which location was furthest from the mean. This location was removed and the mean recalculated. This procedure was repeated until 5% of the locations were excluded, leaving polygons containing 95% of the locations (Kenward 1987).

Home ranges were calculated for the 26 fishers (10 males and 16 females) for which there were ≥ 25 locations during one or both of two time intervals: annually (March 2003- February 2004, or June 2003 – May 2004 for those fishers radio-collared in June 2003) and during the non-breeding period (June 2003 – February 2004). A repeated measures (RM) ANOVA, with method (100% MCP, 95% MCP or 95% kernel) as the repeated factor, was used to compare home range size between method and sex. Separate tests were run for the annual and non-breeding periods since the samples were not independent; fishers that had an annual home range always had a non-breeding season home range, although the converse was not always true if the fisher died before the annual home range measure was complete. Home range size was log transformed so that the residuals of the RM ANOVAs were normal and homoscedastic. The RM ANOVA assumption of sphericity was assessed by comparing the within subject p-values to the Huynh-Feldt corrected p-values; equivalence indicates that the sphericity assumption has been met (Systat 1996). Significant relationships were assessed with Bonferroni-corrected paired t-tests (Maxwell 1980, Hochberg and Tamhane 1987), calculated separately for each sex when the interaction between sex and method was significant ($p < 0.05$), and pooled across sex when the interaction term was not significant.

Spacing patterns: Static interactions

Intrasexual percent home range overlap for each individual, *i*, was defined as the percentage of *i*'s home range used by at least 1 other collared fisher of the same sex. Thus, the average intrasexual percent home range overlap for *n* individuals (not including individuals with no overlap- see discussion) was calculated as (adapted from Poole 1995):

$$\frac{\sum_{i=1}^{n}\left(\frac{\text{area of overlap of } i\text{'s home range}}{\text{area of home range } i}\right)}{n} * 100 .$$

The amount of overlap was calculated using 95% MCPs and 95% kernels as estimates of home range, and using 50% MCPs (Poole 1995) and 50% kernels as estimates of core area, during the period of time when both fishers were monitored. The breeding period (February - April) was omitted for male fishers due to unstable home ranges during this time (Leonard 1986, Arthur et al. 1989, Arthur and Krohn 1991, Powell 1993a) An RM ANVOA, with method (95% MCP, kernel; 50% MCP, kernel) as the repeated factor, was used to compare home range overlap between sex and method. The data were cube root transformed to make the residuals normal and homescedastic, although no transformation was found that made the residuals for the 50% MCP or kernel data normally distributed.

Population density

Territory mapping was used to estimate fisher population density (Arthur et al. 1989, Garant and Crête 1997, Fuller et al. 2001). Ninety-five percent MCP home ranges of radio-collared fishers during the non-breeding period were mapped, with one map being generated for each sex. Fishers with an insufficient number of locations to accurately map their home range (between 13 and 18 locations) were also included since I know that they occupied approximately that area. These individuals were not included in the calculation of the mean home range size, but I know where their minimum home ranges are located. Home range maps were overlaid with FRI land cover data and the wetland layer from the Natural Resources Values Information System (NRVIS: OMNR) (see Chapter 2) in a GIS (ArcMap 8.2, ESRI). Where suitable fisher habitat existed, an uncollared fisher of the same sex was assumed to inhabit an unoccupied circular home range of average size (6.4 km^2 for males and 2.1 km^2 for females). Suitable habitat was defined as an area composed of roughly the same amount of each habitat type found within the home ranges of radio-collared fishers (Table 1.1, see also chapter 2). For example, an area with < 6% or > 61% coniferous forest was considered unsuitable. Assumed home ranges of uncollared fishers were allowed to overlap slightly. A 100%

MCP around locations for all fishers included in the population density estimate was used to define the boundary of the study area (894 km^2; Figure 1.2). Minimum convex polygons were used as home range estimates for population density calculations rather than kernel estimates because the outer boundary of the home range is of interest, and contouring methods, such as the kernel, tend to overestimate home range size (Kernohan et al. 2001). A maximum population density estimate of resident fishers was also estimated by assuming that the 894 km^2 area was saturated by fishers of average home range size for each sex, irrespective of the habitat composition.

Results

Sample

The ratio of adult males: adult females: juvenile males: juvenile females in the sample population of fishers was 24: 20: 1: 15 (one unidentified female). The ages of individual fishers are presented in Appendix 3. Seventy-three percent of the fishers were adults when captured. Juveniles trapped in January and February were adults by March of the same year, since fishers ≥1 year old are considered to be adult (Krohn et al. 1994) and fishers are usually born in March (Wright and Coulter 1967, Leonard 1986). Thus, by March 2003 all of the tagged fishers were adults. One juvenile female fisher captured in October 2003 was at least 6 months old. This individual was not considered in the home range or habitat use analysis. The age determined by the sagittal crest corresponded with the age determined from the tooth analysis 78% of the time. Where there were discrepancies between the 2 aging techniques, the tooth analysis was considered more accurate. Eight out of 11 of the discrepancies were females (Figure 1.3).

Number of locations used to estimate 100% MCP home ranges

By visual inspection, the area of the 100% MCP home ranges began to asymptote at 20 locations (Figures 1.4 and 1.5). In some cases, such as 155.940, 155.740, 155.540, 155.720, 155.460 and 155.480, MCP home ranges were adequately sampled with 22 locations (i.e. home range size failed to increase with more locations). Home ranges of fishers 155.840b, 155.500, 155.480 and 155.999, however, did not asymptote even after 30 locations. There is a trade-off between including, in subsequent analyses, many animals with fewer locations and few animals with many locations (i.e., there are many animals in the sample with > 25 locations, and fewer animals with > 35 locations). I chose to include home ranges with ≥ 25 locations only, since home ranges had, for the most part, reached an asymptote by that point.

MCP home ranges

Minimum convex polygon home ranges were calculated for fishers with ≥ 25 locations between June 2003 and February 2004. Because the size and shape of MCPs are sensitive to outlying locations (Anderson 1982, Boulanger and White 1990, Burgman and Fox 2003), a range of 100 to 50% MCP home ranges were estimated for each fisher; the area of these polygons was plotted against the proportion of locations included in the polygon (Figure 1.6). The polygon area should decrease as more peripheral locations are removed; the leveling off of this rate is an indication that all of the outlying locations have been removed, leaving only locations within the core area (Kenward 1987, Mizutani and Jewell 1998). One-way RM ANOVAs (1 for each sex) with area of x% MCP (log transformed) as the repeated factor were used to determine at what x% MCP the variation among area of MCP became non-significant. For both sexes, there were significant within-subject effects (male: $F = 142.2$, $df = 6$, $p < 0.001$; female: $F = 170.6$, $df = 6$, $p < 0.001$). Difference contrasts using a C-matrix (Systat 1996) showed that there were significant differences in MCP area between all adjacent x% MCPs for both sexes (Table 1.2). These results make justifying the use of a particular x% MCP home range difficult. There is a tradeoff between using large % MCPs, such as 100% MCPs that, potentially, contain outlying locations, and small % MCPs that have fewer locations and, thus, may not contain enough locations for accurate home range estimation. I have chosen to use 95% MCPs so as to include most of the locations, and for comparison with other studies (see Fuller et al. 2001).

Home range estimation

Fisher home ranges during the non-breeding period are presented in Figures 1.7 (95% MCP), 1.8 (95% kernel: male) and 1.9 (95% kernel: female). Log transformed estimated home ranges of male fishers were significantly larger than those of females (Table 1.3), both annually ($F = 99.638$, $df = 1$, $p < 0.001$) and during the non-breeding period ($F = 62.829$, $df = 1$, $p < 0.001$). There was a significant difference in home range size between methods, both annually ($F = 29.057$, $df = 2$, $p < 0.001$) and during the non-breeding period ($F = 57.552$, $df = 2$, $p < 0.001$). There was a significant interaction between method and sex for annual home ranges ($F = 3.719$, $df = 2$, $p = 0.033$), indicating that the difference in home range size between sexes was larger for some methods than others. Thus, separate 1-way RM ANOVAs between methods were performed for each sex. There was a significant difference between methods for both females ($F = 27.768$, $df = 2$, $p < 0.001$) and males ($F = 14.533$, $df = 2$, $p < 0.001$). Bonferroni-corrected paired t-tests between methods revealed, for both sexes, no significant difference between 100% MCP and 95% kernel home range size, while the 95% MCP was significantly smaller than both the 100% MCP and the 95% kernel ($\alpha = 0.017$; Table 1.4). There was no significant interaction between method and sex for the non-breeding season home ranges ($F = 0.632$, $df = 2$, $p = 0.536$) (power = 0.736; see Appendix 7). Bonferroni-corrected paired t-tests between methods, pooled across sex, showed that home range size using 100% MCP > 95% kernel > 95% MCP ($\alpha = 0.017$; Table 1.4) during the non-breeding period.

Movements, home range shifts and breeding season dispersals

The movements of several fishers from the area where they were originally live trapped and radio-collared are presented in Appendix 4. Three adult fisher home ranges shifted throughout the study: shifts in home ranges of female fishers 155.020b and 155.980 and male fisher 155.560 are shown in Appendix 5. Two adult male fishers monitored over the breeding season of 2004 showed breeding season movements outside of their normal home ranges: movements of 155.460 and 155.560 are presented in Appendix 5. 155.560 had a shift in home range following the breeding season. Other male fishers either showed no breeding season movements (155.720) or there was not enough data to discern these movements (155.500, 155.300b).

Spacing patterns: Static interactions

Nine of the 19 female and 6 of the 13 male non-breeding period home ranges overlapped at least 1 other home range belonging to a member of the same sex (Table 1.5; see also Figures 1.7 – 1.9 and Appendix 6). Female fishers 155.139, 155.940 and 155.980 had overlapping home ranges between March and November 2003 (Appendix 6a), but in November 2003, 155.980 shifted her home range (Appendix 5d). The home ranges of these 3 fishers also overlapped from November 2003 – June 2004, but in a different orientation (Appendix 6b). Therefore, home range overlap during these two time periods was included in Table 1.5, but only November 2003 – June 2004 was used (chosen randomly by flipping a coin) in the RM ANOVA since these 2 periods are not independent samples.

There was no significant difference in home range overlap between males and females ($F = 0.002$, $df = 1$, $p = 0.967$), although the power to detect the observed difference was low (power = 0.050). There was a significant difference in home range overlap between methods ($F = 28.130$, $df = 3$, $p < 0.001$). There was no significant interaction between sex and method ($F = 0.434$, $df = 3$, $p = 0.730$; power = 0.237; Appendix 7). Bonferonni-corrected paired t-tests (Maxwell 1980, Hochberg and Tamhane 1987) between methods (cube-root-transformed) and pooled over sex revealed significant differences between all comparisons except 50% MCP and 50% kernels ($\alpha = 0.0083$; Table 1.6).

Population density

The population density of resident fishers in the 894 km^2 area, as determined by territory mapping assuming saturation by fishers where suitable habitat exists, is approximately 32.6/100 km^2 (male: 8.8/100 km^2, 79 individuals; female: 23.8/100 km^2, 213 individuals) (Figures 1.10 and 1.11). The maximum population density of resident fishers in this same area, assuming that the area is saturated by fishers, irrespective of habitat, is 63.2/100 km^2 (male: 15.6/100 km^2, 140 individuals; female: 47.6/100 km^2, 426 individuals). Locations for an additional 9 male and 13 female radio-collared fishers that were not used in the home range or density estimates were used to confirm the presence of fishers in other parts of the study area (Figures 1.12 and 1.13).

Discussion

Home range

Average male home ranges are four and three times larger than females for annual and non-breeding periods, respectively. The significant interaction between sex and method for annual home ranges indicates that this difference is greater for some estimation methods than others. This in turn suggests that direct comparisons of home range estimates among studies will be problematic if the studies involve different estimation methods (see also Harris et al. 1990). Using similar estimators for comparison, fisher home ranges in this study are smaller than those reported in the literature for both males and females, with the exception of Garant and Crête (1997), who reported home range sizes comparable to this study (Table 1.7). Powell (1993a) compared fisher home ranges across studies and found no geographical pattern in home range size.

Arthur et al. (1989) documented fishers shifting their home ranges in response to nearby home ranges becoming available, as evidenced by the movements of a collared fisher when a neighbouring collared fisher was removed. I did observe home range shifts of radio-collared fishers, although this was not likely in response to movements of other radio-collared fishers. It is possible that observed home range shifts resulted from the home ranges of uncollared fishers becoming available. Arthur et al. (1989) observed that an adult male fisher obtained a new home range following breeding season movements, as male fisher 155.560 in my study did.

Spacing patterns: Static interactions

Fisher showed considerable intrasexual overlap in home range: up to 71% (95% MCP) or 74% (95% kernel; Table 1.5) overlap of home ranges, and up to 32% (50% MCP) or 40% (50% kernel) overlap of core areas. There was a significant difference between the percent overlap of 95% home ranges and core areas, indicating that, though home ranges of adjacent fishers of the same sex may overlap, they share a significantly smaller percent of their core areas. Not all fishers exhibited intrasexual home range overlap, but this could be an artifact of not all fishers in the area being radio-collared and does not mean that they did not share portions of their home range with other, uncollared individuals. Thus, we do not know the true distribution of overlap. My study contrasts with previous work on fishers, reporting minimal (Arthur et al. 1989, Fuller et al. 2001, Zielinski et al. 2004) or no (Garant and Crête 1997) overlap of intrasexual fisher home ranges.

Territoriality should be evident in a population when the benefit of maintaining a territory is greater than its cost (Davies 1978). Powell (1993a, 1994) examined this concept mathematically in fishers. He suggested that carnivores in general (Powell 1993b) and fishers specifically (Powell 1993a, 1994), will vary their spacing based on prey population density and vulnerability. When prey densities are very low, fishers will be transient. As prey population densities increase, fishers will permit a gradient of overlap of home ranges such that at low prey density fishers will maintain exclusive territories, at medium prey density they will defend intrasexual territories, and at high prey density fishers should permit extensive home range overlap (Powell 1993a, 1994). If fisher spacing depends on food availability, as hypothesized by Powell, then the relatively small home ranges and intrasexual home range overlap in this study could be explained, in part, by high prey population densities in the study area. However, this study was not designed to test Powell's hypothesis, thus, I cannot conclude with any certainty that, in fact, fisher density in eastern Ontario is high because food density is high. Instead, it is one possible explanation of why fishers permit overlap of territories belonging to members of the opposite sex and, to some extent, the same sex.

Population density

Fisher population densities reported here are comparable to a study by Garant and Crête (1997) in Québec, < 100 km north of this study. However, genetic analysis has shown that these 2 populations are distinct (Carr et al., unpublished data). Population densities are higher than those reported in other areas (Table 1.8). Capture data from the Rabies Research and Development Unit's trap-vaccinate release program in eastern Ontario has shown that fisher population density in eastern Ontario has been declining since a peak in 2001 (Bowman et al., unpublished data; see Figure 5.1 Chapter 5). Thus, although fisher population densities are relatively high in eastern Ontario, they are likely declining.

Fuller et al. (2001) and Garant and Crête (1997) found that territory mapping gave similar population density estimates as camera mark-resight and track counting techniques, respectively. Fuller et al. (2001) cautioned that using territory mapping to estimate density requires that a high proportion of the population be marked and the presence of all non-marked animals is confirmed. In this study, roughly 10% of the population (9% of females, 15% of males) was marked, which is much lower than Fuller et al.'s (2001) study (≥ 53-55%). Additionally, there were no radio-collared female fishers in the northwestern portion of the 894 km^2 area (Figure 1.11). I feel that there is no reason why this area would not be colonized by female fishers of average home range size, since radio-collared male fishers inhabit this portion of the study area (Figure 1.10). Furthermore, locations of radio-collared fishers that were not used in the home range or density estimates show that fishers of both sexes inhabit some areas that I have assumed were occupied by uncollared fishers (Figures 1.12 and 1.13).

These estimates of fisher population density are for resident adult fishers only. During the time when fisher kits are born but before they disperse (between October and February; Arthur et al. 1993), the population density of fishers will be greater than described here (Arthur et al. 1989). This estimate also does not include transient individuals.

In summary, fisher home ranges in eastern Ontario are relatively small and overlap slightly. Both of these observations imply that fisher population densities in eastern Ontario are relatively high. Management implications of this will be discussed in the final chapter.

Tables and Figures

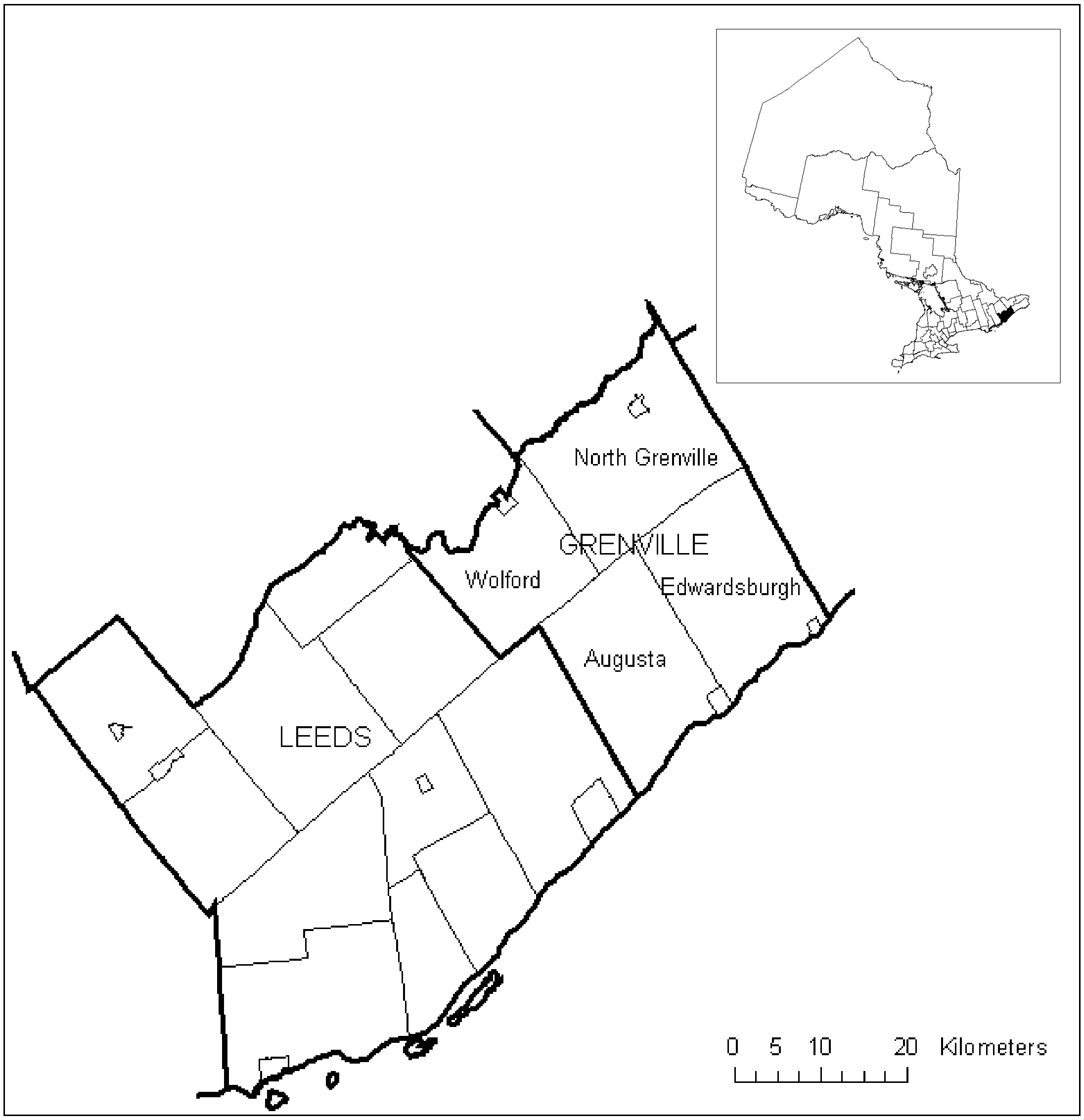


Figure 1.1. Leeds and Grenville County, Ontario, showing the townships of Grenville County, and the County's position within Ontario (inset).

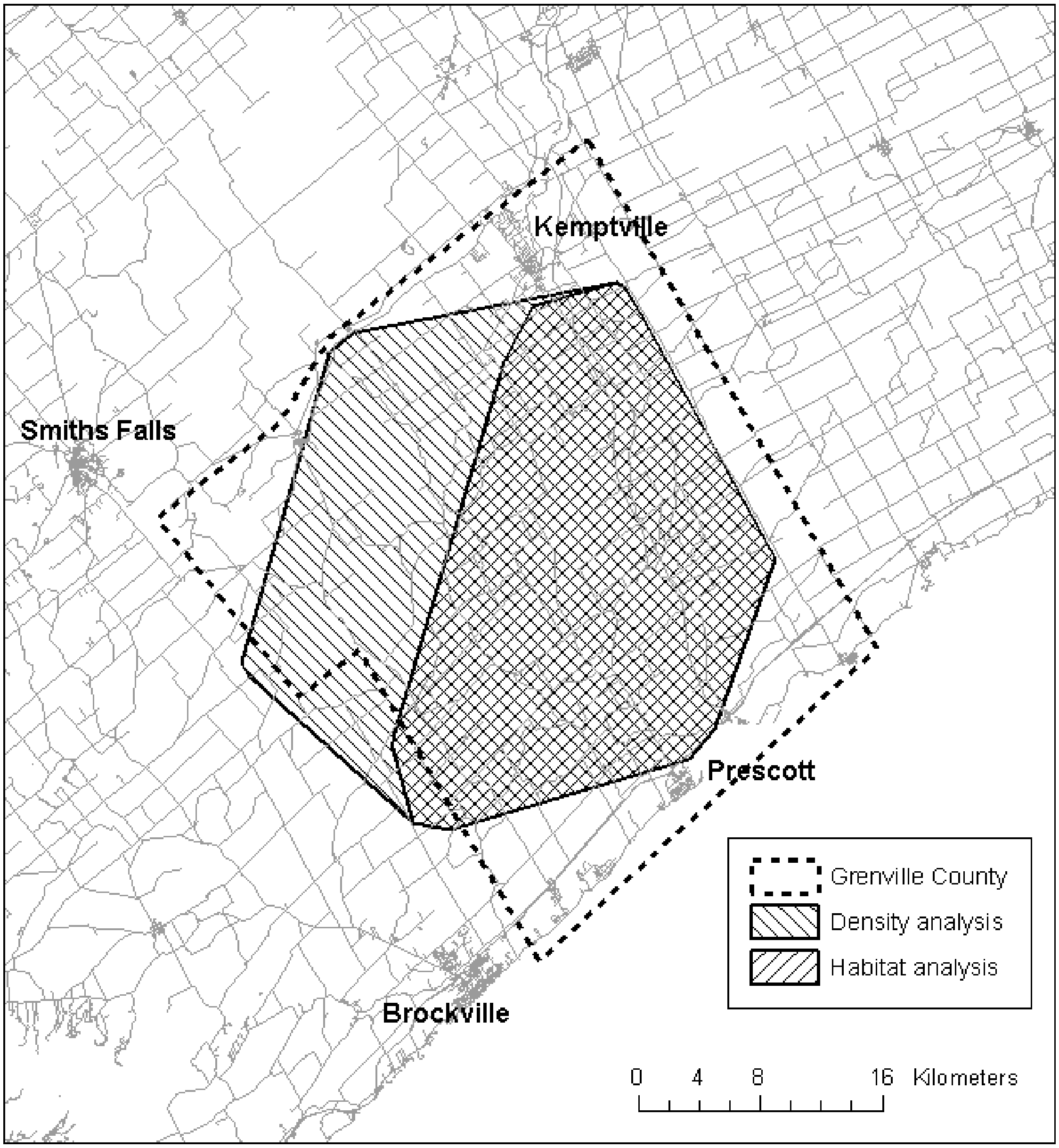
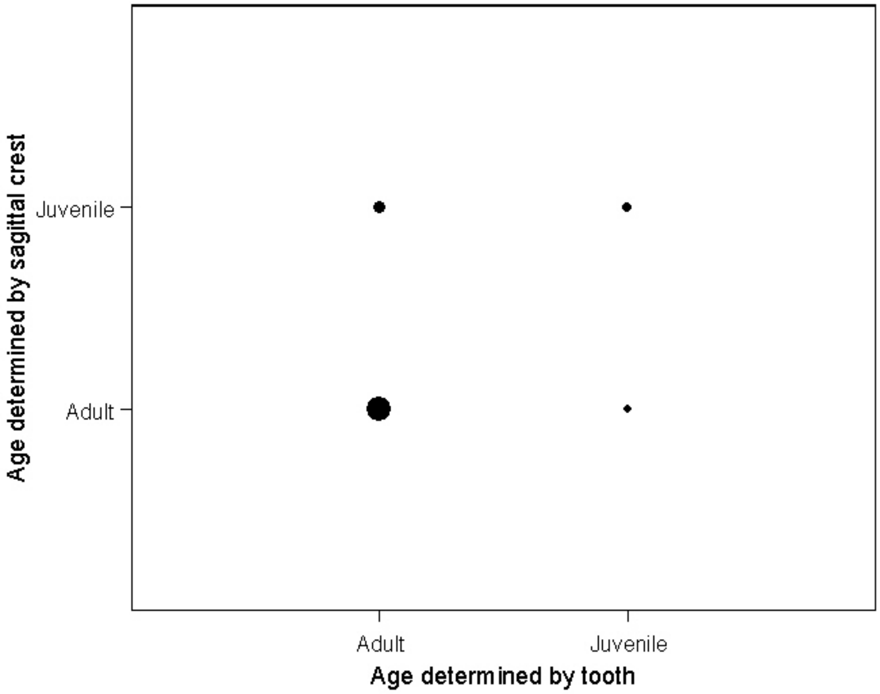


Figure 1.2. The study area, showing Grenville County, the area used for the population density analysis, and the area used for the habitat analysis.

Table 1.1. Minimum and maximum proportions of habitat for 26 (16 female, 10 male) adult fisher home ranges (95% kernel) between June 2003 and February 2004 in eastern Ontario (see Chapter 2).

	Coniferous	Deciduous	Wetland	Field
Minimum proportion	0.06	0.03	0.00	0.10
Maximum proportion	0.61	0.44	0.64	0.51
Mean proportion (± 1 SD)	0.29 (0.14)	0.25 (0.11)	0.16 (0.17)	0.30 (0.11)

a)



b)

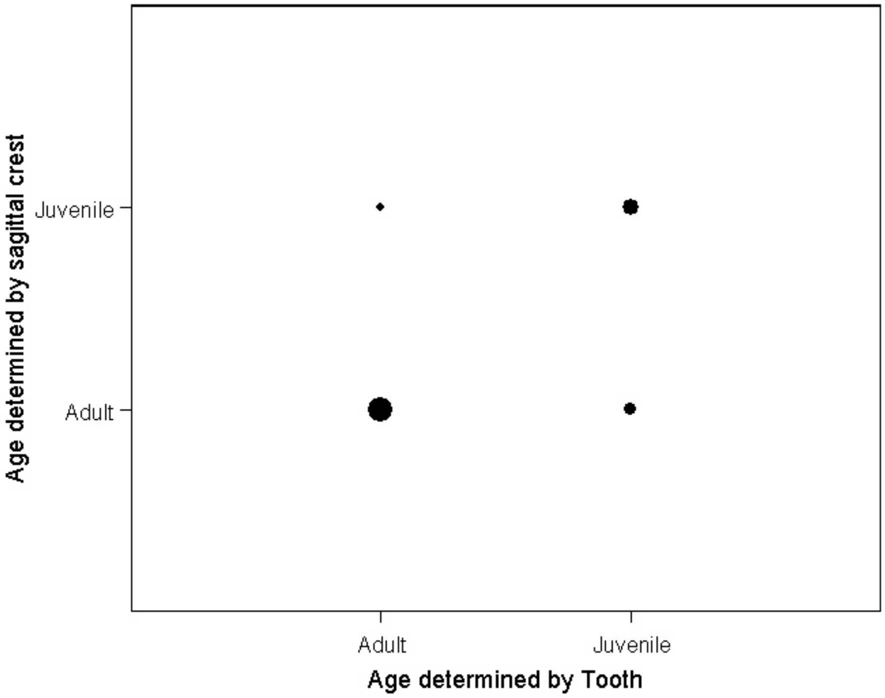


Figure 1.3. Age of 50 fishers as determined by analysis of tooth pulp cavity and cementum annuli, and size of sagittal crest. Size of circles represents frequency within each category as a proportion. a) males (n = 21), b) females (n = 29). The age determined by the tooth is considered more accurate than the saggital crest.

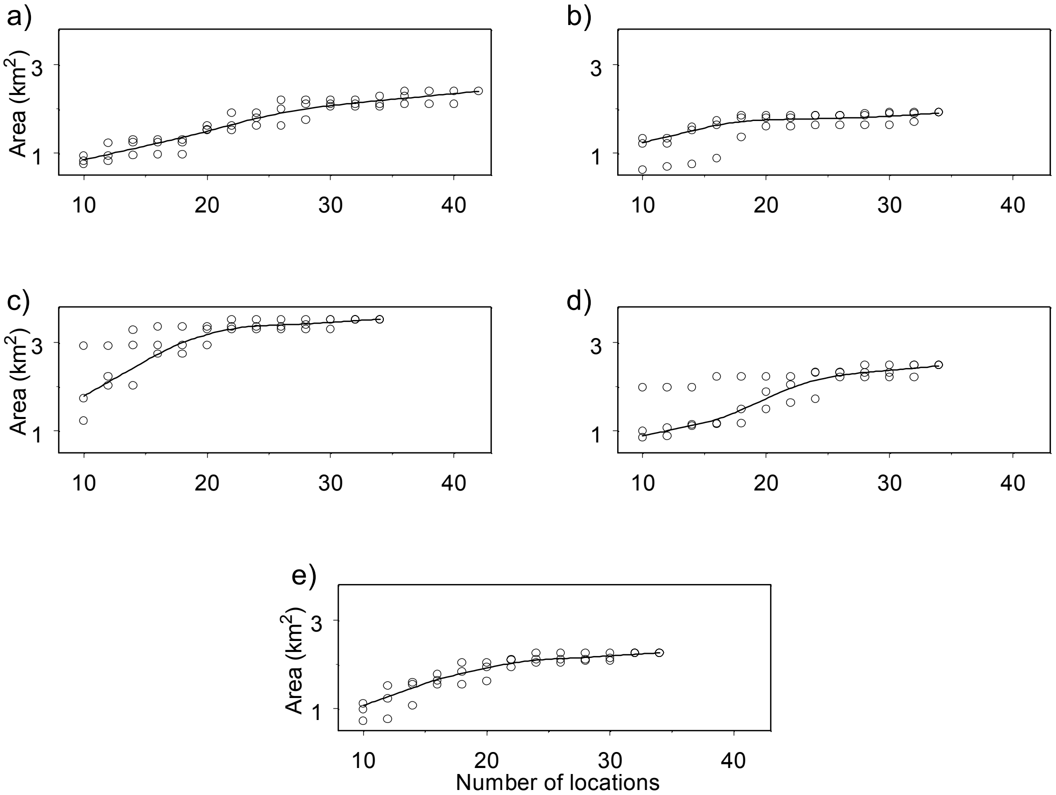


Figure 1.4. Estimated area of female fisher 100% minimum convex polygon home range vs. the number of locations comprising the home range, with a fitted LOESS curve. Locations were randomly selected and the home range calculation repeated 3 times for each individual, for a given number of sample locations (N = 10, 12, …). a) 155.840b, b) 155.940, c) 155.740, d) 155.581, e) 155.540.

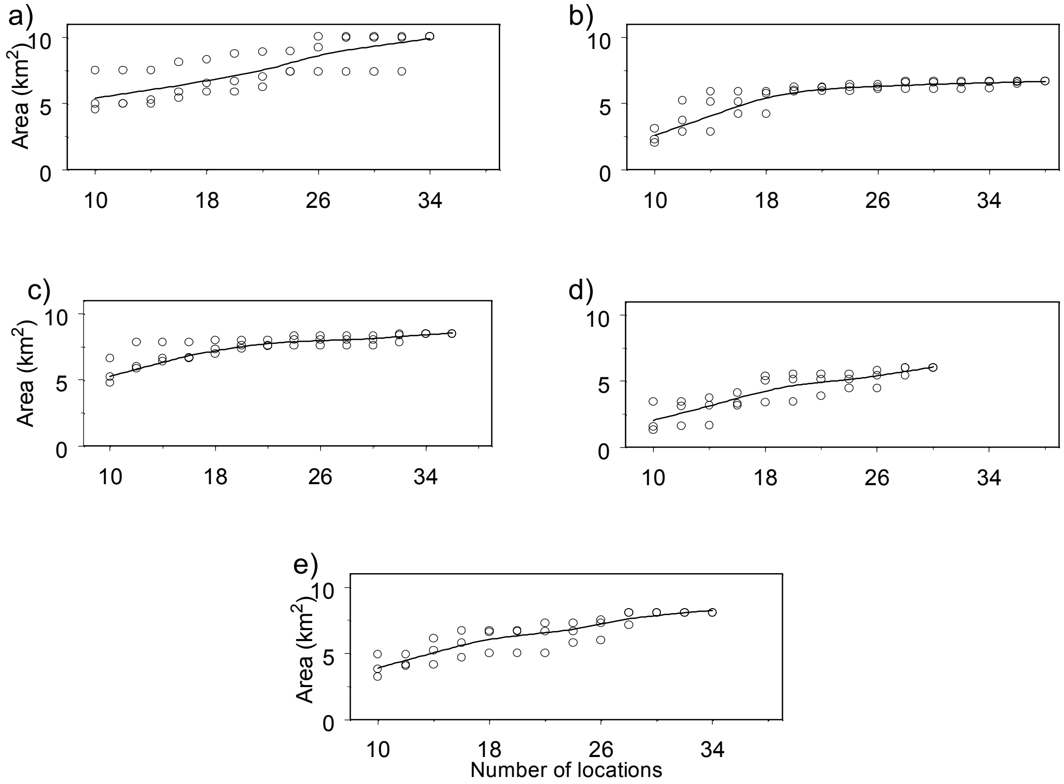


Figure 1.5. Estimated area of male fisher 100% minimum convex polygon home range vs. the number of locations comprising the home range, with a fitted LOESS curve. Locations were randomly selected and the home range calculation repeated 3 times for each individual, for a given number of sample locations (N = 10, 12, …). a) 155.500, b) 155.720, c) 155.460, d) 155.480, e) 155.999.

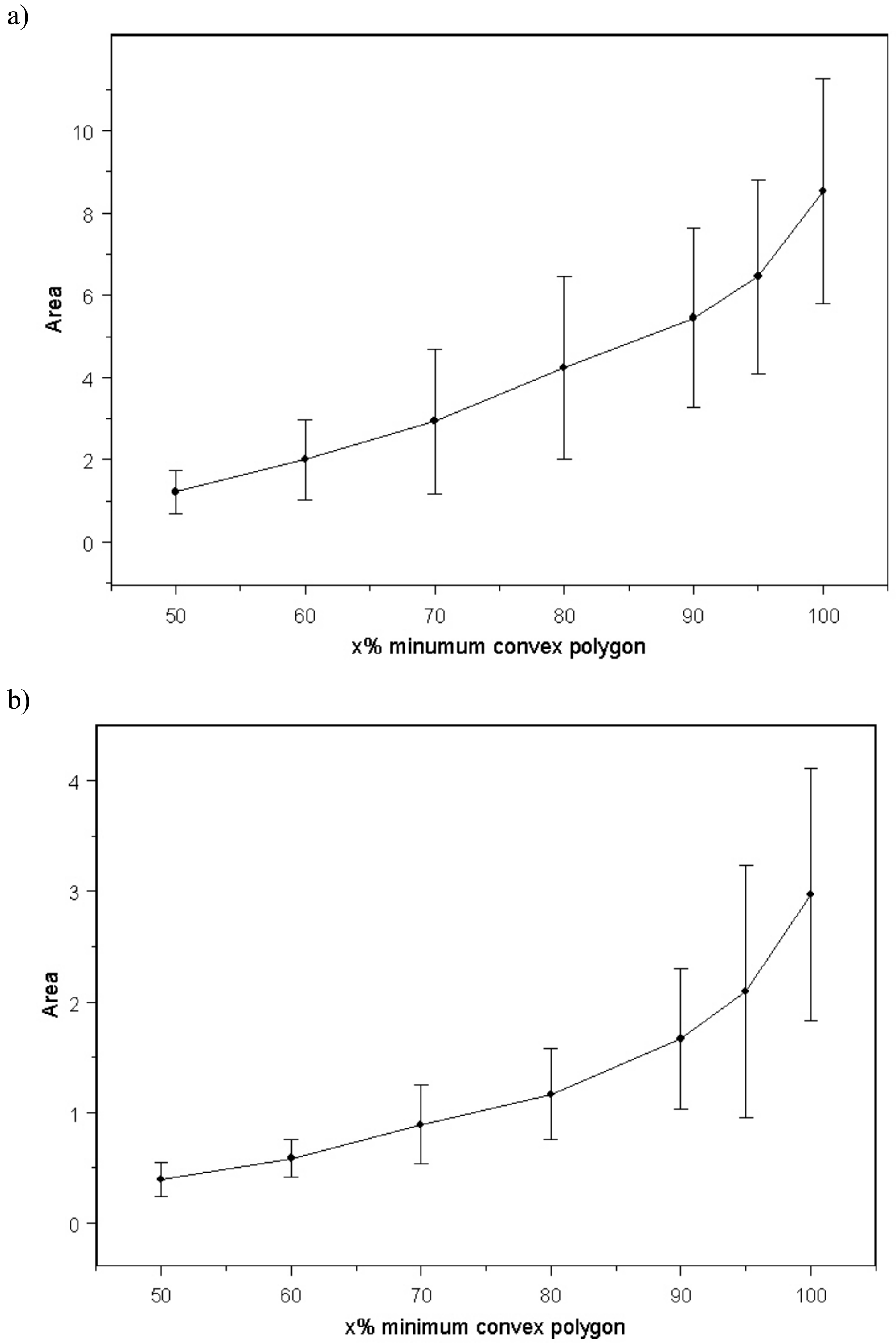


Figure 1.6. Mean (± 1 SD) estimated area (km^2) of minimum convex polygon home ranges with 5 to 50% of the locations furthest from the arithmetic mean of x,y coordinates excluded. Home ranges are for a) male (n = 10) and b) female (n = 16) fishers in eastern Ontario between June 2003 and February 2004.

Table 1.2. Results (F and p values) of difference contrasts between adjacent x% MCP home ranges for male and female fishers.

x% MCPs compared	Male		Female	
	F	p	F	p
100 and 95	41.7	< 0.001	28.4	< 0.001
95 and 90	19.7	0.002	25.1	< 0.001
90 and 80	21.3	0.001	76.3	< 0.001
80 and 70	40.7	< 0.001	131.4	< 0.001
70 and 60	51.2	< 0.001	75.5	< 0.001
60 and 50	33.4	< 0.001	65.6	< 0.001

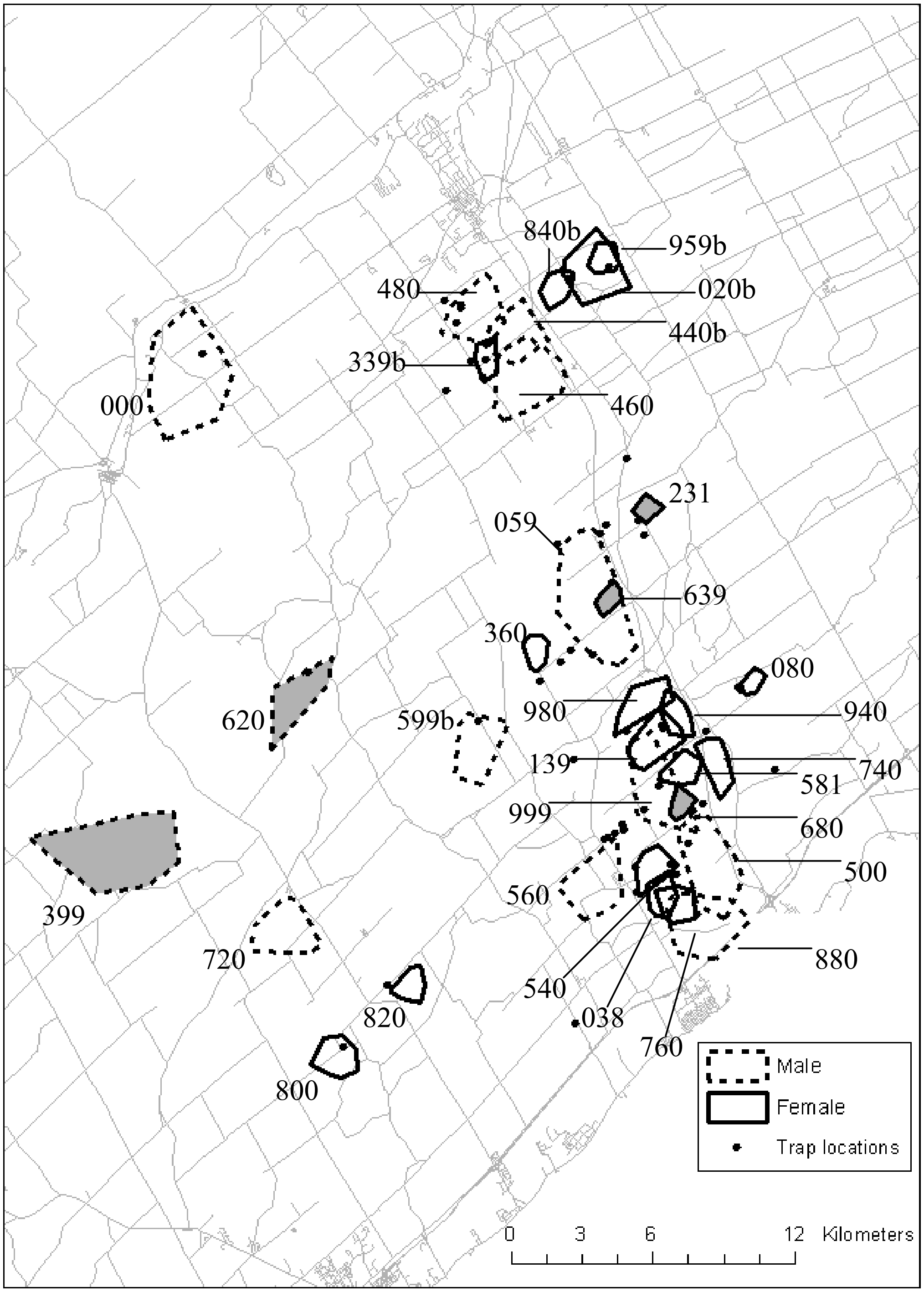


Figure 1.7. 95% minimum convex polygon home ranges for fishers during the non-breeding period (June 2003 - February 2004). Two male and 3 female home ranges were incomplete (shaded polygons): 155.399 and 155.620; and 155.231, 155.680 and 155.639 (based on 13 and 14; and 15, 17 and 18 locations, respectively). Locations where fishers were live trapped are also shown.

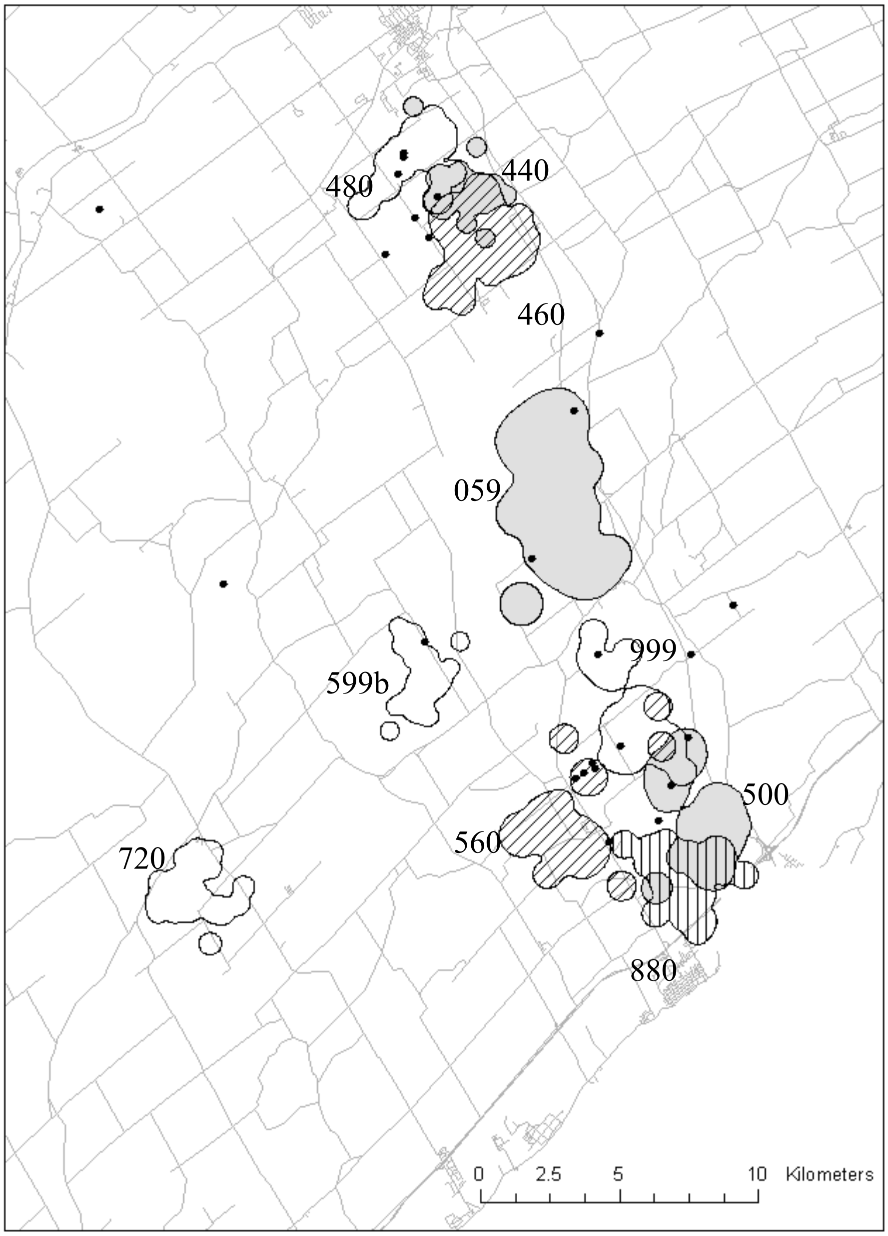


Figure 1.8. 95% kernel home ranges of 10 male fishers during the non-breeding period (June 2003 - February 2004). Shading and cross-hatching is used to distinguish between individuals with overlapping home ranges. Locations of live traps for male fishers are shown (●).

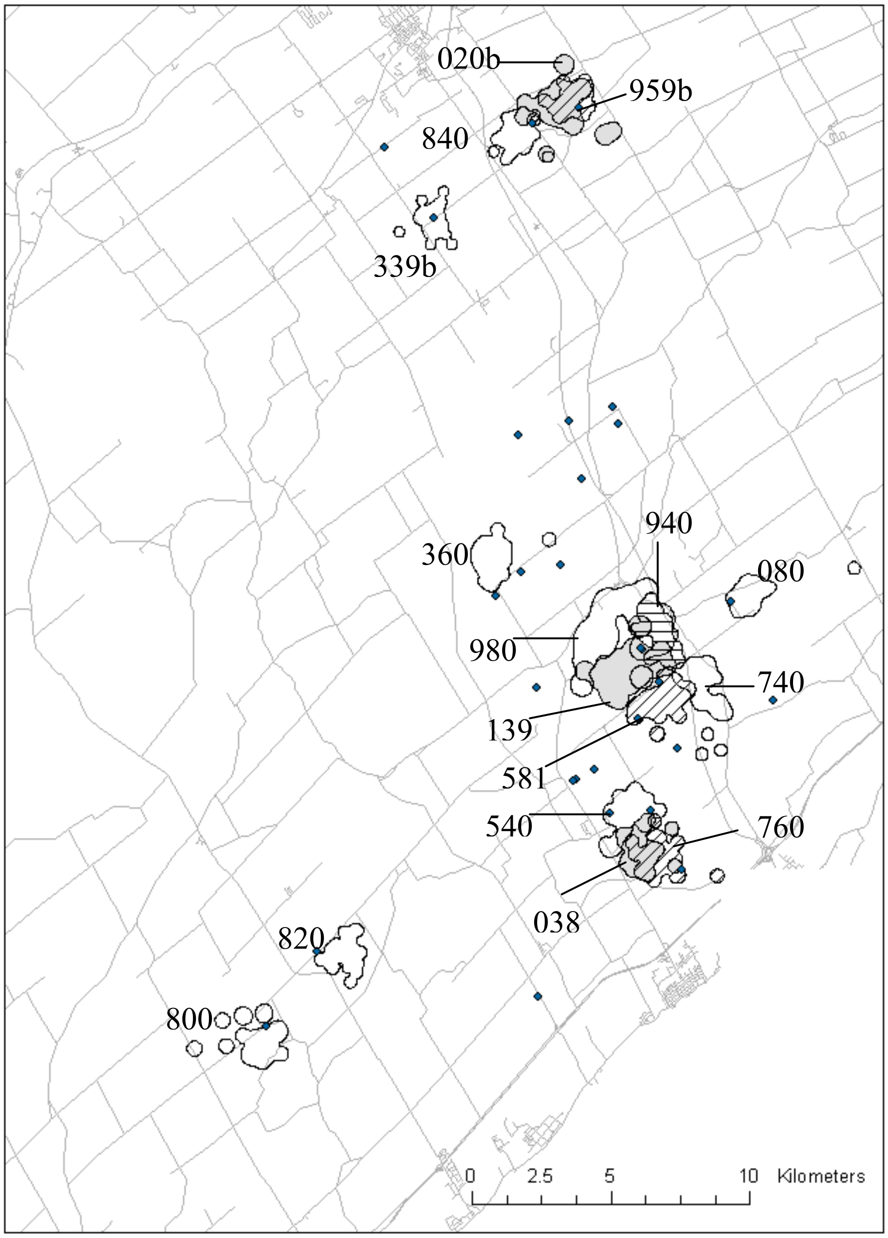


Figure 1.9. 95% kernel home ranges of 16 female fishers during the non-breeding period (June 2003 - February 2004). Shading and cross-hatching is used to distinguish between individuals with overlapping home ranges. Locations of live traps for female fishers are shown (●).

Table 1.3. Average (± 1 SD) annual[1] and non-breeding season[2] home range size (km^2) of adult fishers in eastern Ontario.

Sex	n[3]	n (loc)[4]	100% MCP[5]	95% MCP[6]	Kernel[7] 95%	Kernel[7] 80%	Kernel[7] 60%	Kernel[7] 40%	Kernel[7] 20%
Male[1]	8	42	14 (5.8)	11 (4.4)	16 (6.0)	8.0 (3.5)	4.0 (1.9)	1.9 (0.83)	0.67 (0.22)
Male[2]	10	34	8.5 (2.7)	6.4 (2.3)	11 (6.0)	5.3 (2.6)	2.5 (1.3)	1.1 (0.50)	0.39 (0.22)
Female[1]	15	42	3.6 (1.7)	2.1 (0.8)	3.6 (1.3)	1.7 (0.59)	0.82 (0.30)	0.37 (0.17)	0.12 (0.057)
Female[2]	16	34	3.0 (1.1)	2.1 (1.1)	3.3 (1.2)	1.6 (0.64)	0.76 (0.26)	0.35 (0.15)	0.12 (0.060)

[1] Home ranges from March 2003 - February 2004 or June 2003 - May 2004 (latter range for “b” fishers or fishers with a shift in home range)

[2] Home ranges from June 2003 - February 2004

[3] Number of fishers

[4] Average number of locations per home range

[5] Area of 100% minimum convex polygon home range

[6] Area of 95% minimum convex polygon home range

[7] Area of x% kernel home range

Table 1.4. Results of multiple comparisons tests (t and p values) between methods of home range size estimation annually and during the non-breeding period for male and female fishers.

Period	Sex	df[1]	100% MCP vs. 95% MCP		100% MCP vs. 95% Kernel		95% MCP vs. 95% Kernel	
			t	p	t	p	t	p
Annual	Female	14	5.11	< 0.001	-0.486	0.634	6.33	< 0.001
	Male	7	4.76	0.002	-1.79	0.116	6.16	< 0.001
Non-breeding[3]	Pooled[2]	25	7.22	< 0.001	-3.26	0.003	11.8	< 0.001

[1] Degrees of freedom

[2] Home ranges pooled over males and females

[3] June 2003- February 2004

Table 1.5. Intrasexual percent overlap of fisher home ranges estimated by minimum convex polygon (MCP) and kernel techniques.

Fisher ID	Sex	Time Interval[1]	MCP		Kernel	
			95%	50%	95%	50%
155.139[2]	Female	Mar '03 – Nov '03	14.3	0	33.2	0
155.940[2]	Female	Mar '03 – Nov '03	41.8	0	64.4	0
155.980[2]	Female	Mar '03 – Nov '03	20.8	0	38.7	0
155.139	Female	Nov '03 – Jun '04	0.761	0	15.9	0
155.940	Female	Nov '03 – Jun '04	38.5	0	66.9	0
155.980	Female	Nov '03 – Jun '04	39.1	0	47.6	0
155.540	Female	Feb '03 – Jun '04	17.1	0	24.3	0
155.760	Female	Feb '03 – Jun '04	0.904	3.94	49.4	6.80
155.038	Female	Feb '03 – Jun '04	71.0	5.12	73.2	10.7
155.020b	Female	June'03 – Nov '03	66.4	13.9	66.7	2.47
155.959b	Female	June'03 – Nov '03	61.5	13.4	61.8	2.46
155.840b	Female	June'03 – Nov '03	0	0	10.1	0
155.460	Male	Jun '03 – Feb '04	14.7	0	27.3	0
155.480	Male	Jun '03 – Feb '04	8.57	0	20.0	0
155.440b	Male	Jun '03 – Feb '04	34.3	0	73.8	0
155.999	Male	Jun '03 – Dec '03	2.75	0	17.5	0
155.500	Male	Jun '03 – Dec '03	14.7	30.8	43.9	24.0
155.880	Male	Jun '03 – Dec '03	12.1	31.7	31.5	40.3
Average	Female		32.8	4.04	46.2	2.49
			(29.3)	(5.78)	(23.8)	(3.82)
Average	Male		14.5	10.4	35.7	10.9
			(10.7)	(16.1)	(20.9)	(17.5)

[1] Time interval when fishers with overlapping home ranges were monitored simultaneously. Data from February 2004 was included for male fishers 155.460, 155.480 and 155.440b since these individuals had not begun breeding season movements

[2] These data were not included in the average or the RM ANOVA because the inclusion of both violates the assumption of independence

Note: numbers in parentheses are standard deviations

Table 1.6. Results of multiple comparisons tests for differences in home range overlap between 4 estimates of home range. Degrees of freedom = 17.

	95% MCP		50% MCP		95% kernel	
	t	p	t	p	t	p
50% MCP	3.980	0.001				
95% kernel	-4.263	0.001	7.871	< 0.001		
50% kernel	3.964	0.001	-0.752	0.464	-7.871	< 0.001

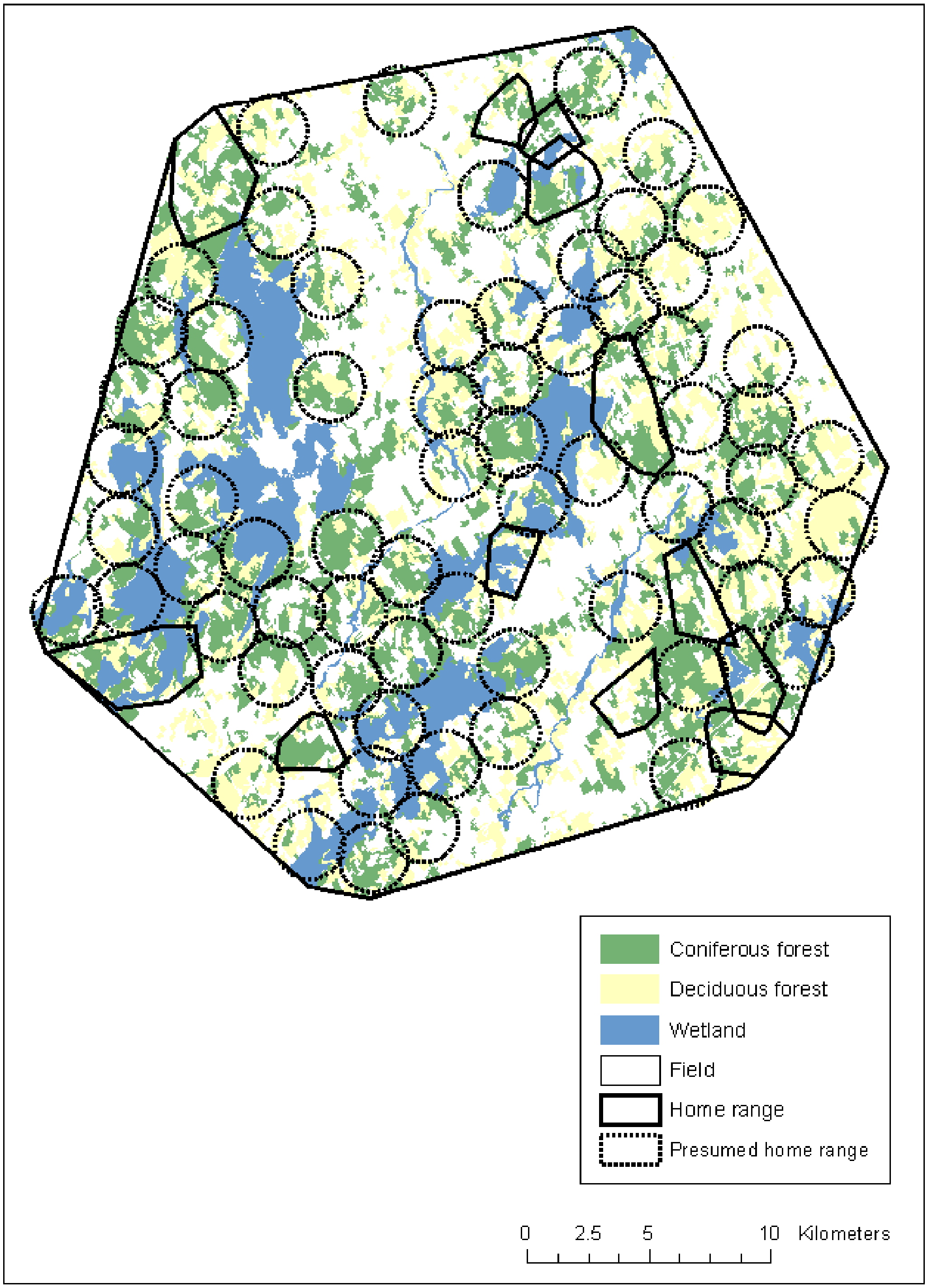


Figure 1.10. Estimate of male fisher population density during the non-breeding period (June 2003 – February 2004). Home ranges of collared male fishers are estimated using 95% minimum convex polygons and presumed home ranges of uncollared fishers are of average home range size (6.4 km^2) for collared male fishers during this time. Landcover data is from 1978 FRI (OMNR) and the wetland layer is from NRVIS (OMNR).

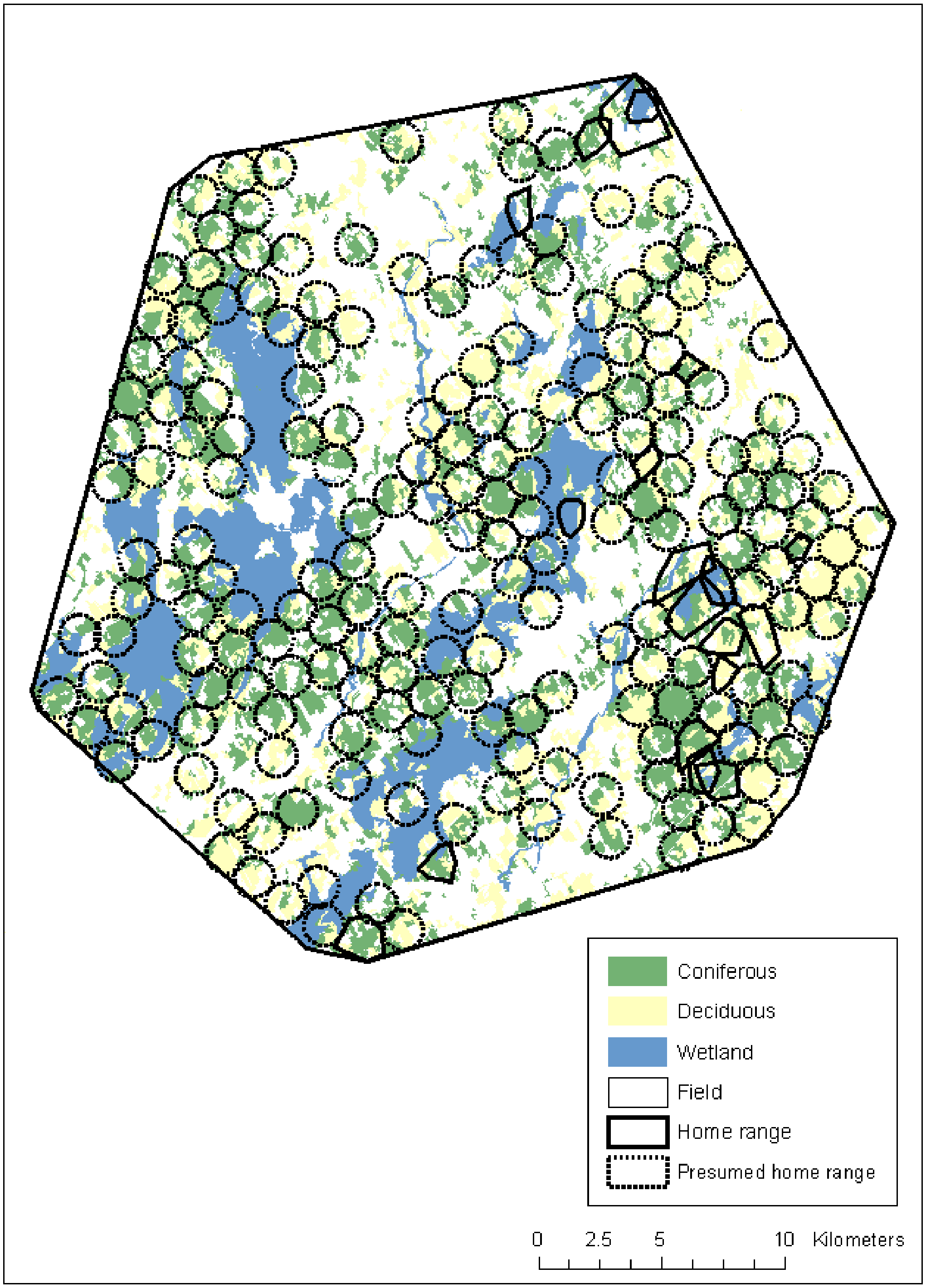


Figure 1.11. Estimate of female fisher population density during the non-breeding period (June 2003 – February 2004). Home ranges of collared female fishers are estimated using 95% minimum convex polygons and presumed home ranges of uncollared fishers are of average home range size (2.1 km^2) for collared female fishers during this time. Landcover data is from 1978 FRI (OMNR) and the wetland layer is from NRVIS (OMNR).

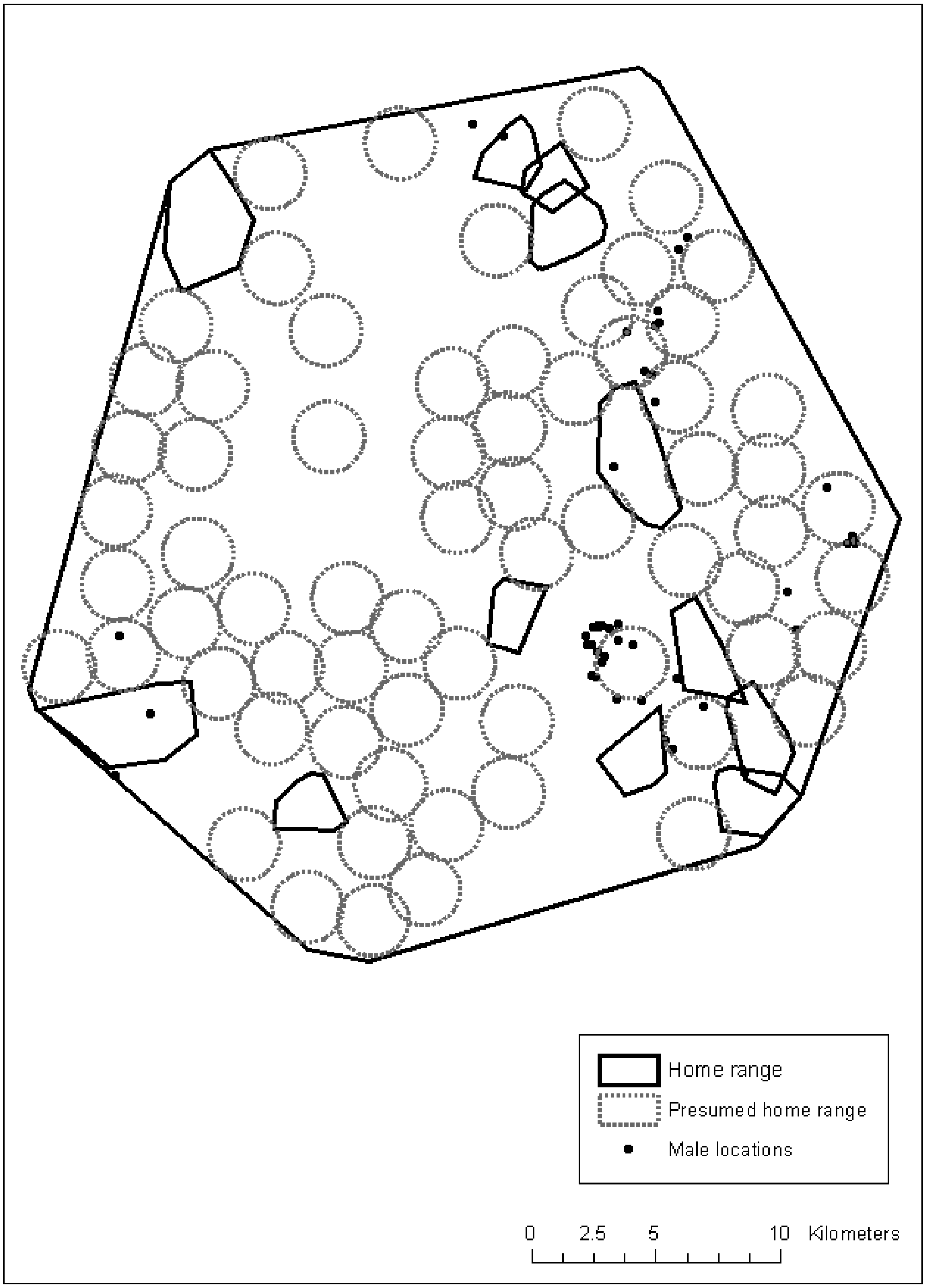


Figure 1.12. Male fisher home ranges, presumed home ranges of uncollared fishers and locations of radio-collared fishers that were not included in the home range or density estimates. Home ranges of collared male fishers are estimated using 95% minimum convex polygons during the non-breeding season (June 2003 - February 2004).

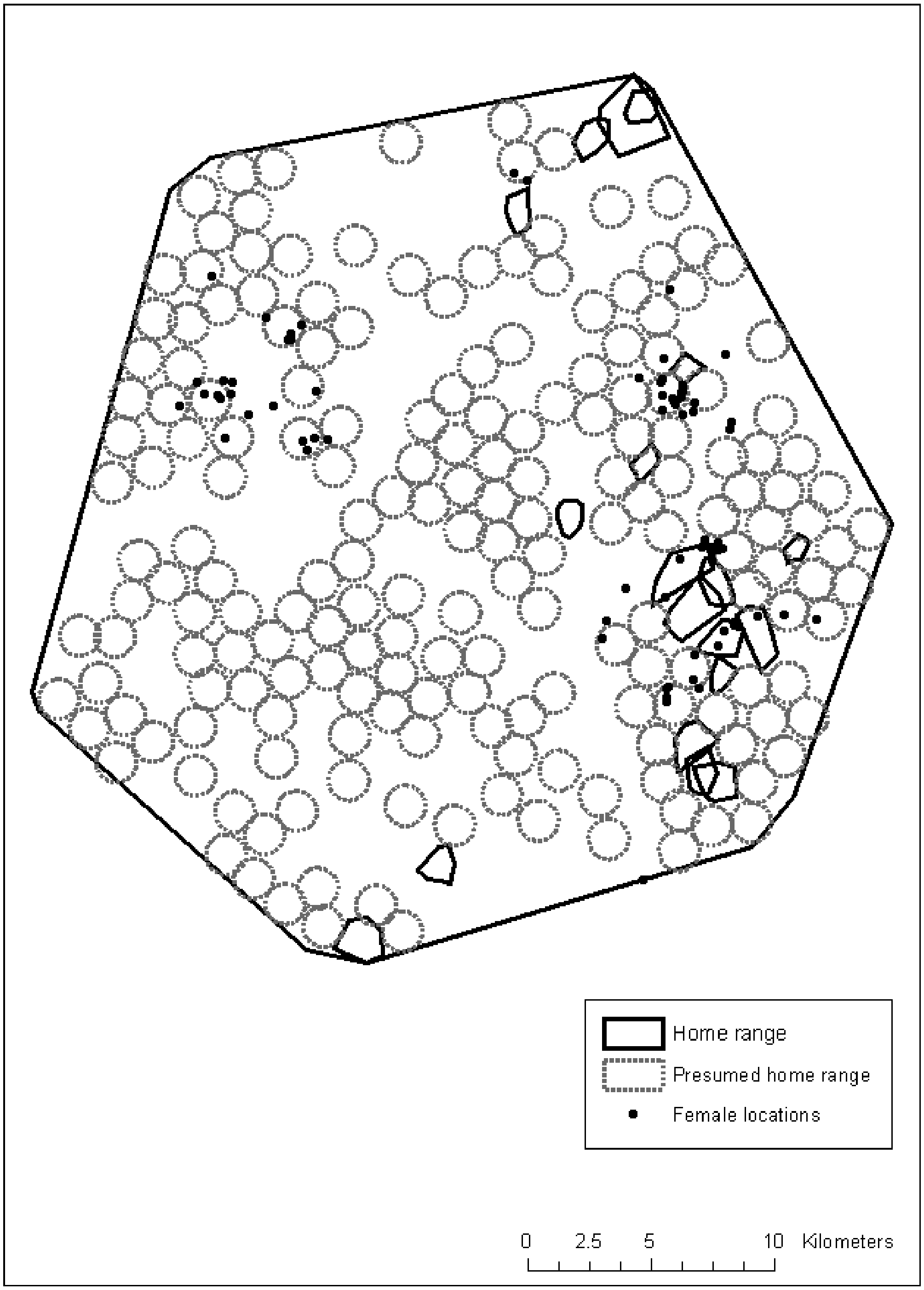


Figure 1.13. Female fisher home ranges, presumed home ranges of uncollared fishers and locations of radio-collared fishers that were not included in the home range or density estimates. Home ranges of collared female fishers are estimated using 95% minimum convex polygons during the non-breeding season (June 2003 - February 2004).

Table 1.7. Mean adult fisher home range size (km^2) reported in the literature.

Location	Sex	n	Home range size	Estimator used	Reference
Ontario (eastern)	M	10	6.4 (2.3 SD)	95% MCP (Jun-Feb)	This study
	F	16	2.1 (1.1 SD)		
	M	8	14 (5.8 SD)	100% MCP (annual)	
	F	15	3.6 (1.7 SD)		
Québec	M	3	9.2 (1.8 SE)	100% MCP	Garant and Crête (1997)
	F	7	5.4 (0.9 SE)		
Massachusetts	M	7	10 (range 6.5 – 16.6)	95% MCP (annual)	Fuller et al. (2001)
	F	15	7.6 (range 2.9 – 11.1)		
Maine	M	7	31 (9.3 SE)	100% MCP (May-Dec)	Arthur et al. (1989)
	F	6	16 (4.7 SE)		
California (coastal)	M	2	58 (29.6 SE)	100% MCP	Zielinski et al. (2004)
	F	7	15 (2.16 SE)		
California (Sierra)	M	4	30 (7.8 SE)		
	F	8	5.3 (0.65 SE)		
British Columbia	M	1	73.9	90% adaptive kernel (winter)	Weir and Harestad (2003)
	F	9	25.0		

Table 1.8. Estimated fisher population densities reported in the literature.

Location	Density (/100km^2)	Method used	Reference
Ontario (eastern)	32.6	Territory mapping (suitable habitat)[1]	This study
	63.2	Territory mapping (max. density)[2]	
Ontario (Central)	15	Harvest records (pre-harvest density)	Douglas and Strickland (1987)
Québec	27	Territory mapping (max. density)[2]	Garant and Crête (1997)
	30	Track count survey	
Massachusetts	19 - 23	Territory mapping[3]	Fuller et al. (2001)
	21 - 25	Camera-mark-resight	
Maine	9.5[4] – 35.7[5]	Territory mapping (summer)[6]	Arthur et al. (1989)
	5[4] – 12[5]	Territory mapping (winter)	
Michigan (Upper peninsula)	7.7	Live trapping, tracking and questionnaires	Powell (1993a)

[1] Population density considering suitable habitat only
[2] Population density irrespective of suitable habitat
[3] Unoccupied territories were identified by camera capture-mark-resight techniques
[4] Minimum estimate is the number of tagged fishers/area
[5] Maximum estimate is irrespective of suitable habitat
[6] Summer estimates include an estimate of number of untagged offspring in the area
[7] Number of fishers harvested per area; represents a minimum density

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Chapter 2: Habitat selection of fishers in eastern Ontario using compositional analysis at multiple scales

Introduction

Analysis of a species' use of habitat is valuable for wildlife management. This knowledge can aid in preservation of habitat used by a population for feeding, resting, reproducing and rearing young, which can have direct effects on production and survival (Van Horne 1983). Habitat use data can also be used in planning reintroduction programs (Kilpatrick and Rego 1994, Aubry and Lewis 2003). These data are vital for fisher management; the fishers' recent recolonization of eastern Ontario has been correlated with an increase in forested land (Lancaster et al., unpublished data), illustrating the importance of suitable habitat for fisher population growth. Knowledge of local fisher habitat preference on a relative scale is also valuable for estimates of population density.

Fishers are conventionally thought to be habitat specialists (Buskirk and Powell 1994), selecting dense, mature coniferous forests (Allen 1983). Powell (1993, 1994) concluded that fishers in Michigan used lowland coniferous forests more than what was expected based on the availability of this forest type. A study in Maine, however, found that fishers inhabit secondary forests with a mixture of forest types (Arthur et al. 1989). Similarly, Kilpatrick and Rego (1994) found that fishers in Connecticut used hardwood, softwood, and mixed forests for resting. Allen (1983) assessed the habitat suitability of different forest types for fishers, concluding that forests with large trees and extensive canopy closure comprising less than 50% deciduous trees had the highest habitat suitability index (HSI). A test of the HSI for fishers in Michigan showed that fishers preferred habitats with a high HSI value (Thomasma et al. 1991, 1994).

Resource selection is often measured as the use of a resource versus its availability: if the proportion of habitat that an animal uses is greater than the proportional availability of the habitat, the animal is said to have selected that habitat (Johnson 1980, White and Garrott 1990). The comparison between habitat use and availability can be done at several scales, including an animal's selection of habitat within its home range and selection of its home range within the landscape (Johnson 1980).

There are many methods of analysing habitat use versus availability that have been reviewed and critiqued (Alldredge and Ratti 1986, 1992, Aebischer et al. 1993, Manly et al. 1993, McClean et al. 1998, Erikson et al. 2001, Bingham and Brennan 2004). Methods that use radio-telemetry *locations* as the sample unit (i.e. labeling each location according to the habitat in which it falls) are not desirable since the aim of habitat preference studies is to estimate the preference of habitats for the population. As such, the appropriate sampling units are individual *animals* (Aebischer et al. 1993). Additionally, using animals as the sample unit allows the investigation of differences in habitat selection between sexes and seasons (Aebischer et al. 1993).

Methods that compare proportional use and availability of habitats are limited due to the unit sum constraint: because the proportions of habitats within a defined area sum to one, the use of one habitat type will necessarily result in apparent avoidance of at least one other habitat type. Moreover, when using radio-telemetry locations obtained remotely by triangulation, triangulation error can result in misclassification of the habitat at those locations, especially when the size of habitat patches is smaller than the estimated triangulation error (Nams 1989). Thus, methods of habitat selection analysis that do not classify individual points into their respective habitats are needed when triangulation error is large. Proponents of both compositional and Euclidean distance analyses claim to avoid these problems (Aebischer et al. 1993, Conner et al. 2003).

The landscape in eastern Ontario is highly agricultural and extrapolation of habitat selection results from other fisher studies in more forested landscapes (e.g., Weir and Harestad 2003) may not be wise. Therefore, this study uses compositional analysis to determine if fishers in eastern Ontario are selectively using different habitats during the non-breeding season. Habitat selection will be estimated at 2 scales: selection of the home range within the study area (coarse scale), and selection of the core use area within the home range (fine scale).

Methods

Map accuracy assessment

Remote-sensed land cover maps

Two different land cover maps were used to assess fisher habitat selection: Forest Resource Inventory (FRI) and Provincial-Scale Ontario land cover maps, both based on remotely-sensed data. Digital FRI maps for Leeds and Grenville County were obtained from the Ontario Ministry of Natural Resources (OMNR). FRI maps were created from aerial photographs from the summer of 1978 at a scale of 1:10,000. Accuracy assessment of these data has not previously been conducted. FRI data recognizes 32 tree species (Appendix 8a), categorized into 23 working groups (Appendix 8b). A working group label is assigned to the tree species making up at least 60% of the stand. The classification system of the FRI data was subsequently merged into 3 categories (coniferous forest, deciduous forest, and field; Appendix 8c), and a 4th category, wetland, was added from the Natural Resources Values Information System (NRVIS; OMNR). The merging of working groups into the 4-class FRI map was done *a posteriori* to reduce the resolution of the classification, thereby increasing map accuracy relative to the ground. The accuracy of the 2 FRI maps (the 23 working group and 4-class) was compared.

The OMNR generated remotely-sensed satellite imagery from multispectral Landsat Thematic Mapper (TM) data to create the Provincial-Scale Ontario land cover maps (Ontario Ministry of Natural Resources 1991-1998). The imagery was recorded between 1986 and 1997, with most data collected in the early 1990s, at a 25 m spatial resolution. A description of the 28 land cover classes and a condensed 15-class version can be found in Appendix 9. The accuracy of these data has not previously been assessed, but producers assert that 90% of forest classes and 85% of agricultural land cover is accurately mapped (Ontario Ministry of Natural Resources 1999).

Collection of ground-truthed data

Reference habitat data were collected on the ground between March and July, 2004 within the study area (Appendix 10). Due to time and budget constraints, sample sites were not randomly selected. Instead, conveniently located sites were chosen while driving or walking through the study area. Tree species present, dominant tree species (comprising $\geq$ 50% of the stand), and/or wetland type were recorded for 300 approximately 25 m^2 plots. Location coordinates were recorded with a GPS (Garmin GPS12, Garmin International, Inc. in NAD 83) with a spatial resolution of 4-12 m. Based on these data, each plot was classified according to the classification defined for the land cover map of interest. For example, a ground-truthed sample plot that contained mostly trembling aspen (*Populus tremuloides*) with some white cedar (*Thuja occidentalis*) was assigned a reference label of "PO" (poplar; see Appendix 8b) for the FRI data, "deciduous forest" for the 4-class FRI data, "mixed forest, mostly deciduous" for the 28-class Landsat TM data, and "mixed forest" for the 15-class Landsat TM data. In some instances, reference plots could not be assigned unequivocally to a single class: in these cases, plots were assigned to both a primary and secondary habitat classification.

Accuracy of land cover maps

The classification of a reference plot based on ground-truthed data was compared to the corresponding classification based on the two land cover maps (FRI and Landsat TM). The classification of a plot was considered to be accurate if (1) the map classification corresponded to the primary or secondary ground-truthed label assigned to the plot; or (2) if either the primary or secondary reference label matched a map polygon or pixel within 10 meters of the reference plot.

As reference data were not sampled randomly, some reference plots were located in close proximity to others (for example, if data plots were chosen along a trail). This could cause over or underestimation of the map error as follows: if a particular forest stand was misclassified (perhaps it had been harvested since the map was created), and many reference samples happened to be taken within this stand (if the trail went through the stand), then the error for this type of stand would be inflated. If, on the other hand, the same stand had been correctly classified and many reference points were taken within the stand, the error for that stand type would be underestimated. Reference plots taken within a stand or polygon are not independent. To ensure independence, only one reference plot per polygon (with respect to the FRI data) or series of adjacent pixels with the same label (in the case of the Landsat TM data) was used in the classification error matrix. This reference plot was chosen randomly from the set of plots within the polygon by assigning a number to each and using a random digits table (Zar 1999) to select one.

The data were compiled into an error matrix which gives, for each ground-truthed category (rows), the number of reference plots that were assigned to each map land cover class, so that correctly classified plots appear on the diagonal. Three measures of classification accuracy were defined: 1) *overall accuracy*, the trace of the error matrix divided by the total number of reference plots; 2) *producer's accuracy*: for each land cover class, the frequency of correctly classified reference plots divided by the total number of plots in the sample assigned (on the map) to the class in question. This provides an estimate, for a given class, of the probability of a ground-truthed land plot being correctly classified by the map. For example, if the producer's accuracy is 90% for fields, it means that 90% of plots that are fields (on the ground) are correctly identified as fields on the corresponding map; (3) *user's accuracy*: for each land cover class, the frequency of correctly classified reference plots divided by the total number of plots assigned to the class in question based on ground-truthed information. This quantifies the map's ability to correctly identify plots on the ground. For example, if the user's accuracy for hardwood forests is 80%, it means that 80% of the time that the map identifies an area as hardwood forest, it is indeed a hardwood stand.

Statistical comparisons between the FRI and Landsat TM error matrices cannot be made for 2 reasons: First, the reference plots are not independent (i.e. the same plots are used for the accuracy assessments of both maps), nor are they suitable for a paired comparison because reference plots were removed from both maps (see above), but these removed plots were not necessarily the same for both maps. Secondly, the land cover categories differ between the FRI and Landsat TM maps. The classification of the Landsat TM map is of a finer resolution than the 4-class FRI data (for example, the FRI data does not have a category for mixed forest or treed wetland). There is no way to amalgamate the Landsat TM data into the same categories as the FRI data because mixed forests could not be merged with either coniferous or deciduous categories without more information. Thus, differences in overall map accuracy between the FRI and Landsat TM maps may not reflect differences between aerial photographs and satellite imagery, but rather differences due to the classification categories. Thus, the following map accuracy assessment will serve only as a guide for choosing one land cover map for the fisher habitat preference analysis.

Habitat preference analysis

Johnson (1980) defined habitat *use* as the quantity of habitat used by the consumer, and habitat *selection* as the disproportionate use of a habitat relative to its availability. Aebischer et al. (1993) use the term habitat *preference* on a relative scale only, for ranking habitat selection relative to specific, alternate habitats. These definitions will be used here.

Habitat use versus availability was compared at two scales (Johnson 1980, Thomas and Taylor 1990): within the study area (coarse scale) and within the home range (fine scale). Specifically, used and available habitat were defined as (1) the proportion of habitat of a particular type within the home range compared to its proportion within the study area; and (2) the proportion of a habitat type within the defined core area compared to its proportion within the home range. The non-breeding season home ranges of 26 fishers (10 males, 16 females, see Chapter 1) were defined using the 95% kernel estimate (Worton 1989), the core area using the 50% kernel estimate, and the study area using a 100% minimum convex polygon (MCP) around pooled fisher locations (Figure 2.1, see also Figure 1.2 in Chapter 1). The core area was used as the measure of use within the home range, rather than individual locations, because triangulation error (mean 251 m, or 0.198 km^2 confidence circle; see Chapter 3) could result in misclassification and potentially incorrect inferences (Retti and McLoughlin 1999).

Habitat preference was calculated using compositional analysis (Aebischer et al. 1993). If there are D habitat types, an individual's proportional use of these habitats is defined as $x_{u1}, x_{u2}, \ldots, x_{uD}$, where x_{ui} is the proportion of habitat i in an animal's estimated home range. Likewise, the availability of D habitats is defined as $x_{a1}, x_{a2}, \ldots, x_{aD}$, where x_{ai} is the proportion of habitat i that is, at least in principle, available for use. By definition,

$$\sum_{i=1}^{D} x_{ui} = 1 \quad \text{and} \quad \sum_{i=1}^{D} x_{ai} = 1 .$$

Because of this unit-sum constraint, use of one habitat type means avoidance of another, resulting in non-independence of the habitat components. To overcome this constraint, Aitchison (1986) suggested using the log-ratio of use (or availability) for each habitat ($i = 1, \ldots, D, i \neq j$):

$$y_{ij} = \ln \frac{x_i}{x_j},$$

where x_i is the proportion of habitat i and x_j is the proportion of any habitat j in a defined area (whether it be used area or available area). This transformation makes the y_{ij}s linearly independent. An animal's preference for a habitat type will be apparent if there is a difference (d_{ij}) between use (x_{ui}) and availability (x_{ai}) of habitat type i:

$$d_{ij} = \ln\left(\frac{x_{ui}}{x_{uj}}\right) - \ln\left(\frac{x_{ai}}{x_{aj}}\right)$$

or, equivalently

$$d_{ij} = \ln\left(\frac{x_{ui}}{x_{ai}}\right) - \ln\left(\frac{x_{uj}}{x_{aj}}\right). \tag{1}$$

$d_{ij} > 0$ signifies that habitat i is preferred relative to habitat j, and $d_{ij} < 0$ indicates that the animal prefers habitat j over habitat i (Aebischer et al. 1993).

Aebischer et al. (1993) then use multivariate analysis of variance (MANOVA) to test the null hypothesis of no selection. However, because used and available proportions are computed on the same individuals, the two samples are not independent. Hence, I used the paired Hotellings T^2 test statistic (calculated using PAST (PAleontological STatistics) software (Hammer et al. 2001)), rather than MANOVA, to test for significant habitat selection. The data were tested for multivariate normality using PAST software. Rejection of the null hypothesis of no selection leads to a series of Bonferroni-corrected paired t-tests, which can be used to rank habitat types from most preferred to least preferred (Aebischer et al. 1993). For example, if there are 4 habitat types (A, B, C, and D), and D is used as the "reference" habitat, preference of habitat A relative to D is assessed with a paired t-test between the ratios of use and availability for habitats A and D. A preference for habitat A relative to B is assessed with a paired t-test between d_{AD} and d_{BD}; a significant difference with mean $d_{AD} > d_{BD}$ indicates that habitat A is preferred over B.

In the above analysis, if a particular habitat type is available but not used, the log ratio is undefined. Aebischer et al. (1993) and Aitchison (1986) suggest substituting a small, non-zero proportion for zero use (e.g., 0.01) provided that the substitution value is less than the smallest non-zero value in the composition. When a habitat type is neither used nor available, missing values may be substituted with the mean log ratio for that habitat type; however, this can result in ill-defined habitat rankings (Aebischer et al. 1993). Alternatively, they suggest eliminating those animals whose home ranges do not contain any of a particular habitat type (zero availability), while acknowledging that this may generate bias (Aebischer et al. 1993). In the following analysis, when a particular habitat type is neither used nor available, I have substituted a small value (0.01) for both the used and available proportions. This gives a value for d of zero, which contributes nothing to the overall test statistic, and additionally, does not reduce the sample size (had those individuals simply been omitted).

The proportion of each habitat type (coniferous, deciduous, wetland, and field) was calculated in a GIS (ArcMap 8.2, ESRI) by overlaying the 4-class FRI habitat map on the home range or study area. For selection at the fine scale, there were 2 fishers with

0% use of coniferous forest, 6 with 0% use of wetland and 1 with 0% use of field. There were 6 additional fishers for which there was 0% available wetland within the home range. These same 6 individuals also had 0% use of wetland at the coarse scale. In each case, 0.01 was substituted for 0% use and/or availability of a habitat type.

Results

Map accuracy assessment

The results of the reference and map data comparison, using randomly selected points from the 300 ground locations, were compiled into error matrices (Figures 2.2 - 2.4). The FRI map had an overall accuracy of 71% (Figure 2.2). The map correctly identified wetlands, red pine, spruce, other conifers, and fields at least 75% of the time (user's accuracy). It performed moderately well when identifying spruce, other hardwood and white birch (user’s accuracy between 50 and 75%), and poorly when identifying poplar (user’s accuracy 25%). The map misclassified maple as other conifers and white pine as poplar in 21% and 60% of the plots, respectively.

When the FRI land cover classes were reduced to 4 classes, the overall map accuracy increased to 75% (Figure 2.3). The map identified wetlands, coniferous forests and fields well (user’s accuracy > 80%) and deciduous forests only moderately well (user’s accuracy 56%). The map misclassified deciduous forests as coniferous forests and fields 19 and 16% of the time, respectively. The map misclassified coniferous forests as deciduous in 27% of the plots.

The 15-class Landsat TM data performed poorly, with an overall accuracy of 57% (Figure 2.4). The map adequately identified marshes, but identified coniferous forests, mixed forests and open areas only moderately well (user's accuracy between 51 and 75%). It performed poorly in classifying deciduous forests, treed wetlands and sparse forests (user's accuracy ≤50%). The map classified mixed forests as treed wetlands, coniferous forests and open areas 16, 15 and 16% of the time, respectively.

The 28-class Landsat TM data had the lowest overall accuracy of the 4 maps (51%; matrix not shown). The map was moderately accurate with respect to marshes, coniferous plantations, mixed mainly deciduous forests, sparse deciduous forests, pasture, abandoned fields and cropland (user's accuracy between 50-75%), but performed poorly at classifying all other land cover types.

The 4-class FRI data had a higher overall and within-class accuracy than the Landsat TM data, and therefore will be used for fisher habitat selection analyses here.

Habitat preference analysis

Fishers in eastern Ontario showed significant overall non-random habitat selection at the coarse scale ($T^2 = 18.4$, $F = 5.65$, $p = 0.05$) and at the fine scale ($T^2 = 11.1$, $F = 3.40$, $p = 0.03$). Multiple comparison tests showed that at the coarse scale, the order of preference was wetland^{abc} > coniferous^{a} > field^{b} > deciduous^{c} (different letters indicate significant differences at $\alpha = 0.008$; Table 2.1a). However, at the fine scale, multiple comparisons tests did not detect a significant habitat preference (i.e., fishers used habitat in proportion to availability), despite the significant overall selection (Table 2.1b).

Discussion

Map accuracy assessment

Classification error in thematic map accuracy assessment is the discrepancy between the map and reality (Foody 2002, Stehman and Czaplewski 1998). When using thematic maps to draw conclusions about other processes, such as habitat preference, I assume that the thematic map accurately represents the true habitat at any point in space and time. Classification error can affect estimates of habitat preference: if the error is the same for all habitats, the result will be reduced precision of the estimate, and if the error is different across habitats, the result will be reduced accuracy of the estimate. The overall accuracies of 71% (FRI), 75% (4-class FRI), 57% (15-class Landsat TM) and 51% (28-class Landsat TM) fall well below the target set out by Thomlinson et al. (1999) of an overall accuracy of 85%.

The 4-class FRI map did not identify deciduous forests as well as it did coniferous forests, wetlands and fields and, thus, may affect the accuracy of the habitat preference estimates. If the misclassification of deciduous forest is random across space, then it should not affect the estimate of use relative to availability of deciduous forest. If, on the other hand, the distribution of misclassified deciduous forest is not random, then the estimate of use relative to availability could be biased high or low depending on whether the map mistook deciduous forest on the ground for another land cover type (i.e., the amount of deciduous forest is greater than that portrayed by the map) more often in the "available" or in the "used" areas, respectively. If the estimate is biased high, then at the coarse scale, fishers may avoid deciduous forests to a greater degree than observed, and visa versa if the estimate is biased low. A map portraying the locations of correctly and incorrectly mapped areas relative to the kernel home ranges is presented in Appendix 10.

Differences between the reference data and the map could exist for various reasons, as outlined by Congalton (1993). Vegetation present in the lower canopy is usually not detected by remote-sensing technology. Additionally, observer bias when determining the dominant tree type in the field or when categorizing habitat can cause the map to appear less accurate than it is. The time lag (26 years for FRI, 10-15 years for Landsat TM) between when remotely-sensed imagery was collected, and when the resulting habitat polygons were ground-truthed can also affect the perceived accuracy of the maps: fires, forest harvesting, and succession appear as map errors when, in fact, the discrepancy is due to landscape changes.

Habitat preference analysis

Fishers showed overall habitat selection at both coarse (placement of home range within the landscape) and fine (use of core area within the home range) scales. At the coarse scale, fishers prefer coniferous forests > fields > deciduous forests, with no significant preference for wetlands relative to these 3 habitat types, while at the fine scale, I did not detect significant habitat preference. My results are generally consistent with other habitat preference studies which have documented a preference for coniferous forest (Allen 1983, Arthur et al. 1989, Powell 1993, 1994, Thomasma et al. 1991, 1994). Fishers are thought to avoid open areas (Arthur et al. 1989, Powell 1993, 1994, Jones and Garton 1994), and my findings that, at the coarse scale, fishers prefer fields to deciduous forests differ from these studies. Raine (1987) noted that fishers readily crossed open areas in the winter in Manitoba. Arthur et al. (1989) found that fishers avoided wetlands except during the fall, whereas I found no significant preference for or avoidance of wetlands relative to the other habitat types, at either scale. Arthur et al. (1989) and Powell (1994) also found that fishers avoided deciduous forests, consistent with this study.

It is possible that differences in preference at the 2 scales could be an artifact of differences in the variability of available habitat between the 2 scales. For example, if the variability in available habitat is larger at the fine scale than at the coarse scale, there would be more power to detect deviations from zero use at the coarse scale. Unfortunately, differences in the variability of available habitat cannot be assessed because there is no variability at the coarse scale (available habitat is defined as being the same for all individuals). At the fine scale there are 2 sources of between-individual variability (in both used and available habitat), whereas at the coarse scale there is only variability in used habitat. Thus, at the fine scale there will be less power to detect habitat preferences because of the additional variability.

That this study was unable to detect habitat preference at the fine scale differs from Weir and Harestad (2003), who found that fishers selected habitat at 3 spatial scales; this contradiction could be an artifact of the study design. This study used the 50% kernel home range as a measure of use within the 95% kernel home range, rather than using individual locations, and although this measure reduces misclassification of locations when triangulation error is large, it may not be powerful enough to detect significant avoidance of some habitats, such as fields, if that habitat is found within the core area but the animal did not use it directly (see Retti and McLoughlin 1999). This would be especially problematic if habitat types are not randomly distributed throughout the landscape; if fishers prefer coniferous forest, and coniferous forests are more likely to be adjacent to fields than any other land cover type, there will be an apparent selection for fields relative to other habitat types, even if fishers are not using fields directly.

Fishers spend much of their time during the non-breeding season resting, hunting, and capturing prey (Powell 1979). All of these activities occur primarily in forested areas: fishers typically hunt snowshoe hares in coniferous forests and porcupines in deciduous forests (Powell 1993, 1994), and often rest in coniferous trees (Arthur et al. 1989, Powell 1993, 1994). Paragi et al. (1996) found that 94% of 33 natal fisher dens were in hardwood trees. In this study, one natal den was found in a black ash (*Fraxinus niger*) tree (Appendix 11). Thus, the conclusion that habitat selection is non-random (at least at the coarse scale) and fishers prefer forested to non-forested habitat for placement of their home range is not surprising.

Assumptions of compositional analysis

When animals are used as sample units, the animals must be independent. This assumption does not hold for territorial or gregarious species. The fishers in this study are intrasexually territorial (see chapter 1), thus, home range placement of one animal within the study area may be a function of the territorial behaviour of another animal of the same sex in its vicinity, rather than the availability (or lack thereof) of a particular habitat (Aebischer et al. 1993). This could cause the test of selection to be less powerful since some individuals will be forced to use suboptimal habitats. Despite this, the investigation of fisher habitat preference at this scale is still useful in terms of selection of suboptimal habitat, if that is indeed what is occurring (i.e., for population density calculations, see chapter 1). Selection of habitat within the home range, however, is not likely influenced by conspecifics (Aebischer et al. 1993).

Habitat preference analyses that use home ranges or core areas as measures of time spent in particular areas assume that these measures are representative of areas actually used by the animals. I used kernel home ranges as measures of the probability of finding an animal in a particular area. There is no way to address the accuracy of this measure, although simulation studies have shown that kernel estimates reflect actual space use better than other methods of home range estimation (Worton 1995).

Habitat preference studies also assume that time spent in a particular habitat is proportional to the animal's preference for that habitat; that low use relative to availability implies avoidance, rather than that the habitat contains necessary resources that simply do not require a lot of time to obtain (see Mysterud and Ims 1998, Cooper and Millspaugh 2001). For example, Powell (1994) found that fishers used deciduous forests less than expected, but noted that fishers spent a large proportion of their time there, indicating that deciduous forests are still important for fishers.

Study limitations

This study did not compare habitat selection across seasons or between diurnal and nocturnal behaviour due to insufficient data for these different classes, or between sexes. Pooling data across seasons, time periods and sex could mask potential differences in selection (Schooley 1994, see Retti and McLoughlin 1999). Another weakness of the habitat selection analysis is that it does not consider interspersion or juxtaposition of habitats (Porter and Church 1987, Retti and McLoughlin 1999), which may also be important for fishers.

The definition of available habitat for the landscape level comparison can influence the results of habitat selection studies. McClean et al. (1998) suggest using the entire study area as the measure of habitat availability, rather than buffers around individual locations. Katnik and Wielgus (2005) suggested using the composition of random home ranges as estimates of available habitat rather than the entire study area when using compositional analysis. Porter and Church (1987) found, through simulation, that when the study area boundaries were varied, conclusions of habitat selection were greatly influenced if the habitat types were aggregated, but had little effect when the habitat distribution was more random. The effect of varying the study area boundary on habitat selectivity was not assessed in this study. Since the land cover patches are small and do not appear aggregated, this would not likely alter the results substantially.

Recently, Bingham and Brennan (2004) used simulations to compare type I error rates for 3 methods of analysing habitat use versus availability data, including compositional analysis. They found that substituting small, non-zero values for 0% use of particular habitats in compositional analysis inflated type I error, causing incorrect rejection of the null hypothesis in up to 100% of the simulated trials. They contend that substituting small non-zero values for 0% use when availability is also small will incorrectly indicate significant avoidance of the habitat. As long as all habitats have reasonable availability, as in this study (Table 1.1, Chapter 1), this should not affect the results.

Given the assumptions and limitations of compositional analysis, it can still be concluded that fishers prefer coniferous forests > fields > deciduous forests when choosing a home range, however, I was unable to detect habitat preferences within the home range. This information is useful when using home ranges to estimate population density because it identifies suitable fisher habitat. Additionally, habitat preference data for fishers provides fur managers with information that can be used if managing fisher populations indirectly by managing forests becomes necessary.

Tables and Figures

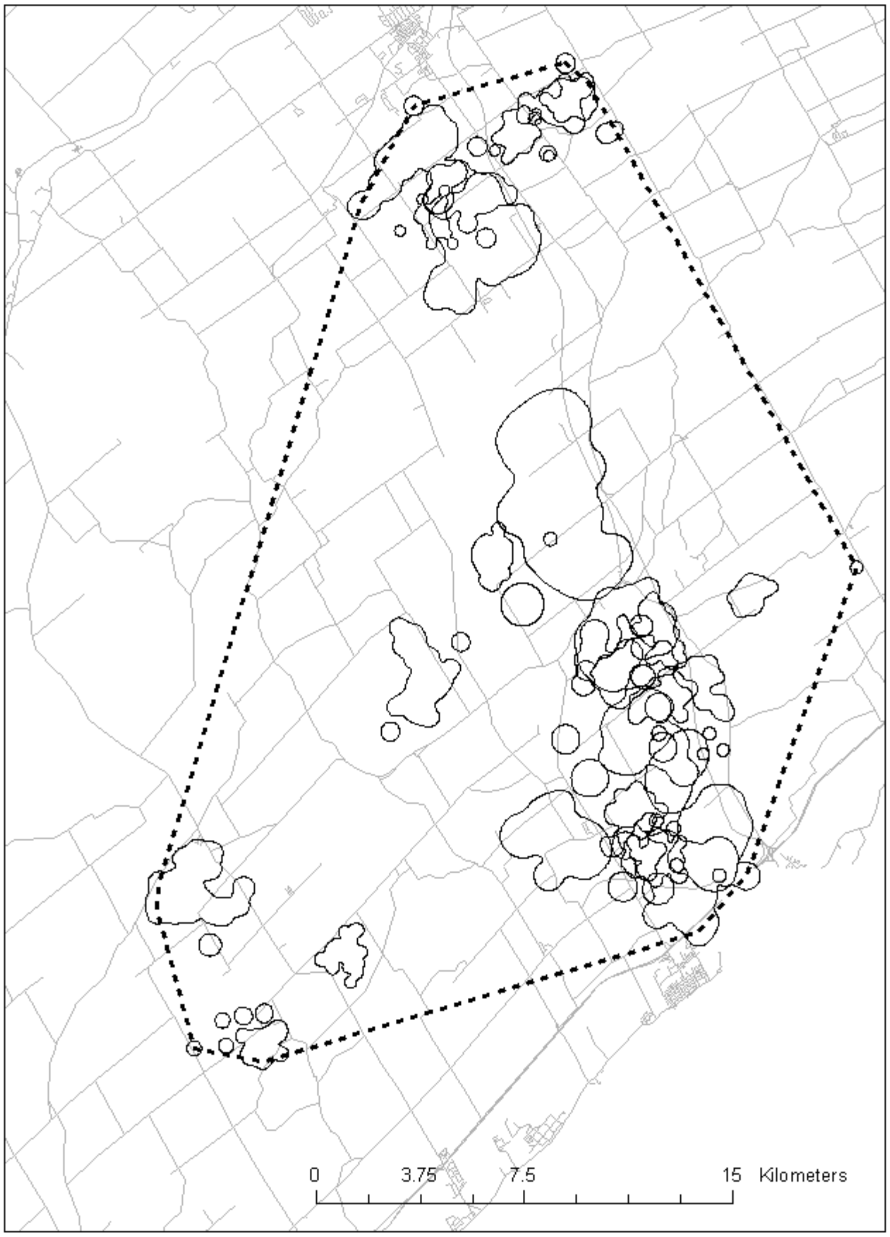

Figure 2.1. 100% minimum convex polygon (dashed line) of all fisher locations, defining available habitat within the study area. Solid lines represent 95% kernel home ranges of 26 fishers (10 male, 16 female), denoting used habitat.

Ground-truthed land cover classification

FRI map land cover classification

	Wet.	WP	RP	JP	S	OC	M	YB	OH	P	WB	Field	**Total**
Wet.	10												**10**
WP													**0**
RP			8										**8**
JP						1							**1**
S				3					1				**4**
OC	1	1				32	3		3		1	1	**42**
M		1	1	1		2	9	1	2			1	**18**
YB													**0**
OH	1					3			7			1	**12**
P		3				5	1			4	1	2	**16**
WB						1			1		2		**4**
Field	3					1	1		3	1	2	48	**59**
Total	**15**	**5**	**9**	**4**	**0**	**45**	**14**	**1**	**17**	**5**	**6**	**53**	**174**

Figure 2.2. Classification error matrix for the Forest Resource Inventory (FRI) 1978 land cover map. Cell entries are counts of the number of ground-truthed reference plots that were classified as a particular land cover type in the FRI map. Wet = wetland, WP = white pine, RP = red pine, JP = jack pine, S = spruce (all), OC = other conifer (mixture of coniferous), M = maple (all), YB = yellow birch, OH = other hardwood (mixture of hardwood), P = poplar, WB = white birch, field = non-forest.

Ground-truthed land cover classification

FRI map land cover classification

	Wet.	Conif.	Decid.	Field	**Total**
Wet.	10				**10**
Conif.	1	45	8	1	**55**
Decid.	1	17	28	4	**50**
Field	3	1	7	48	**59**
Total	**15**	**63**	**43**	**53**	**174**

Figure 2.3. Classification error matrix for the accuracy assessment of the Forest Resource Inventory (FRI) 1978 land cover map (4-class). Cell entries are counts of the number of ground-truthed reference plots that were classified as a particular land cover type in the FRI map. Wet = wetland, Conif = coniferous (white pine, red pine, jack pine, spruce, other conifer), Decid = deciduous (maple, yellow birch, other hardwood, poplar, white birch), field = non-forest.

Ground-truthed land cover classification

Landsat TM land cover classification

	Marsh	Open wet.	Treed wet.	Decid.	Conif.	Mixed	Sparse	Open	**Total**
Marsh	2								**2**
Open wet.			1				1	1	**3**
Treed wet.			17	3	4	12		4	**40**
Decid.			3	15	2	5	1	5	**31**
Conif.			3	1	21	11	2	1	**39**
Mixed			1	4	3	34	1	4	**47**
Sparse							2	2	**4**
Open			3	2	2	12	1	31	**51**
Total	**2**	**0**	**28**	**25**	**32**	**74**	**8**	**48**	**217**

Figure 2.4. Classification error matrix for the accuracy assessment of the Landsat TM 15 class land cover map (early 1990s). Cell entries are counts of the number of ground-truthed reference plots that were classified as a particular land cover type in the Landsat TM map. Open wet. = open wetland, Treed wet. = treed wetland, Decid. = deciduous forest, Conif. = coniferous forest, Mixed = mixed deciduous and coniferous forest, Sparse = mixed forest with 30-40% canopy closure, Open = field, pasture or cropland.

Table 2.1. Matrix of t and (p) values from paired t-tests, using 4-class FRI data. 0.01 is substituted for zero use and availability, and data is pooled over sex. Selection is measured as (a) use of the home range relative to the study area (coarse scale) and (b) use of the core area relative to the home range (fine scale). Bonferroni-corrected α = 0.0083. A rank of 1 is most preferred and 4 is least preferred.

a) Coarse scale

	Coniferous	Deciduous	Wetland	Field	Rank
Coniferous		5.67 (>0.001) *	0.774 (0.446)	4.27 (>0.001) *	2
Deciduous			-2.63 (0.014)	-4.13 (>0.001)	4
Wetland				-1.69 (0.104) *	1
Field					3

b) Fine scale

	Coniferous	Deciduous	Wetland	Field	Rank
Coniferous		-1.03 (0.314)	0.666 (0.511) *	1.08 (0.292) *	2
Deciduous			-1.67 (0.108) *	2.76 (0.011) *	1
Wetland				-0.005 (0.996)	4
Field					3

* indicates that the row habitat is selected relative to the column habitat (and visa versa when there is an absence of *)

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Chapter 3: Estimating triangulation error

Introduction

Triangulation error in radio-telemetry is the distance between the actual location of the source of the signal and the location determined by triangulation (Figure 3.1). The estimation of triangulation error is important since it indicates the confidence that we have in the data when using radiolocations for other applications, such as home range and habitat preference assessments (Kauhala and Tiilikainen 2002). It is a topic that has been considered in detail in the methodological literature, yet it is not often measured in applied radio-telemetry studies (Springer 1979, Harris et al. 1990, Saltz 1994).

Triangulation error is the result of angle error, the distance between the transmitter and receiver, the intersection angle of the bearings, and movement of the signal (Springer 1979, Lee et al. 1985, White and Garrott 1990). Angle error is the angular difference between the observed and true bearings (Figure 3.1). The discrepancy between observed and true bearings can be due to physical obstructions such as buildings, hills, and dense vegetation that cause the signal to refract and/or reflect (Heezen and Tester 1967, Springer 1979, Lee et al. 1985), or observer error. The latter can occur when determining the direction of the signal, reading the compass, or recording bearings and coordinates (Saltz and Alkon 1985). For moving transmitters, the observer may receive a modulating signal, and this fluctuation in signal intensity will generally increase observer error (Lee et al. 1985). Properties of the triangulation, such as the distance between the transmitter and the receiver and the angle of intersection of the bearings, also contribute to the accuracy of estimated signal locations (Heezen and Tester 1967, Springer 1979, White 1985, Zimmerman and Powell 1995): errors are minimized when the signal is close to the receiver (Heezen and Tester 1967, White and Garrott 1990) and angles of bearing intersections are between 90 and 100° (Zimmerman and Powell 1995). Finally, time delays between sequential bearings will result in a magnified location error if the signal is moving (Heezen and Tester 1967, Schmutz and White 1990, Lee et al. 1985).

Triangulation accuracy

Because transmitter locations are estimated based on observed bearings, which rarely intersect, researchers have developed methods of deriving the most likely location of the transmitter. Two commonly used methods are the calculation of the geometric mean of the bearing intersections (GM), and maximum likelihood estimation (MLE) (Lenth 1981). MLE finds the (x,y) coordinates that maximize the probability of observing the set of recorded bearings. It uses the Von Mises distribution, a unimodal, circular distribution akin to the normal distribution of linear statistics (Batschelet 1981). The data are assumed to fit this distribution, with the concentration parameter, k, being the same for all bearings and calculated using the standard deviation of the angle error (see White and Garrott 1990), and each observed bearing being the mean of its distribution. Regardless of which method is used to estimate the location, the accuracy of the estimated location is calculated by measuring the linear distance between true and estimated locations from a set of test data that are, at least in principle, representative of actual data. This distance is referred to as the location error (Figure 3.1).

Triangulation precision

The precision of an estimated location is represented by an area around that location. With 2 bearings, error polygons are used as error estimators (Heezen and Tester 1967, Nams and Boutin 1991). When there are more than 2 bearings, MLE confidence ellipses are often used (Lenth 1981). The mean location error can also be used as an estimate of precision (Zimmerman and Powell 1995).

Error polygons are formed by the intersection of error arcs around 2 bearings. The calculation of error arcs is derived from the angle error, or the difference between the estimated and true bearings (see Figure 3.1). A mean angle error of 0 represents an unbiased telemetry system (Springer 1979, White and Garrott 1990). The standard deviation (SD) of the mean angle error for a sample of bearings is used to estimate 95% confidence error arcs (Springer 1979, Lee et al. 1985). The polygon created when the 95% confidence error arcs of 2 bearings intersect is the area that has a 90% probability of containing the true location (Springer 1979). Attributes of this error polygon, such as its area (White and Garrott 1990) or the length of the longest diagonal (Saltz and Alkon 1985), can then be used as estimates of triangulation precision.

The MLE 95% confidence ellipse represents the area in which the true location should be located 95% of the time. Both the area of the confidence ellipse and the lengths of different ellipse axes (Saltz and White 1990) may be used as measures of triangulation precision. The standard deviation of the angle error derived from a set of test data is the value of the concentration parameter of the MLE when estimating locations for actual data (Lenth 1981, White and Garrott 1990). The size and shape of the confidence ellipse is a function not only of angle error, but also the angle of intersection of the bearings and the distance between the transmitter and receiver (White and Garrott 1990, Saltz 1994).

Zimmerman and Powell (1995) recommended using location error, or the distance between estimated and true locations, as a measure of both triangulation accuracy and precision. Location error is calculated for a set of test data and the 95% confidence distance of the mean location error can be used as the radius of a circle around the estimated point. Thus, location error can be used as a measure of precision, representing the area that has a 95% probability of containing the true location. Since this measure can be influenced by the distance between receiver and estimated transmitter location, Zimmerman and Powell (1995) suggested regressing location error on distance with test data and then using the resulting regression equation to correct for the effect of distance on the actual data, giving a unique confidence circle for each location.

Triangulation error is often calculated by placing transmitters in the study area and calculating the accuracy and precision of triangulation estimates. Statistics calculated for the test data are then applied to the actual data, for which the true location of the transmitter is unknown. This chapter investigates the radio-telemetry error of this study, defined as the distance between the actual and estimated location of a transmitter, using a stationary radio-transmitter located by both aerial telemetry and triangulation from the ground. The average bias in the bearings was determined and corrected for mathematically. The accuracy of triangulations, measured by location error, was compared between 2 methods of estimating locations: the MLE and the GM. The ability of MLE confidence ellipses and location error confidence circles to predict triangulation precision was tested using a second data set. Triangulation error of a moving transmitter was also calculated and compared to that of a stationary transmitter. These values are assumed to be representative of the accuracy and precision of the entire radio-telemetry study.

Methods

The data

Test data A includes a large set of triangulation data for fixed stationary transmitters at known locations. Radio-collars were placed in 35 locations throughout the study area in forests that fishers are known to inhabit. Investigators located these collars using triangulation as per the protocol in chapter 1, with a mobile handheld 2-element yagi antenna. Each of the 35 collars was located by triangulation an average of 2.4 times (±1.2 SD) for a total of 83 estimated locations. This procedure was conducted at various times throughout the study so that changes in error as investigators gained more experience would be included in the estimate. In 75% of the trials, the person taking the bearings did not know the true collar location prior to triangulation. This situation occurred when the collar was placed in the forest by a different investigator or when a radio-collared fisher died and the signal was used for error testing prior to retrieving the collar. In the remaining trials, the investigator knew the location of the transmitter. In all cases, the investigator knew that error was being evaluated.

Test data B is a smaller set of triangulation data collected near the end of the study. During the estimation of error from moving radio-collars (see below) 19 stationary radio-collars were placed in the forest. The location of the stationary collars was unknown in all cases, although the investigator was aware that error was being evaluated.

Triangulation accuracy

Test data A were used to compare the accuracy and precision of 2 different methods of estimating transmitter locations. The raw triangulation data were first examined with Location of a Signal (LOAS; Ecological Software Solutions 3.0.2) software that calculates locations from triangulations, without plotting the true locations. Bearings were excluded from the triangulation if they were identified in the field as being questionable or if they did not intersect other bearings in the vicinity of other intersections (Garrott et al. 1986, White and Garrott 1990). Questionable bearings invariably had weak signal strength, the signal was modulating, or there was signal interference. Non-intersecting bearings or bearings that intersected far from other intersections were usually due to a large distance between the receiver and radio-collar or the angle between two intersecting bearings was small (Figure 3.2). Angle error was calculated for each bearing with LOAS software by calculating the angular distance between observed and true bearings.

Locations of the transmitters in test data A were estimated by the GM and MLE using LOAS software. The number of bearings per location was between 2 and 8 (mean 3.7 ± 1.3 SD, n = 83). Each estimation method generated a set of estimated locations, with estimation error for each location defined as the absolute difference between estimated and true locations. This yielded a set of location errors, one for each method, which could then be compared. Location error was regressed against the average distance between the receiver and transmitter to assess the effect of distance on location error for each estimation method.

Triangulation precision

Maximum likelihood confidence ellipses were calculated for each location in test data A with LOAS software. The relationships between location error and both the attributes of the confidence ellipse (area and lengths of the major and minor axes) and mean angle error were assessed with regression analyses.

Location error was used to create confidence circles around location estimates. Because location error increases with distance between the transmitter and receiver, distance was incorporated into the calculation of confidence circle radii as follows, so that each location has a uniquely sized confidence circle based on its distance from the receiver. First, location error was regressed against the distance between the transmitter and receiver for test data A. The resulting relationship was used to predict location error for each location in test data B, given the distance between transmitter and receiver. Finally, the radius of the confidence circle was calculated using the 95% confidence interval of location error for test data A:

$$radius = predicted\ location\ error + 95\%\ CI\ \ for\ exponential\ distribution, \qquad (1)$$

where the predicted location error is calculated by substituting distance into the regression equation, and the 95% CI is derived from the table of critical values for the exponential distribution (Appendix 12). The exponential distribution was used rather than the normal distribution since I expect most location errors to be close to zero, with no values less than zero.

Error polygons as precision estimates were not assessed in this study. Nams and Boutin (1991) showed that error polygons cannot accurately be created for >2 bearings. The error arcs from >2 bearings are not independent since, if the third arc does not intersect that of the first 2, the third arc is less likely to contain the true location than if all 3 intersected. Because of this lack of bearing independence, the size of the resulting error polygon does not correlate with the precision of the estimate (Nams and Boutin 1991). The data for this study use >2 bearings for most of the locations. Using more than 2 bearings is advantageous because it allows one to censor erroneous bearings, which are difficult to identify when there are only 2 bearings (White and Garrott 1990). Thus, only those precision estimation methods that can accommodate >2 bearings, MLE confidence ellipses and location error confidence circles, were considered in this study.

Attributes of the precision estimates from test data A were applied to test data B to assess the ability of the estimators to predict precision. MLE confidence ellipses were calculated for each location in test data B using the standard deviation of the angle error from test data A as the concentration parameter. Likewise, confidence circles were created for each location in test data B using confidence distances from the regression of location error and distance in test data A (see above) as the radii of the circles. The ability of these estimators to predict precision was tested by comparing the number of locations that actually fell within confidence ellipses and circles at different confidence levels between 95% and 50%, to the expected number of locations (Zimmerman and Powell 1995).

Moving transmitter

Triangulation error associated with a moving radio-collar was compared to that of a stationary collar. Volunteers placed one radio-collar in the forest to imitate a resting fisher, and walked through the forest carrying another collar to mimic a moving fisher. The volunteers walked along trails at an average of 0.6 m/s (± 0.19 SD) for 20 minutes in fairly straight lines, through a variety of habitats that fishers were known to occupy. Every minute volunteers recorded their location with a GPS (Garmin GPSmap 76s, Garmin International Inc.), which had a spatial accuracy between 3 and 15 m. Meanwhile, the investigator located both the stationary and moving collars using triangulation as per the protocol in chapter 1. The investigator was unaware of which collar was moving and which was stationary, and which route the volunteers carrying the moving collar followed. The investigator was, however, aware that error was being evaluated. One location for both the moving and stationary collar was obtained for each trail that the volunteers walked. An average of 3.6 (± 0.75 SD, n = 20) bearings per transmitter were taken sequentially over a period of less than 20 minutes.

Locations of both stationary and moving transmitters were estimated using both MLE and GM. Location error of the stationary collars was calculated as the distance between estimated and true locations, while the accuracy associated with moving collars was calculated as the distance between the estimated location and the true transmitter location, averaged over the set of all sequential bearings (Figure 3.3). General linear models of location error in relation to distance (between transmitter and receiver) and transmitter type (stationary or moving) were used to identify a potential interaction between distance and transmitter type. Location error was then compared between moving and stationary transmitters for both the MLE and GM with a repeated measures (RM) ANOVA, with method (MLE or GM) as the repeated factor.

Aerial telemetry

Location error during radiolocation from the airplane was measured as the distance between the estimated and true locations. The location error of the estimate obtained from the air was compared to the location error from ground triangulations of the same transmitter using a paired t-test. The method of signal location from the plane is described in chapter 1. The radio-collars were placed in the study area or were on a dead fisher not yet retrieved. The investigator in the plane did not know the actual location of the transmitter or that he was being monitored for accuracy. One investigator collected all of the aerial data.

Results

Angle error

The average angle error of 6.4° (± 14.5 SD) for test data A was significantly different from 0 ($t = 7.812$, $df = 314$, $p<0.001$), and represents angle bias. This bias was corrected for in all subsequent calculations by subtracting this value from each bearing. Angle error did not vary with the distance between the receiver and the estimated transmitter location (Figure 3.4; $F = 0.391$, $df = 1$, $p = 0.532$, $r^2 = 0.001$, $n = 315$), nor was it correlated with location error (Figure 3.5; $p = 0.680$, $r = 0.225$, $n = 276$). Because angle error was not normally distributed (Figure 3.6; Lilliefors $p < 0.001$), 95% CI of observations was calculated by ranking the data and then finding the interval which included 95% of the observations in the sample; between -21.1 and 30.8°; an interval of 51.9° (n = 315). Thus, error arcs were calculated as the observed bearing ± 25.9°, which is slightly smaller than the calculation of error arcs using the observed bearing ± 1.96*SD, which assumes a normal distribution of error angles (± 28.4°). Although error polygons were not used in this analysis, error arcs were nonetheless calculated to allow comparison with values reported in the literature.

Triangulation accuracy

Descriptive statistics of the location error for test data A are presented in Table 3.1. The average distance between the receiver and estimated transmitter location was 1014 m (± 446.2 SD, n = 66). The effect of distance between transmitter and receiver and method of location estimation (MLE or GM) on location error was assessed with an ANCOVA. There was a significant positive linear relationship between the location error (cube root transformed) and the average distance between the receiver and estimated transmitter location (Figure 3.7; $F = 53.732$, $df = 1$, $p < 0.001$, $r^2 = 0.276$). There was no difference in location error between the MLE and the GM ($F = 3.368$, $df = 1$, $p = 0.068$). The effect of distance on location error was the same for both methods (interaction: $F = 1.575$, $df = 1$, $p = 0.117$).

The effect of number of bearings on the location error for both methods (MLE or GM) was assessed with a 2-way factorial design ANOVA. Mean location error (cube root transformed) did not depend on the number of bearings used to produce location estimates ($F = 1.743$, $df = 1$, $p = 0.189$), nor did the effect of the number of bearings depend on the method (interaction: $F = 0.286$, $df = 1$, $p = 0.593$). However, with the exception of 7 and 8 bearings, the precision of the estimates appears to increase (standard deviation decreased) with more bearings (Figure 3.8).

Triangulation precision

Confidence ellipses

The mean area of the confidence ellipse for test data A was 0.99 km^2 (± 1.46 SD), the mean length of the major axis was 770.6 m (± 799.6 SD) and the mean length of the minor axis was 308.1 m (± 147.4 SD) ($n = 66$). There were significant positive linear relationships between the log of location error and the log of the area of the confidence ellipse (Figure 3.9a; $F = 54.591$, $df = 1$, $p < 0.001$, $r^2 = 0.460$, $n = 66$), the log of the length of the major axis (Figure 3.9b; $F = 50.966$, $df = 1$, $p < 0.001$, $r^2 = 0.443$, $n = 66$), and the log of the length of the minor axis (Figure 3.9c; $F = 42.782$, $df = 1$, $p < 0.001$, $r^2 = 0.401$, $n = 66$).

Confidence circles

Ninety-five percent of the location error values for the MLE derived locations were less than 600 m ($n = 66$). A regression of location error on average distance between the transmitter and receiver for test data A was positive and significant ($t = 6.947$, $p < 0.001$, $r^2 = 0.434$). The resulting regression equation was

$$\sqrt[3]{location\ error} = 3.360 + (0.002 * distance)\,. \tag{2}$$

This relationship was used to predict location error for test data B, given the distance between transmitter and receiver. The slope of the regression between observed and predicted location error (1.56) was not significantly different from 1 (Figure 3.10; t =

0.925, df = 14, $0.2 < p < 0.5$). The radius of confidence circles for each location in test data B was calculated using equation (1), where predicted linear error is calculated by substituting distance into equation (2) and the 95% CI is the product of the standard deviation of mean linear error from test data A and the critical values are from the exponential distribution, since location error is distributed exponentially rather than normally (Figure 3.11). The radius was used to create a circle around the point location for test data B and represents precision.

Confidence ellipses and circles were calculated for test data B with varying levels of precision; between 50 and 95%. Then, the proportion of actual locations that fell within these confidence areas was compared to what was expected, i.e. a 95% confidence ellipse or confidence circle is expected to contain the actual location 95% of the time. The slope of the line comparing the observed number of locations falling in the x% confidence area to x was significantly different from 1 for both the confidence ellipse (Figure 3.12; slope = 0.652, $t = 6.566$, $p < 0.001$) and the confidence circles (Figure 3.12; slope = 0.859, $t = 2.82$, $0.02 < p < 0.05$). The mean absolute deviation from the line with a slope of 1 passing through the origin was 5% (± 2.6 SD) for confidence ellipses and 11% (± 3.0 SD) for the confidence circles.

Moving transmitter

The mean distance between transmitter and receiver for test data B was 580.2 m (± 317.1 SD, n = 20) for moving and 526 m (± 354.2 SD, n = 20) for stationary transmitters. The relationship between location error and distance was independent of transmitter type for both the MLE (Figure 3.13; interaction term: $F = 2.194$, df = 1, $p =$

0.148, n = 19) and the GM (Figure 3.14; interaction term: F = 0.004, df = 1, p = 0.949, n = 19). A RM ANOVA compared the log transformed location error between MLE and GM derived locations in test data B for both stationary and moving transmitters (Table 3.2). The mean location error for moving transmitters was significantly larger than for stationary transmitters (F = 8.343, df = 1, p = 0.007, n = 19). There was no difference in location error between the MLE and the GM (F = 0.000, df = 1, p = 0.998, n = 19), for both stationary and moving transmitters (interaction term: F = 0.262, df = 1, p = 0.612, n = 19).

Aerial telemetry

Location error from the air was significantly larger than location error from the ground (Table 3.3; t = 2.550, df = 6, p = 0.043).

Discussion

Angle error

Angle error ranged from -86.6 – 77.2°, comparable to Zimmerman and Powell (1995) (range -34 – 71°) but was higher than that of Mooty et al. (1987) (absolute angle error 0 - 40°). Although angle error did not depend on the distance between the receiver and estimated location (as also reported by Springer (1979) and Zimmerman and Powell (1995)), angle error precision was smaller (less precise) at shorter distances (see also Zimmerman and Powell 1995). This could be because, when close to the transmitter, the arc of audible signal is wider, which reduces an observer’s ability to identify the loudest signal (Zimmerman and Powell 1995). The 95% error arc of ± 25.9° in this study is larger than those reported in other studies using mobile, hand held antennas (± 11.96° (Edge and Marcum 1985), ± 16° (Zimmerman and Powell 1995) and less than ± 10° (Lee et al. 1985)). This difference might be due to observer inexperience in this study, or perhaps there are more physical obstructions or signal interference in this agricultural area.

Triangulation accuracy

I found no significant difference in mean location error between the GM and MLE estimation, consistent with Zimmerman and Powell (1995). MLE location error (251 m ± 242 SD, n = 67) in this study is greater than that reported in the literature (Table 3.4). Again, this could be due to observer inexperience or landscape differences. Location error increased linearly with increasing distance between the receiver and estimated transmitter location. This trend was also found by Zimmermann and Powell (1995)

Triangulation precision

The confidence ellipse parameters in this study indicate that the triangulations are less precise than that reported in Zimmerman and Powell (1995); the area of the confidence ellipse is almost twice as large in this study. Confidence ellipse size was positively correlated with location error in this study, indicating that ellipse size can be used to identify inaccurate triangulations.

Ninety-five percent of the location error values were less than 600 m in this study, which is similar to Zimmerman and Powell (1995) with a corresponding value of 766 m.

Confidence circles were consistently larger than expected; they contained the actual location more often than they should. Confidence ellipses, however, were larger than expected for CI > 75% and smaller than expected for CI < 75% (Figure 3.12). One hundred percent of the actual locations fall within 95% confidence circles, whereas 88% of actual locations fall within 95% confidence ellipses. Other studies have also reported that 95% MLE confidence ellipses contain as few as 70% (Zimmerman and Powell 1995), 60% (Withey et al. 2001) and 40% (Garrott et al. 1986) of the actual locations.

Moving transmitter

Moving transmitters have potentially larger location errors than stationary signals (Heezen and Tester 1967). Schmutz and White (1990) used simulations to estimate the error associated with triangulating a moving animal when bearings were taken sequentially rather than simultaneously. They found that when the signal was close to the receiver, the error associated with triangulation was up to 10-fold greater when the animal was moving. Laundre and Keller (1981) used sequential bearings to estimate point locations and found that when the signal was moving at a mean rate of 0.94 m/s, the location error was 140 m, whereas the location error for a stationary collar was 50 m. There was a significant difference in location error between the stationary and moving transmitters in this study. Although there was no significant difference in location error between location estimation methods, the location error was smaller for the MLE than the GM, which indicates that the MLE performs better than the GM, especially when the transmitter is moving.

Aerial telemetry error

The mean telemetry error of 729.4 m from the airplane was significantly larger than the corresponding ground triangulation error. Lesage et al. (2000) reported an aerial telemetry error of 392 m and a ground triangulation error of 170 m. Conversely, Arthur and Krohn (1991) found that aerial error was less than ground triangulation error (10 m vs. <150 m, respectively). Very few (<1.5 %) of the actual point locations were collected from the air in this study, thus, this large location error will be applied to very few locations.

Conclusions and applications

The actual triangulation data will be analysed by eliminating potentially erroneous bearings and estimating point locations using MLE. Since the MLE cannot calculate point locations if there are only 2 bearings, the GM will be used for the bearing groups for which the MLE fails to generate a location (18% of the cases in test data A). Triangulation error for the entire set of actual data will be assumed to be, on average, 251 m (± 242 SD). Since confidence ellipse size correlated with location error, confidence ellipse area will be used as an indicator of accuracy to omit locations in the actual data that could have a large location error (cf. Saltz and Alkon 1985, White and Garrott 1990). Because confidence circles are more conservative than confidence ellipses, confidence circles based on the location error method will be used to determine overall triangulation precision; 95% confidence circles will be assumed to contain the actual location 100% of the time.

Tables and Figures

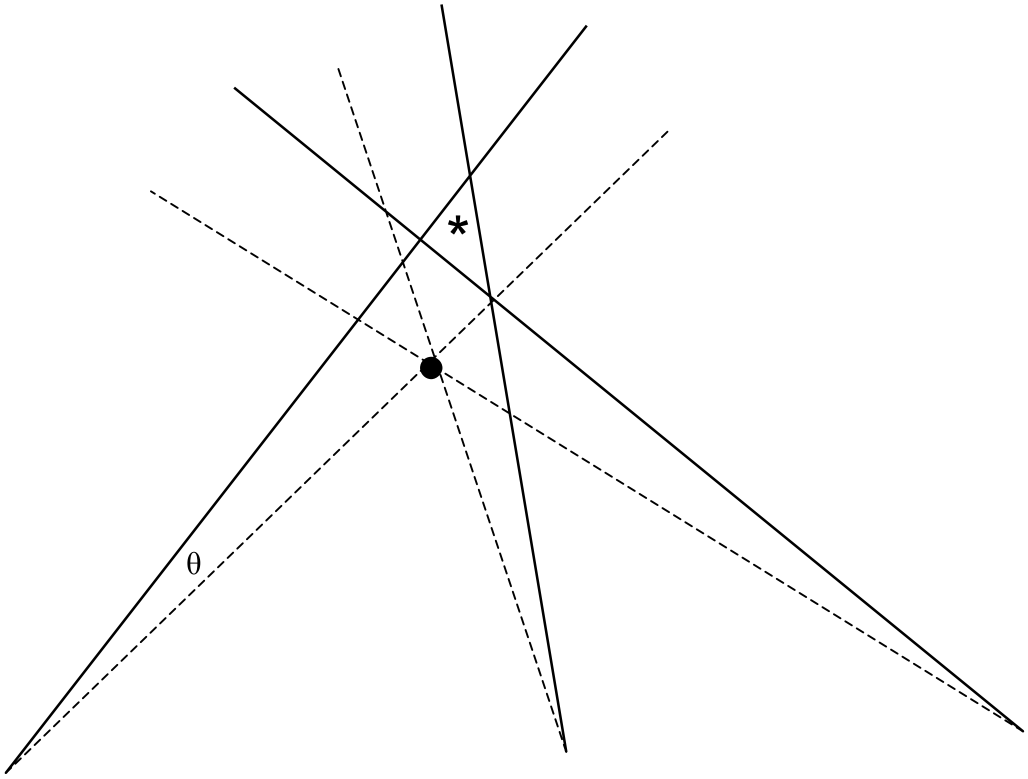


Figure 3.1. Triangulation error for a stationary transmitter. The dot represents the true location of the transmitter. The dashed lines are the actual bearings while the solid lines are the estimated bearings. The star represents the estimated location. The distance between the star and the dot is the triangulation, or location error. θ is the angle error.

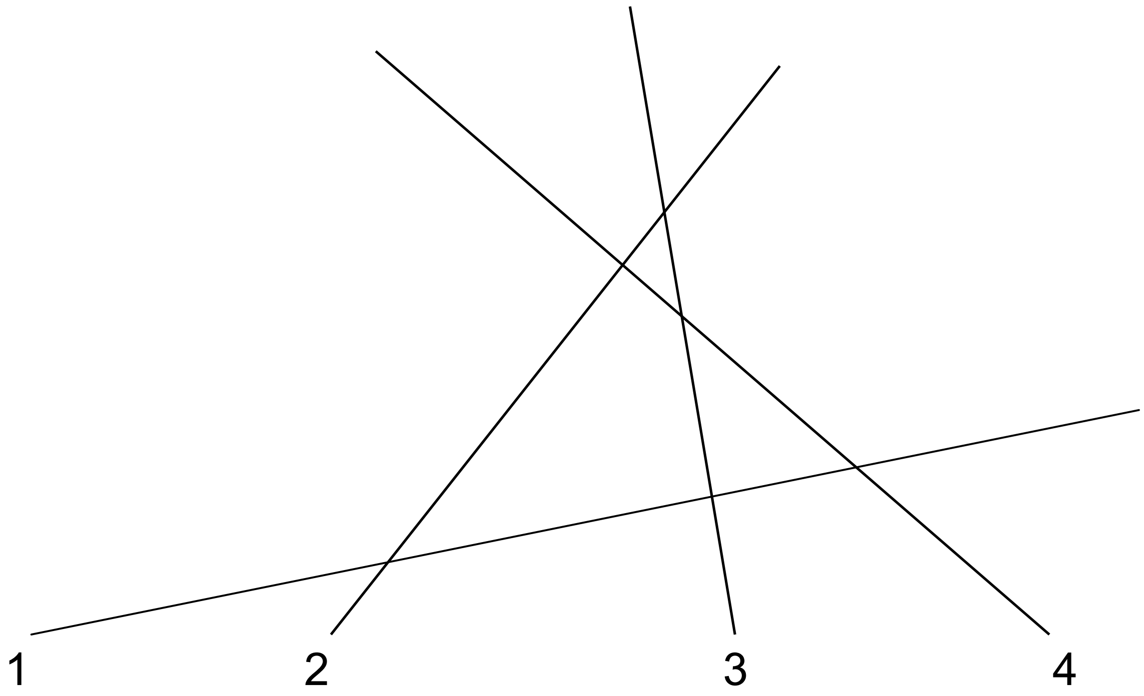


Figure 3.2. Demonstration of a potentially erroneous bearing when assessing triangulation error. The intersections of bearing 1 with bearings 2, 3 and 4 are far from the intersections of all other bearings. Thus, bearing 1 is omitted from the triangulation.

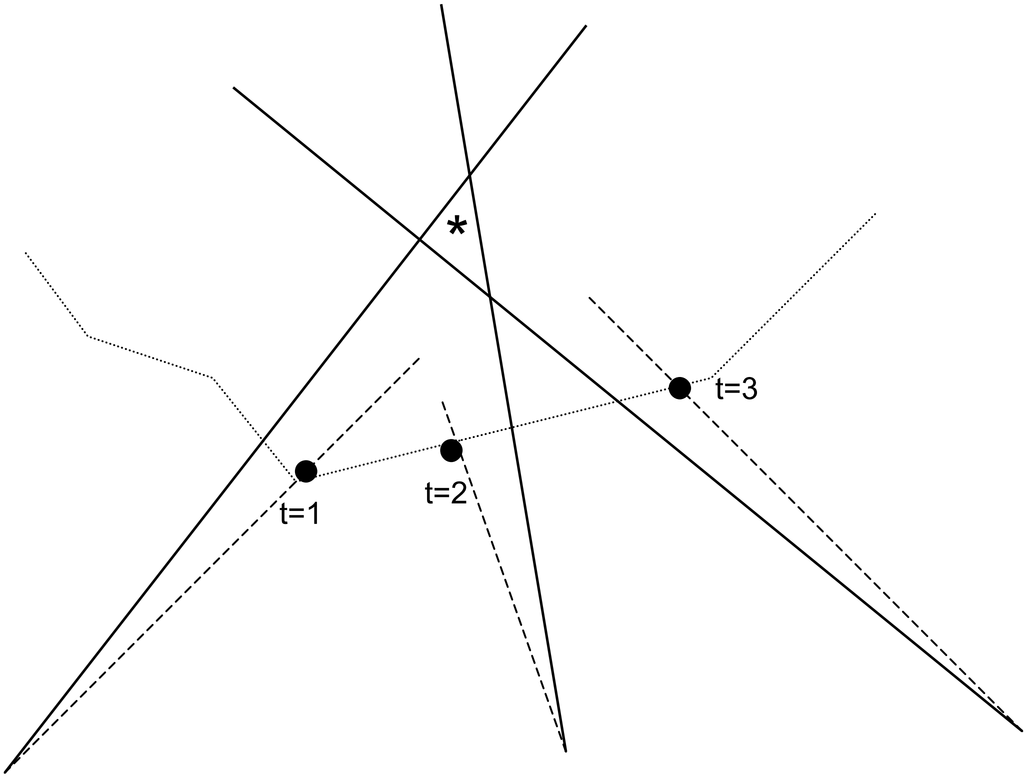


Figure 3.3. Location error when the transmitter is moving. The dotted line represents the path that the radio-collared animal took. The dots represent the true location of the transmitter at times t = 1, 2 and 3. The solid line is the estimated bearing at time t = 1, 2 and 3, and the dashed line is the true bearing to the transmitter at each time. The star is the estimated location based on the 3 observed bearings. Location error is the average distance between the star and the dot at t = 1, 2 and 3.

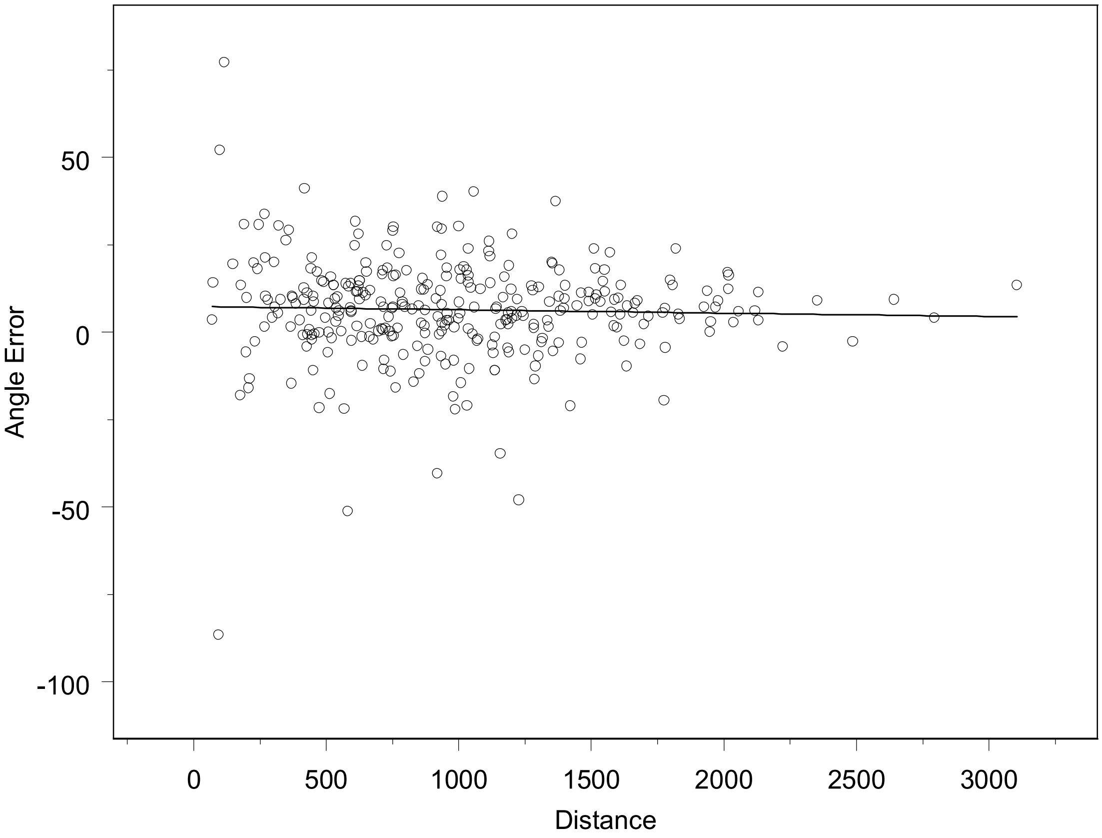


Figure 3.4. The relationship between angle error (°) and the distance between the receiver and the estimated transmitter location (m) for test data A (n = 315).

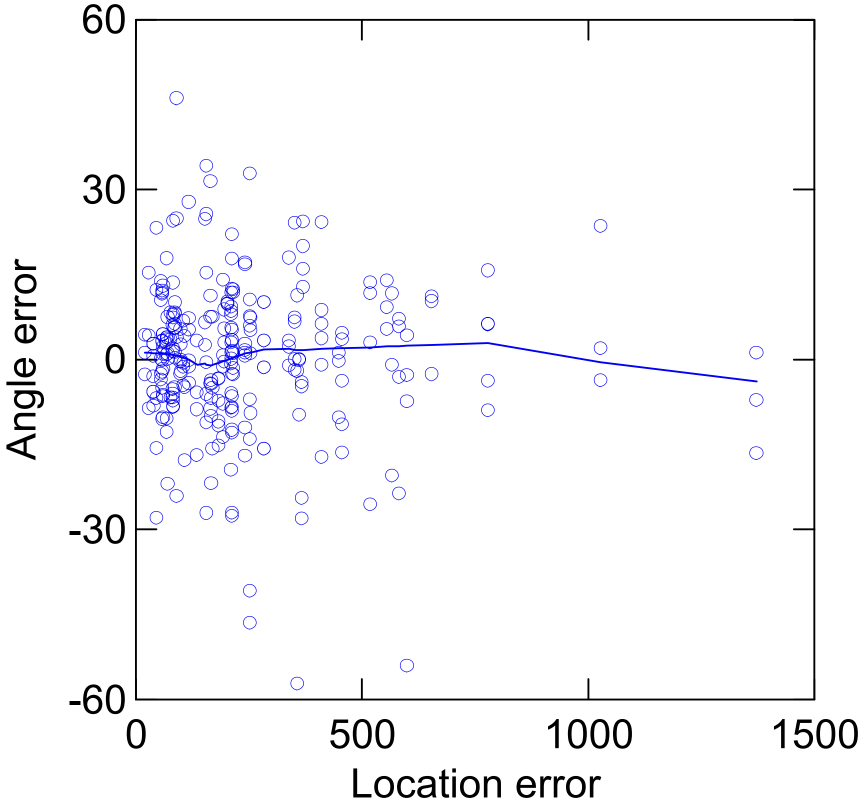


Figure 3.5. The relationship between angle error (°) and location error (m) for test data A (n = 276). Locations were calculated using the MLE.

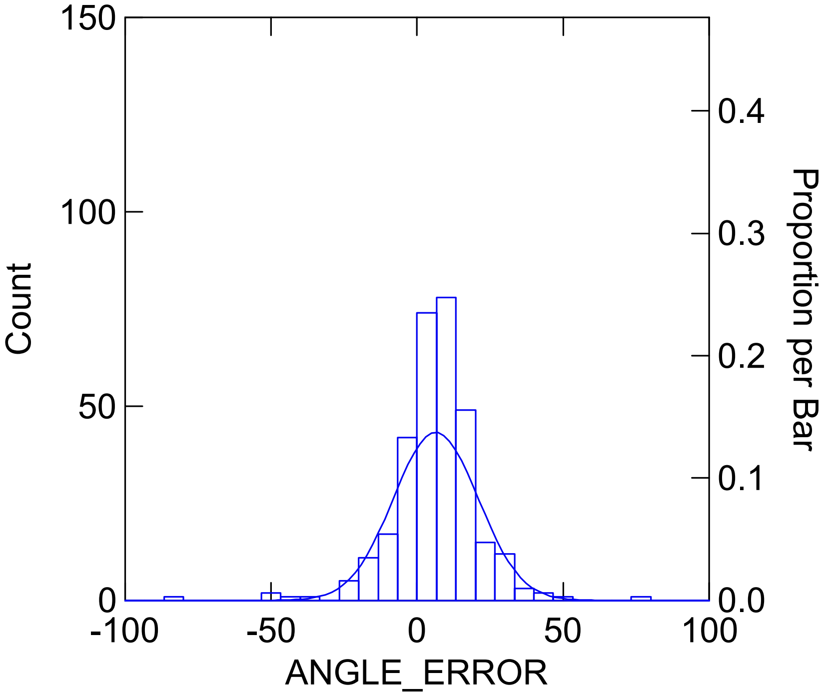


Figure 3.6. The distribution of angle error (°) for test data A with a normal curve superimposed (n = 315).

Table 3.1. Location error (m) for test data A, with bearings adjusted by 6° to account for the angle error bias for locations determined by the geometric mean of bearing intersections (GM) and maximum likelihood estimates (MLE).

Estimator	n	Mean	SD	Range	95% CI
GM	66	268.6	270.9	25.5 - 1513.3	202.5 - 334.7
MLE	66	251.1	242.3	19.8 - 1372.4	192.0 - 310.2

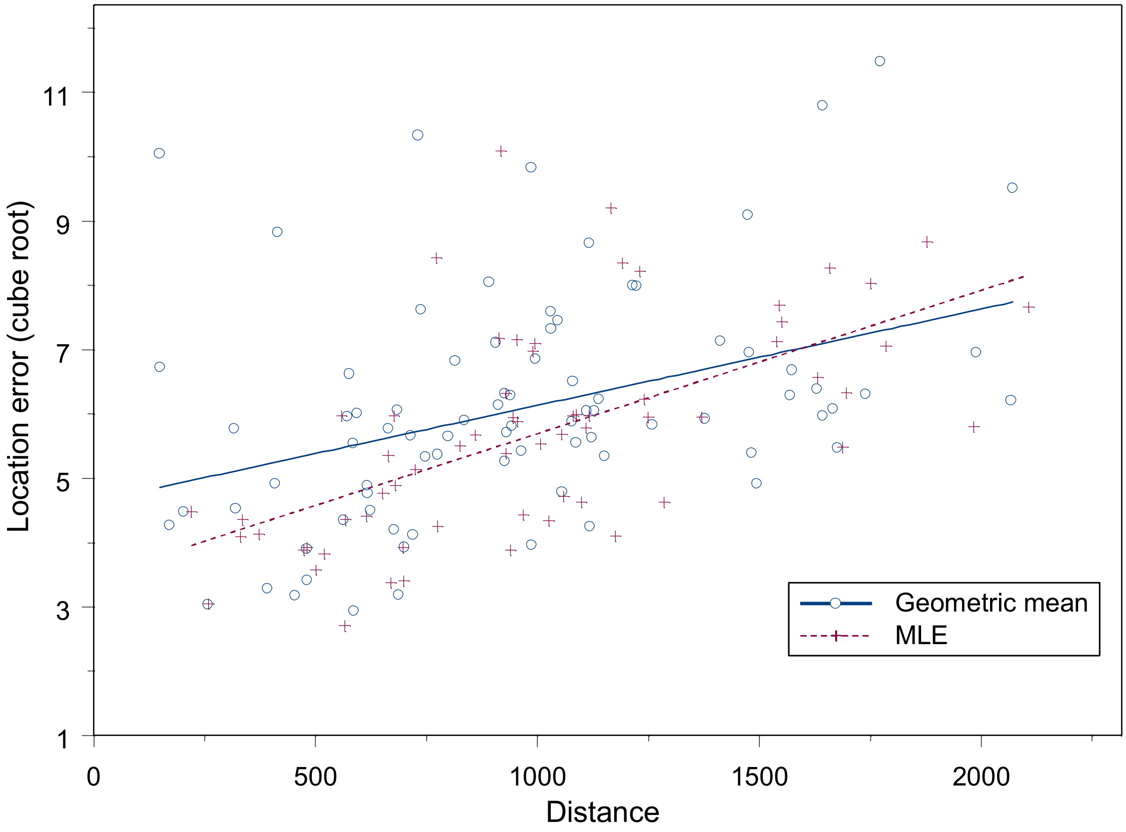


Figure 3.7. The cube root of location error (m) vs. the distance between the receiver and estimated transmitter location (m) for points derived from the geometric mean of bearing intersections (n = 83) and the maximum likelihood estimate (MLE, n = 66) for test data A.

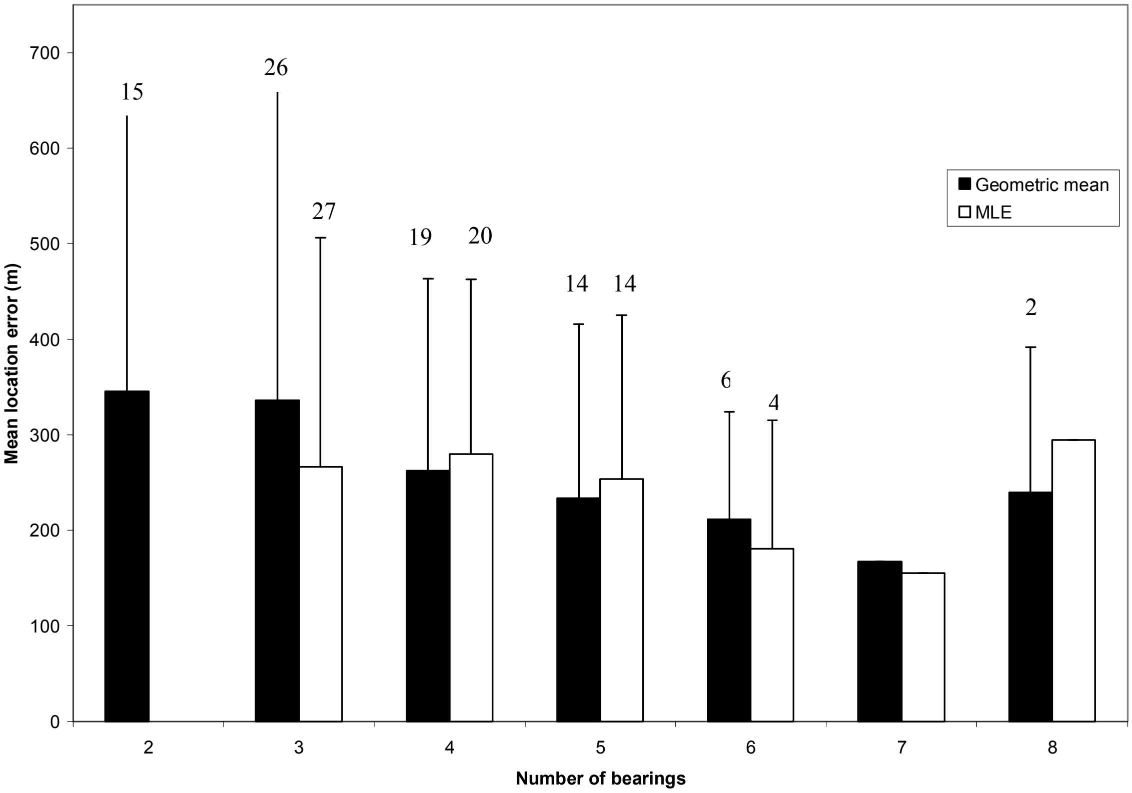


Figure 3.8. Mean location error (+ SD) as a function of the number of bearings used to make up the point location. Locations were derived from the geometric mean of bearing intersections or the maximum likelihood estimate (MLE) for test data A. Numbers above the error bars indicate sample size.

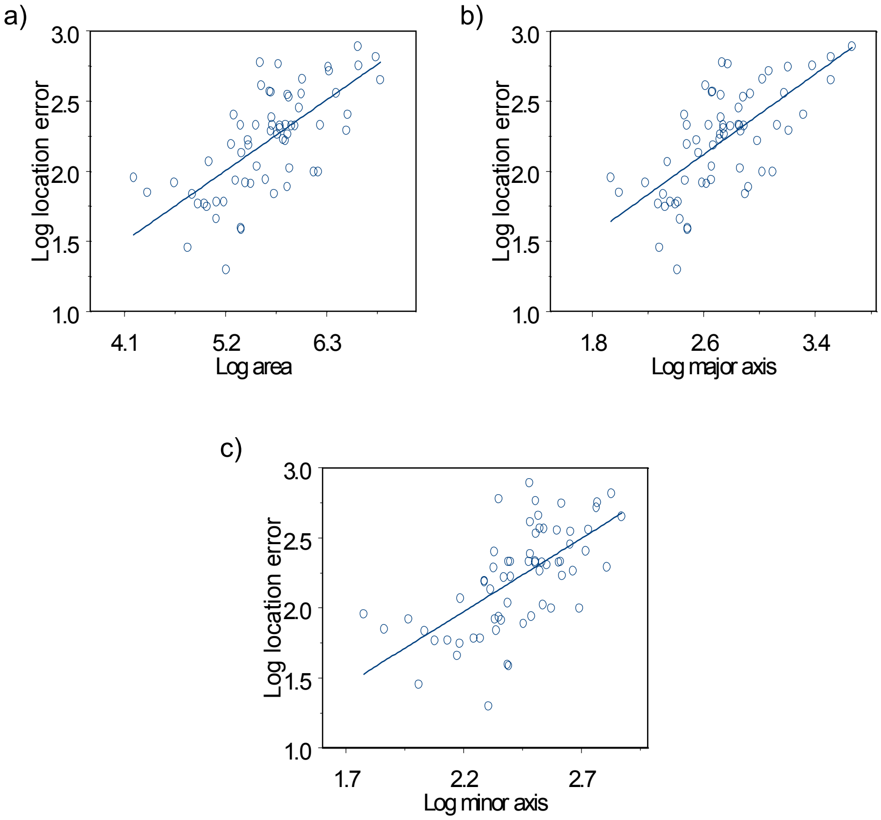


Figure 3.9. Relationship between the log transformed location error (m) and attributes of the MLE confidence ellipse for test data A (n = 66). Confidence ellipse attributes are: a) log transformed area (km^2), b) log transformed length of major axis (m) and c) log transformed length of minor axis (m).

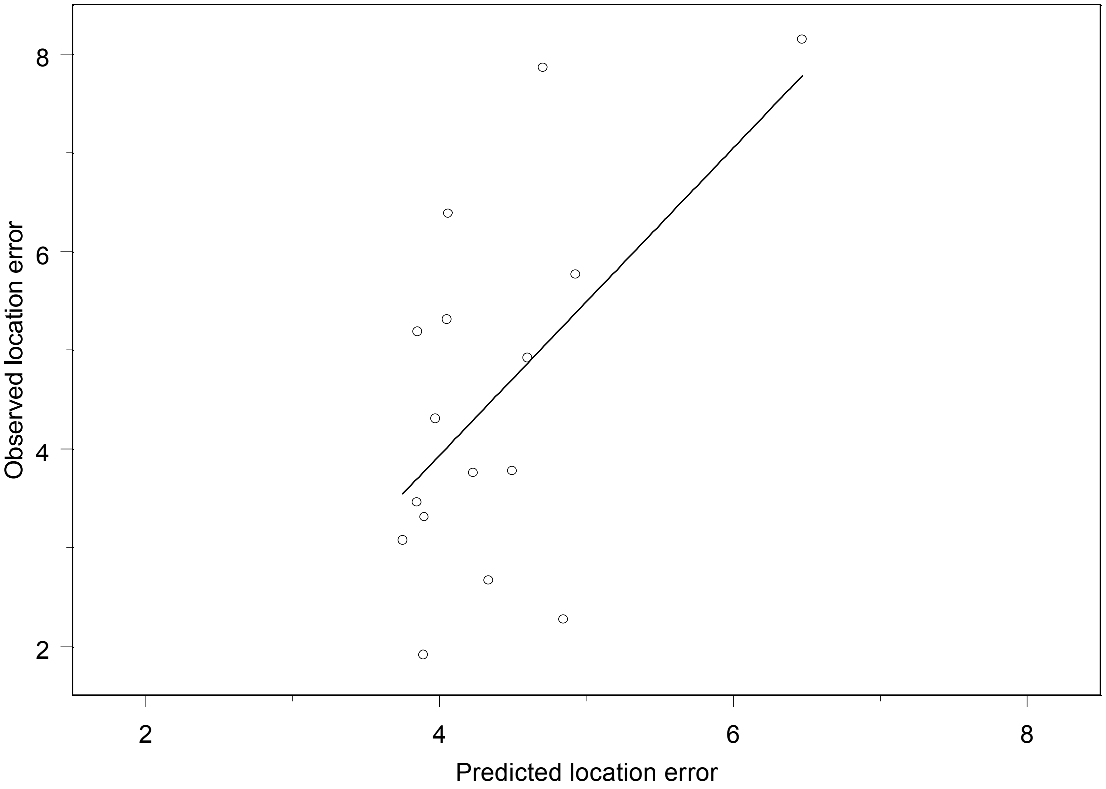
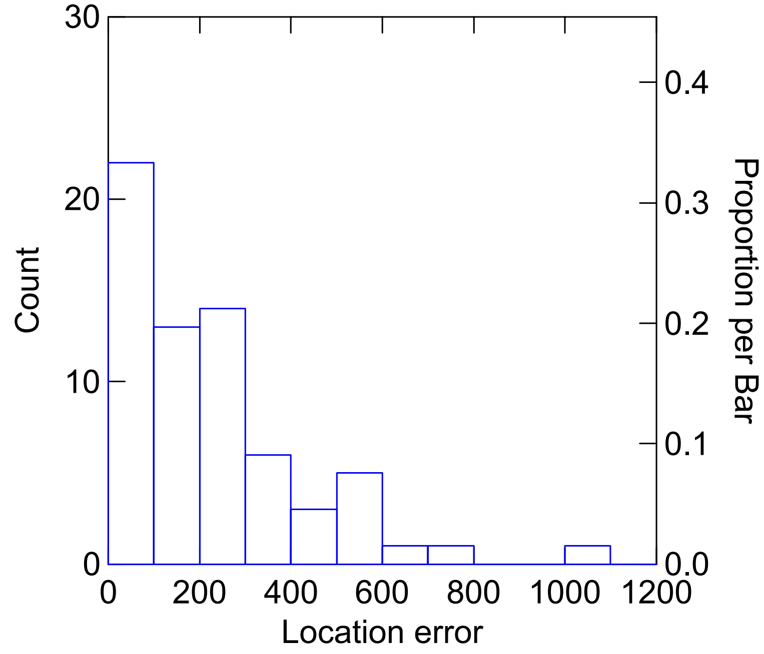


Figure 3.10. Predicted and observed MLE location error (m) (cube root transformed) for test data B (n = 16). Predicted error is derived from the regression of location error and distance between transmitter and receiver for test data A.

a)



b)

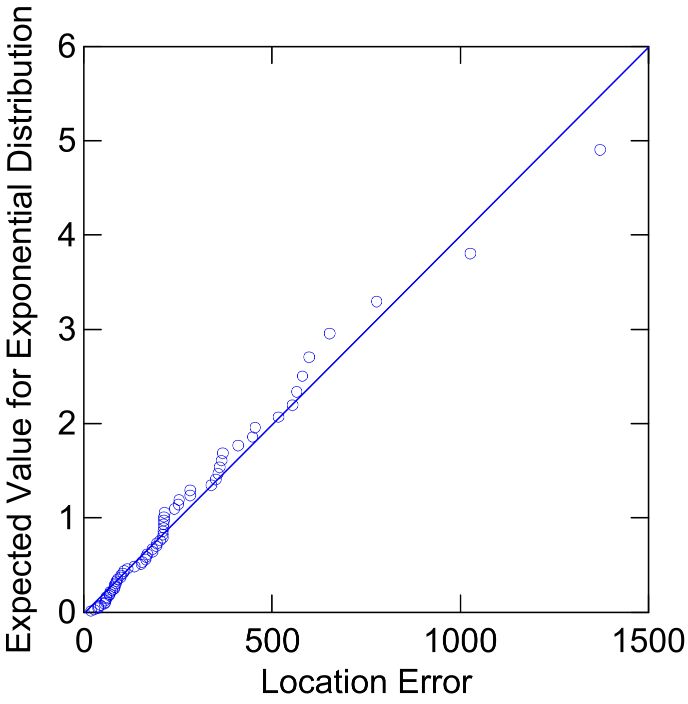


Figure 3.11. a) Distribution of location error (m). b) Probability plot of location error (m) vs. the expected value if the distribution is exponentially distributed.

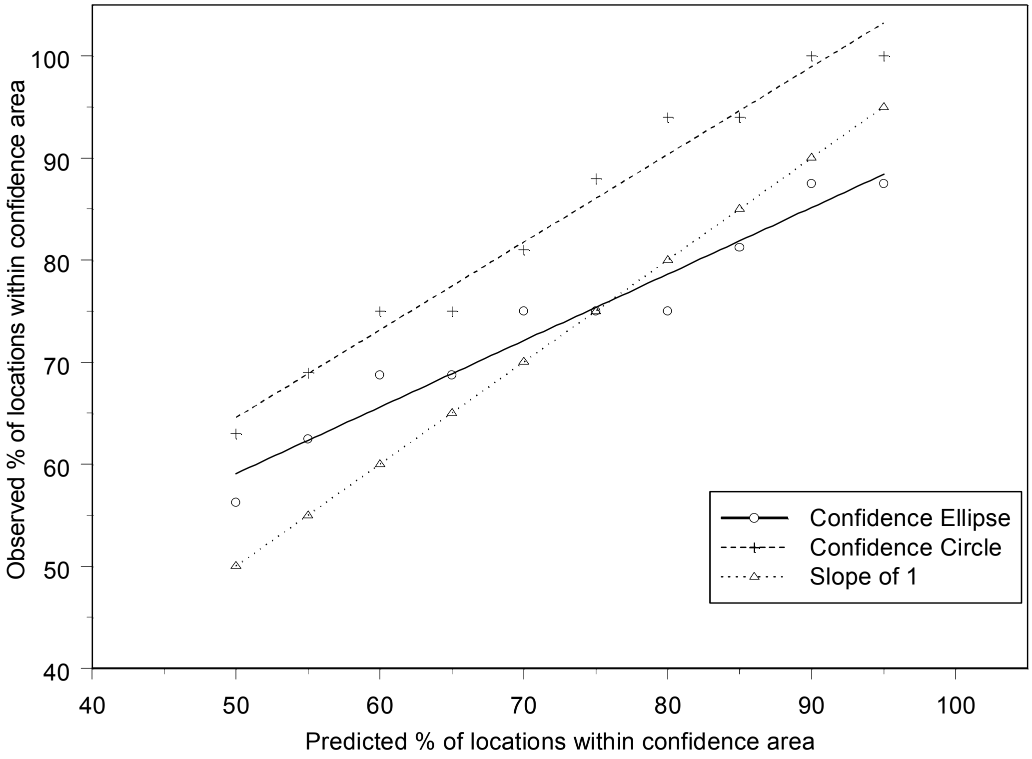


Figure 3.12. Observed percentage of test data B locations falling within x% confidence areas (n = 16), including a line passing through the origin with a slope of 1 for comparison.

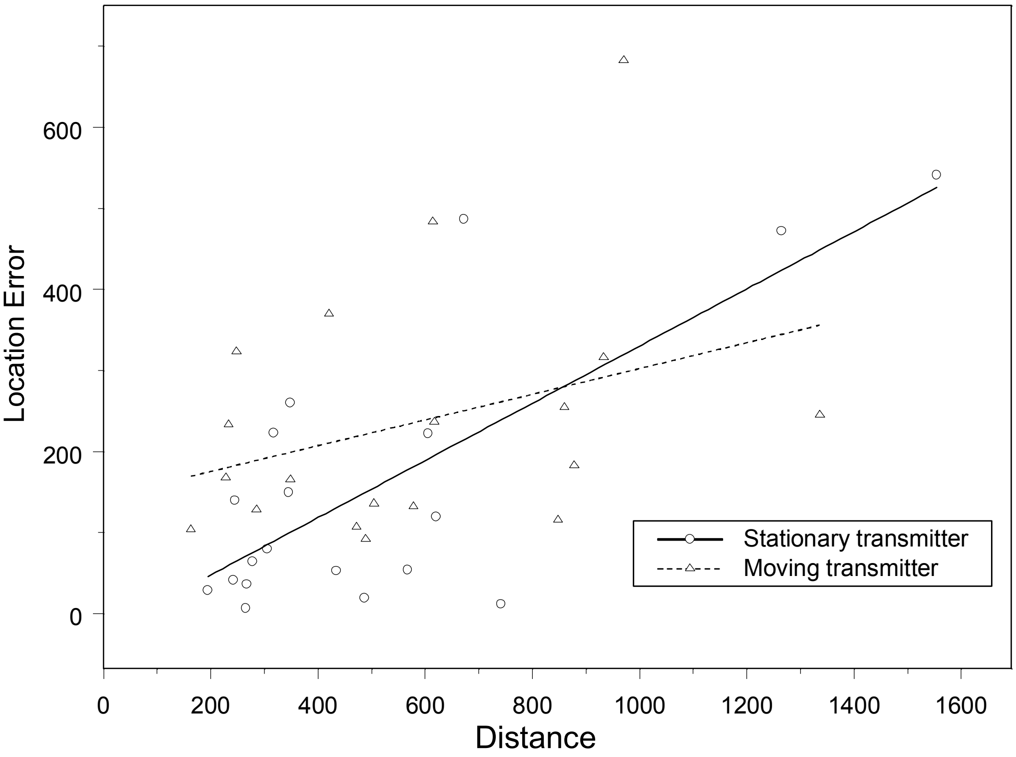


Figure 3.13. The effect of distance (m) between the transmitter and receiver on location error (m) for both moving and stationary transmitters. Locations were calculated using the maximum likelihood estimate (n = 19).

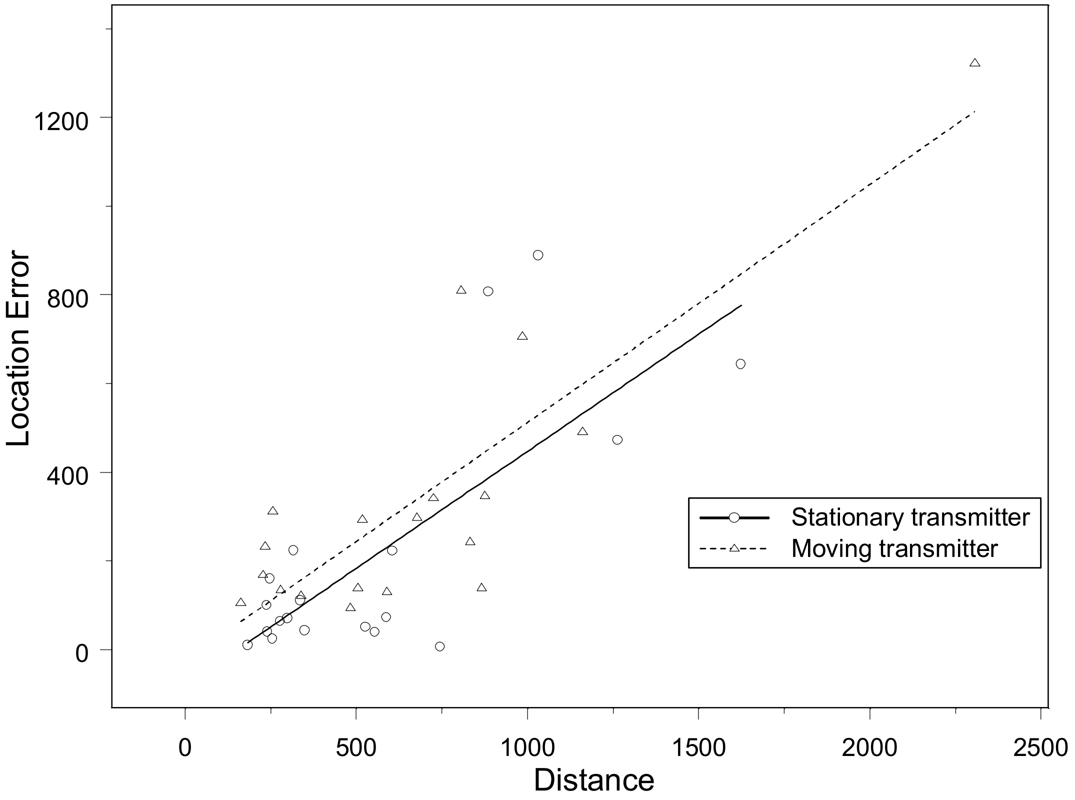


Figure 3.14. The effect of distance (m) between the transmitter and receiver on location error (m) for both moving and stationary transmitters. Locations were calculated using the geometric mean of bearing intersections (n = 19).

Table 3.2. Mean location error (m) for stationary and moving transmitters from test data B. Locations were calculated using the geometric mean of bearing intersections or maximum likelihood estimate (MLE) (n = 19).

Transmitter type	Method	Mean	Standard deviation
Stationary	Geometric mean	213.1	277.4
	MLE	158.4	169.9
Moving	Geometric mean	337.7	309.9
	MLE	235.7	150.2

Table 3.3. A comparison of mean location error (m) for locations estimated from the air and by ground triangulation (n = 7).

Location method	Mean	Standard deviation
Aerial	729.4	422.1
Ground triangulation[1]	314.8	324.0

[1] Locations calculated using maximum likelihood estimation

Table 3.4. Location error (m) reported in the literature for stationary transmitters located using ground triangulations.

Location error	Antenna type[1]	Source
50	Truck-mounted	Laundre and Keller (1981)
≤150		Arthur et al. (1989)
170	7 and 9 element truck-mounted	Lesage et al. (2000)
194	3-element	Theuerkauf and Jedrzejewski (2002)
< 200	Hand-held 2 and 5 element	Fuller et al. (2001)
275	Truck-mounted 8-element	Zimmerman and Powell (1995)
455	Hand-held 3 element	Mooty et al. (1987)

[1] Type of antenna or description of equipment used for data collection.

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Chapter 4: Fisher mortality in eastern Ontario estimated from radio-telemetry

Introduction

Knowledge of survival rates and cause-specific mortality is crucial information for the effective management of wildlife populations, especially those exposed to both natural and human-induced mortality (McLellan et al. 1999). For species subject to trapping, managers often have estimates of the numbers legally harvested each year, but illegal harvesting from poaching or high-grading (keeping valuable pelts and disposing of pelts that would cause the trapper to exceed his/her quota; Buskirk and Lindstedt 1989, Strickland 1994) reduces the accuracy of this estimate. Animals are killed on roadways or die due to natural causes and these deaths also often go undetected.

Radio-telemetry provides an opportunity to quantify deaths that would otherwise remain unknown. With this technique, radio-tagged individuals are monitored regularly, and their status, alive or dead, is recorded. All deaths of the tagged sample are, theoretically, known and the cause of death can potentially be determined.

Survival estimation

The survival function, $S(t)$, is the probability of an individual surviving t time units from the beginning of the study. Equivalently, mortality can be estimated via a hazard function or failure rate, $h(t)/\mathrm{dt}$, the probability of an individual dying in a small interval of time (t, $t + \mathrm{dt}$), given that the individual has survived until the beginning of that interval (Lee and Wang 2003). The relationship between these functions is (see Lee and Wang 2003):

$$h(t) = \frac{f(t)}{S(t)},$$

where *f(t)* is the probability density function defined as:

$$\frac{\text{(number of deaths in an interval)}}{\text{(\# individuals at risk)(interval width)}}.$$

Survival and hazard functions can be estimated either parametrically or non-parametrically. When survival times fit a well characterized distribution (e.g. exponential or Weibull distribution) parametric maximum likelihood estimation should be used to estimate survival rate (Klein and Moeschberger 1989, Miller 1983, Lee and Wang 2003); otherwise, non-parametric methods must be used (Meier et al. 2004).

Several approaches to non-parametric estimation of survival have been developed and used in wildlife research, including Heisey and Fuller's (1985) method based on Trent and Rongstad (1974), and the more traditional Kaplan and Meier (1958) method used extensively in clinical trials research. The Kaplan-Meier method requires fewer assumptions compared to the Heisey and Fuller method (Heisey and Fuller 1985, see also Tsai et al. 1999), and moreover, allows for the staggered entry of individuals into the data set (Pollock et al. 1989a). For these reasons, the Kaplan-Meier method is commonly used in survival studies involving radio-telemetry (Winterstein et al. 2001).

Kaplan-Meier survivorship estimator

Kaplan and Meier (1958) proposed a method to estimate S*(t),* the probability of surviving *t* units of time from the beginning of the study. Time is partitioned into a set of n time intervals, with the duration of the interval determined by the death of an animal. Thus, for a sample of n deaths, there are n such intervals. The probability of surviving a given interval is calculated as 1 – the proportion of animals that died in that interval. $\hat{S}(t)$ is the product of the probabilities of surviving the interval t and all previous intervals:

$$\hat{S}(t) = \prod_{j=1}^{t} (1 - \frac{d_j}{r_j}),$$

where d_j is the number of deaths in the interval j and r_j is the number of animals at risk of death during the interval j. If the study begins with n animals, the number of animals at risk is (n – the number of animals that die or are censored in all preceding time intervals).

To use the Kaplan-Meier estimator, animals must be monitored until their death. Occasionally, radio-tagged animals cannot be relocated due to transmitter loss or failure, or emigration from the study area. As the investigator does not know whether a missing animal is alive or dead, the animal is removed, or right-censored, from the study beyond the date of last contact. Animals that survive beyond the end of the study period are also right censored at that time. The Kaplan-Meier estimator handles censored data by allowing the number of animals at risk to decrease by 1 following the interval that the animal goes missing. Pollock et al. (1989a) adapted the Kaplan-Meier model so that animals could also enter the study at different times by allowing the number of animals at risk to increase as new individuals are added.

Non-parametric survivorship estimators make several assumptions (Pollock et al. 1989a, 1989b, White and Garrott 1990, Bunck et al. 1995, Tsai et al. 1999, Winterstein et al. 2001): (1) radio-tagged animals are randomly sampled; (2) survival times are independent for different animals; (3) the radio-tag does not influence survival; (4) newly tagged animals have the same survival function as previously tagged animals; (5) the time of death is known (except for right censored animals); (6) the probability of relocating an animal in the study area with a functioning radio-collar is 1; and (7) right censoring is independent of the animal's ultimate fate. The Trent-Rongstad and Heisey and Fuller estimators assume also that the survival rate is constant within an interval (Tsai et al. 1999). This assumption is a component of the Kaplan-Meier model given that the end of an interval is defined by a death.

Prognostic factors

The Cox (1972) proportional hazards model is a semi-parametric model that can identify significant prognostic factors (covariates) of survival. In this model, individuals are assumed to have identical baseline hazard functions (the hazard function when the covariates are ignored). The hazard function at time t, $h_i(t)$, is the hazard function for an individual with value i of a specified covariate. It is calculated as the product of the baseline hazard function, $h_0(t)$, and an exponential function of the covariates:

$$h_i(t) = h_0(t)\exp^{\sum_{j=1}^{p} x_{ij}\beta_j},$$

where, for p covariates, x_{ij} is the ith level of the jth covariate, and β_j is the coefficient of the covariate j. If 2 individuals have different values of one covariate x (x_1 and x_2), then under the Cox proportional hazards model,

$$\frac{h(t \mid x_1)}{h(t \mid x_2)} = \frac{h_0(t)e^{x_1\beta}}{h_0(t)e^{x_2\beta}} = \frac{e^{x_1\beta}}{e^{x_2\beta}} \quad \text{(Lee and Wang 2003).}$$

The ratio of the hazards is thus proportional and independent of time. The coefficient of the covariate, β, denotes the magnitude of the covariate's effect on the hazard function and is estimated using a partial likelihood function (see Lee and Wang 2003).

Objectives

Fisher populations in eastern Ontario are harvested, but the magnitude of this cause of death relative to other causes is unknown. The objectives of this portion of the study are to estimate the survival of fishers in eastern Ontario in 2003 and 2004, compare survival rates between sexes, and identify major causes of death. This information will be used in the final section of my thesis to assess whether the current fisher harvest is sustainable.

Methods

Sixty-one fishers were radio-collared in eastern Ontario in 2003 (see chapter 1 for a description of the study area; see also Appendix 3 for details on the trapped sample of fishers). Radio-collars were equipped with mortality sensors (see chapter 1 methods). The status of radio-collared fishers was monitored weekly from the time of radio-collaring until the time of death or censorship; roughly from February 2003 until January 2005 (Appendix 13).

The fisher fur trapping season in eastern Ontario was open from October 25 to January 15 in 2003/2004 and 2004/2005. Fur trappers were permitted to harvest 1 fisher each season, and 1 additional fisher for every 1.62 km^2 (400 acres) of registered trap line.

During the 2003/2004 trapping season, 11.7 fishers /100 km^2 were harvested in Leeds and Grenville County (Figure 4.1). This is the *reported* harvest, and is ≤ the total harvest.

Survival estimation

Fisher survival was estimated using the Kaplan-Meier (Kaplan and Meier 1958) estimator with staggered entry (Pollock et al. 1989a). Calculations were performed in S-Plus and the code is available in Appendix 14. The sampling variance of $\hat{S}(t)$ was calculated in S-Plus using the Greenwood (1926) method:

$$\text{var}\left[\hat{S}(t)\right]=\left[\hat{S}(t)\right]^2\sum_{j=1}^{t}\frac{d_j}{r_j(r_j-d_j)} \text{ (Pollock et al. 1989a),}$$

where d_j is the number of deaths and r_j is the number of animals at risk of dying in the interval j for all intervals from $j = (1,\ldots,t)$. Ninety-five percent confidence intervals were then calculated for each interval as:

$$\left[\hat{S}(t)\right]\pm 1.96\left[\text{var}\,\hat{S}(t)\right]^{1/2} \text{ (Pollock et al. 1989a),}$$

given the large sample normality of $\hat{S}(t)$ (Pollock et al. 1989a). Although most fishers were relocated weekly, some were relocated less frequently due to time constraints or difficulty in finding certain individuals. Thus, 2-week intervals were used for the survival analysis. If a fisher was located once during the 2-week interval, it was considered at risk for that interval, even if, for one of those weeks, the fisher was not relocated. If the fisher was alive one week, and dead during the second week of the same interval, the animal was considered dead for that 2-week interval.

Violations of the Kaplan-Meier estimator assumptions

Because the Kaplan-Meier method assumes that neither the live trapping procedure nor the radio-collar influence survival, 2 fishers (155.440 and 155.519b) that died in the 2 week interval following their initial capture were excluded from the survival analysis (Pollock et al. 1989a). According to necropsies performed on radio-collared fishers, no deaths of known cause were attributable to the radio-collar beyond the first 2-week interval following capture (D. Campbell, personal communication).

Bunck et al. (1995) discussed the bias introduced into survivorship estimators when the probability of relocating a radio-tagged animal is < 1. Animals that cannot be located during an interval, but are relocated alive at a later date, are often recorded as alive for the intervening time intervals. However, this can positively bias the survival estimate since the probability that investigators will record the animal as alive during intervals when it was not relocated is higher if the animal is alive in the following interval than if it is dead. Bunck et al. (1995) suggested a modified approach to survival analysis, whereby animals that are not relocated are temporarily censored, and then re-entered when they are relocated. Thus, the only animals included in the risk group in a given period are those that were relocated with certainty during that period.

For some individuals, the probability of relocation was relatively low due to the area in which the individual lived, i.e. if there were few roads in the area making relocation from the ground difficult. These individuals (155.860, 155.380 and 155.399) were only considered at risk for those intervals in which they were relocated (Bunck et al. 1995). Additionally, some fishers had radio-collars with wrap around antennas that had a smaller range of signal transmission, possibly as much as 20-50% smaller than collars with external antennae (Anderka 1987). These individuals (155.290, 155.211, and 155.231) also had a relatively low probability of relocation, and were only considered to be at risk during the intervals that they were relocated.

Survival analyses assume that the exact time of death is known, except for censored animals. In this study, at least 7 radio-collared fishers emigrated from the study area, a mean linear distance of 26 km ± 8.6 (1 SD) between initial trap location and mortality site or newly established home range (Appendix 4). These individuals were relocated by flying transects over the study area (Appendix 2). There was often a lengthy interval between the last location of an emigrant animal and the next location as determined from aerial transects. If one of these fishers was found dead, the time of death was considered to be the midpoint between the interval of the last location and the interval in which the death was detected (Krohn et al. 1994, Paragi et al. 1994b).

Fishers that permanently emigrated from the study area and outside of the area covered by the flight transects were censored in the first interval that they were not relocated. Fishers that permanently emigrated and died due to trapping were censored. These individuals have a higher probability of being detected provided that the trapper reported the death because, had the fisher died of any other cause, it is unlikely that investigators would have detected it (White and Garrott 1990). Although the cause of death of these individuals is known, they remained censored in the analysis so as to prevent negatively biasing the survival estimate. Likewise, fishers that were trapped and killed after January 2005, when the study was completed, remained censored even though the death was reported. Five known fisher deaths were censored under these criteria: 155.290 was trapped outside of the study area in January 2003, 40 km from where it was originally radio-collared (Appendix 4a); 155.980 and 155.940 were trapped as nuisance animals in February and March, 2005, respectively, and 155.560 was caught in a trap set for another species in February 2005.

Censorship

Thirty-one of the 59 collared fishers included in the survival analysis were censored over the 2-year study period. Minimum and maximum survival rates were calculated by presuming, in the former case, that all censored animals died in the interval that they were censored in, and for the latter, that all censored animals lived until the end of the study (Heisey and Fuller 1985, Pollock et al. 1989a, 1989b, Winterstein et al. 2001).

Seasonal survival rates

Seasons were delineated as follows: the February-April interval encompasses the fisher breeding season (Powell 1993), when male fishers tend to roam outside of their home ranges in search of females (Leonard 1986, Arthur et al. 1989b, Arthur and Krohn 1991, Powell 1993). Most female fishers give birth in March (Wright and Coulter 1967, Leonard 1986). During the May-July interval, female fishers devote considerable time and energy to caring for their young, especially early in the interval (Paragi et al. 1994a). During the August-October period, juvenile fishers are usually independent (Arthur and Krohn 1991, Arthur et al. 1993). The November-January interval overlaps the fisher trapping season in the study area.

The effect of covariates on the hazard function

Cox's (1972) proportional hazards model was used to examine the effect of sex on survival. Age, another obvious covariate, was not investigated due to the small sample of juvenile fishers. Fishers were classified as juvenile (<1 year) or adult (≥1 year) based on cementum annuli and pulp cavity measurements of the first premolar. With the exception of 2 female fishers captured in October, all fishers were adult within the first 2 months of the study (Appendix 3). Thus, survival estimates are pooled over age.

Covariate coefficients (β) for the Cox proportional hazards model were calculated in S-Plus (Appendix 14). S-Plus solves the partial likelihood equation for the coefficients (see Lee and Wang 2003), using Efron's (1977) method when there were ties in the survival times. The ratio of the hazards was tested for proportionality in S-Plus by calculating Schoenfeld residuals using partial likelihood estimates (see Hess 1995, Schoenfeld 1982) at each failure time. The residuals are the difference between observed and expected covariate values, given the number of individuals at risk (Schoenfeld 1980, Grambsch and Therneau 1994). Calculation of the residuals are described in Schoenfeld (1982) and Grambsch and Therneau (1994). If the hazards are proportional, there should be no pattern when residuals are plotted against time (Hess 1995).

Cause of death

When the death of a radio-collared fisher was detected, the body was retrieved as soon as possible and frozen. Necropsies of fishers that died of non-anthropogenic causes were performed by Dr. D. Campbell and Dr. K.M.Welch (Canadian Cooperative Wildlife

Health Centre, Ontario Veterinary College, University of Guelph) who determined, when possible, the cause of death.

Results

Kaplan-Meier survival estimate

The overall 2-year survival rate of fishers was 0.345 (95% CI 0.211 – 0.564; n = 59; Figure 4.2). Annual and seasonal survival rates are presented in Table 4.1, and annual survival rates, stratified by sex, are presented in Table 4.2. When the 30 censored fishers, excluding those that were still alive but censored at the completion of the study (155.080, 155.581, 155.840b), were presumed to have died during the interval that they were censored, the 2 year survival rate was 0.05 (95% CI 0.0178 – 0.140). The 2-year survival rate when all censored fishers were presumed to have lived to the end of the study was 0.560 (95% CI 0.443 – 0.707) (Figure 4.3).

The effect of covariates on the hazard function

Female and male 2-year survival rates were 0.512 (95% CI 0.336 – 0.781), and 0.146 (95% CI 0.115 – 0.505) respectively (Figure 4.4). Estimated survival rates for males were significantly less than females (Cox proportional hazards model, coefficient = 0.54, exponentiated coefficient 1.72, p = 0.007). Scaled Schoenfeld residuals plotted against time show a slight increase over time (Figure 4.5). However, a chi-squared goodness of fit test (see Schoenfeld 1980, 1982) found that residuals were independent of time ($\chi^2 = 0.403$, p = 0.525) indicating that the hazard functions for males and females are proportional.

Cause of death

There were 28 fisher deaths in 2003 and 2004; 12 (43%) were due to trapping or other human causes, 8 (28.5%) were attributable to natural causes and 8 (28.5%) were due to unknown causes (Table 4.3).

Discussion

In 2003 and 2004, adult fishers in eastern Ontario had an average annual survival rate of 0.591 (± 0.095 SD). Krohn et al. (1994) published fisher survival rates ranging from 0.57 (95% CI 0.42 – 0.78) for adult males during the trapping season, to 0.91 (95% CI 0.81 – 1.00) for adult females during the non-trapping season. Paragi et al. (1994b) reported an annual survival rate of 0.65 for adult female fishers in Maine, which is comparable to the annual survival rate of 0.67 and 0.76 (2003 and 2004, respectively) for female fishers in this study. About 23% of known deaths (n = 13) during the fall period were attributable to fur-trapping, whereas at least 61% of the deaths were due to non-harvest related causes during this period (Table 4.3). This proportion of non-harvest deaths is considerably greater than the estimate obtained by Krohn et al. (1994) in Maine, where 40 of the 41 deaths during the trapping season were due to fur trapping. There seems little doubt that non-harvest mortality contributes to the high mortality rate during the trapping season in eastern Ontario.

Of the 8 natural deaths in this study, 3 were the result of attacks. These attacks were likely by conspecifics since the carcasses were not consumed as one would expect if the fisher had been killed by a predator. The male fisher that died as a result of his injuries was found dead 3 km outside of his home range during the mating season of 2004 with puncture wounds on his face. His death was possibly the result of a fight with a resident, uncollared male fisher. Arthur et al. (1989b) documented conflict between male fishers during the mating season and Douglas and Strickland (1987) found evidence of intraspecific conflict by examining fisher pelts. One fisher died of canine distemper virus (CDV). There was evidence of CDV in 5 of the 11 carcasses that were necropsied, although this was not determined to be the cause of death. Addison et al. (1987) reviewed diseases in furbearers and also documented the presence of CDV in fishers.

A high proportion (21%) of known deaths was due to nuisance trapping (Table 4.3). This may not be attributable to a high human population density, which would increase the chance of encounters between fishers and humans, since human population density in Leeds and Grenville County (17/km^2) is comparable to Krohn et al's (1994) study area in Maine (15/km^2; Arthur et al. 1989a), and there were no reports of nuisance trapping in the latter study. The proportion of nuisance fishers trapped may not be accurately extrapolated to other areas since 5 of 8 trapped nuisance fishers (including those fishers that were trapped and reported after the monitoring period was over) were trapped by 1 group of individuals in an area with a large number of radio-collared fishers.

Estimated male annual mortality is about twice that of females (exponentiated coefficient 1.72), consistent with Strickland and Douglas (1981), who found total mortality in fishers to be higher for males than females. Krohn et al. (1994) found that trapping mortality was higher for adult males than adult females, while non-trapping mortality was similar between the sexes. Strickland and Douglas (1981) speculated that this could be due to intraspecific conflict between males. Of the 13 fishers trapped for fur or as nuisance animals in my study, including those trapped outside of the study area or beyond the end of the study, 8 were male. Of the 19 fishers that died of non-trapping related causes, 11 were male. As such, higher male mortality cannot be unequivocally attributed to either trapping or non-trapping mortality in this study.

Censorship

The large number of fishers that were censored from the study decreases the sample size and, thus, the precision of the survival estimate (Pollock et al. 1989a). By assuming that all censored fishers lived until the end of the study or died during the interval that they were censored, maximum and minimum survival functions can be estimated (Winterstein et al. 2001). If all censored fishers did, in fact, die (i.e., survival and censoring are not independent), the rate of mortality would be greater between May and November of 2003 than it otherwise appears (Figure 4.3). A large number of animals were censored near the end of the study, decreasing the precision during this period.

Assumptions of the Kaplan-Meier survivorship estimator

For the survival estimate to apply to the entire population, the sample of radio-tagged animals must be representative of the entire population. The sex and age structure of the sample should reflect that of the population, especially if the survival rates of these categories differ but are pooled in the survival analysis (Winterstein et al. 2001). Juveniles and adult male fishers tend to be overrepresented in a trapped sample (Krohn et al. 1994, Strickland 1994), possibly because juveniles disperse, and males have larger home ranges than females, increasing the frequency with which these two segments of the population encounter traps (Strickland and Douglas 1981, Buskirk and Lindstedt 1989, Krohn et al. 1994).

The extent to which my sample is representative of the population is unknown. The sex ratio in the sample was 1.44 females : 1 male (n = 61). The sex ratio of a sample of harvested fishers in Leeds and Grenville County during the 2002/2003 trapping season (2 months prior to the livetrapping in this study) was 0.85 females : 1 male (n = 144; OMNR, unpublished data). Douglas and Strickland (1987) noted that females tend to comprise a larger proportion of the harvest later in the trapping season relative to earlier. In this study, 73% of the fishers were adult (≥ 1 year old) when initially captured. The 50 fishers trapped in February and March, 2003 were all adults within the first 2 months of the study. One female fisher trapped in October was juvenile, thus, was at least 6 months old when captured. As juvenile fishers tend to be overrepresented in a trapped sample (Krohn et al. 1994), the sample of fishers used in this study likely contains a higher proportion of younger adults than present in the entire population. This could negatively bias the survival estimates since these younger fishers may be more vulnerable to trapping (Douglas and Strickland 1987). On the other hand, survival estimates could be positively biased since fishers that have been captured in live traps for this study may become trap-shy (Strickland 1994) and less likely to be trapped later on. However, tagged fishers seem to be caught readily in traps set for nuisance fishers (Table 4.3), suggesting that the potential positive bias in the survival estimate due to trap-shy individuals is negligible (see also Paragi et al. 1994b).

The survival estimators assume that survival times are independent. Since fishers are not social animals outside of the breeding season (Powell 1993), the survival of one individual should not influence that of others. However, since fishers are territorial, it is possible that a fisher may move into the territory of a recently deceased fisher, and if that habitat is more suitable, it may confer a greater probability of survival on this individual. I observed some shifting of home ranges throughout my 2 year study (Chapter 1 and Appendix 5). However, as I did not radio-collar all individuals, I do not know if this home range shift was in response to the death of an uncollared individual. If this is so, then survival estimates may be biased high.

Survival estimators also assume that live trapping and radio-collars do not influence survival. The two fishers that died immediately following trapping and collaring were eliminated from the survival analysis. No other fishers died as a result of the live trapping procedure or the radio-collars, at least as determined by necropsy. Some fishers had broken canine teeth from chewing the live traps; Arthur (1988) also reported fishers with broken or missing teeth from cage traps. Most radio-telemetry studies using collars on medium to large terrestrial mammals do not report negative effects of the collars on survivorship (Withey et al. 2001). Tuyttens et al. (2002) found that radio-collars affected the body condition of European wolverines (*Meles meles*) after wearing the radio-collar for >100 days. Negative physical effects of radio-collars were also reported for collared fishers by Krohn et al. (1994). Paragi et al. (1994b) reported that 1 juvenile fisher had minor skin irritations due to the radio-collar, while Arthur and Krohn (1991) reported no collar-induced physical stress to female fishers. Arthur (1988) noted 2 male fishers with skin irritations due to the collars and guard hairs were worn off of the necks of most collared fishers. In my study, 7 of the 17 male fishers and 4 of the 11 female fishers which were examined after wearing a radio-collar had worn-off guard hairs and/or neck abrasions. Although the irritation was not likely the cause of death, it cannot be ruled out as a contributing factor. Fishers who had been physically compromised due to the radio-collar may have been more likely to die from another cause; this could negatively bias the survival estimate. However, necropsies of 12 fishers showed that only one fisher (155.100) had reduced fat stores, and this fisher did not have neck abrasions. Furthermore, physical stress due to the radio-collars did not appear to increase the likelihood of being trapped; 5 of the 14 fishers that were trapped and 5 of 13 fishers that died of natural causes had irritation from the collar.

When animals enter the study at different times, these animals are assumed to have the same probability of survival as the animals already in the study (Pollock 1989a). One female fisher that entered the study in October was a juvenile. Although juveniles are more likely to be harvested than adults (Strickland and Douglas 1981, Douglas and Strickland 1987, Buskirk and Lindstedt 1989, Krohn et al. 1994), this individual was not harvested, which lends (weak, albeit) credence to this assumption.

The time of death is assumed to be known exactly, except for right-censored animals. There is some uncertainty surrounding the time of death for fishers 155.779 and 155.700. These 2 fishers were found dead after they had emigrated from the study area (Appendix 4). Eight and 10 weeks, respectively, had passed between when these fishers went missing and when they were located. In these cases, the time of death was assumed to be halfway between the time that the individual went missing and when the mortality signal was detected. If these individuals had, in fact, died in the interval following the date that they were missing, then the Feb-Apr 2003 survival estimate is biased high and the May-July 2003 estimate is biased low. However, given that the 2 individuals traveled at least 14.5 and 27 km, respectively, during the time when there was no contact, and that the carcasses were greatly decomposed when they were discovered, the assumption that the fishers died in the interval between the last contact and their subsequent discovery is the best estimate of time of death.

Tsai et al. (1999) considered the effect of non-random censoring on the Kaplan-Meier estimate. They found that when live and dead animals have different probabilities of emigrating or of being relocated, the survival estimate is biased. It is possible that some fishers were killed by trapping or were hit by a vehicle and the collar was destroyed without the knowledge of the investigator. This would positively bias the survivorship estimate since these deaths would not be included in the analysis.

Survival rate estimation is an important demographic variable to measure because it provides fur managers with an indication of the loss of individuals from a population. Furthermore, partitioning deaths into trapping and non-trapping categories gives an indication of the impact of this human-caused mortality relative to natural mortality. These measures, along with estimates of fisher density, will be assessed collectively in the final section: Management Implications.

Tables and Figures

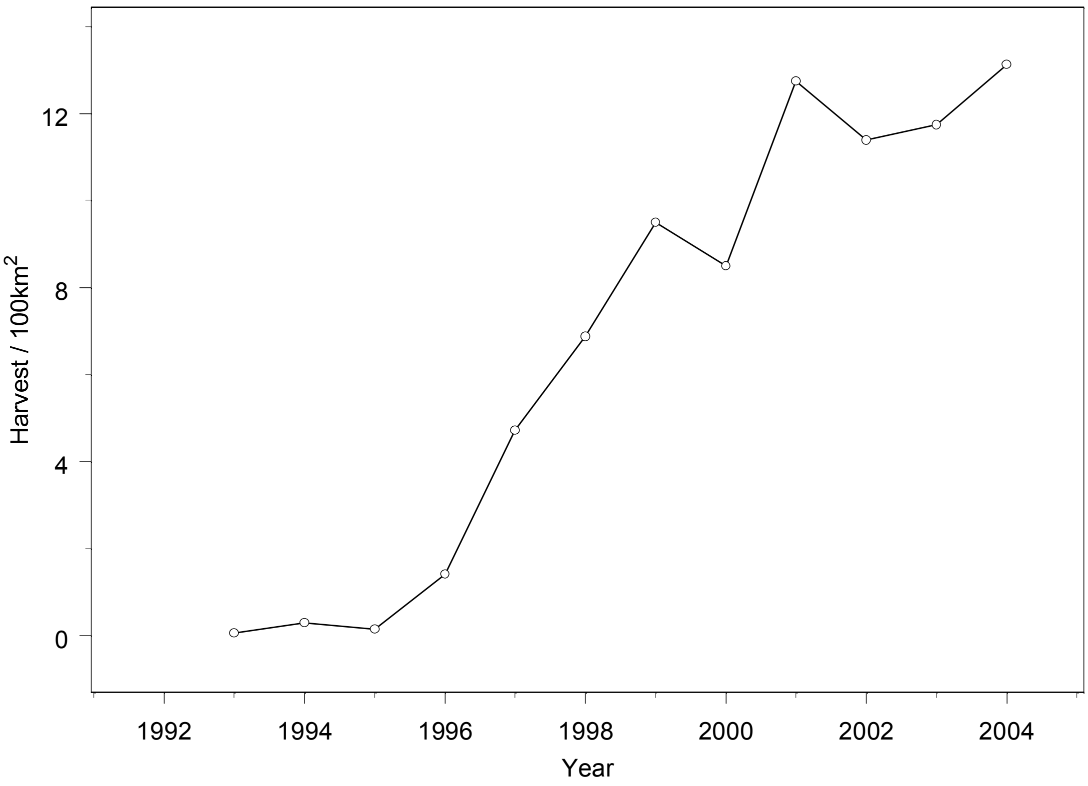

Figure 4.1. Fisher harvest density in Leeds and Grenville County between 1993/94 and 2004/05 from FURMIS (FUR Management Information System) data (OMNR).

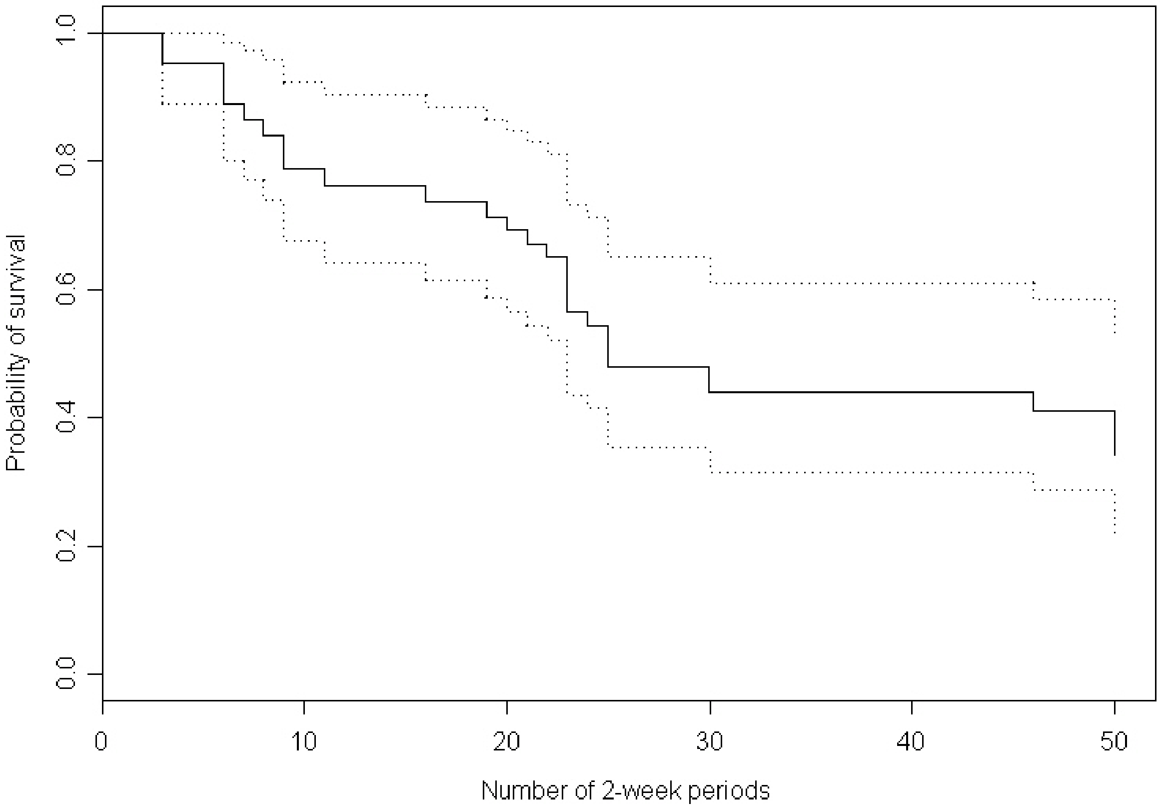

Figure 4.2. Two-year Kaplan-Meier survivorship and 95 % confidence intervals (dotted lines) for fishers between February 2003 and December, 2004 (n = 59). Each time interval represents a 2 week period.

Table 4.1. Annual (Feb-Feb) and seasonal Kaplan-Meier survival estimates for fishers in eastern Ontario in 2003 and 2004 (N = 59).

Time interval	Survival	95% CI	n[1]
2003	0.524	0.389 – 0.706	17-49
2004	0.658	0.542 - 0.800	3-22
Feb-Apr 2003	0.903	0.824 – 0.990	17-49
Feb-Apr 2004	0.916	0.782 – 1.000	20-22
May-Jul 2003	0.879	0.766 – 1.000	28-37
May-Jul 2004	1.000	0.866 – 1.000	18-20
Aug-Oct 2003	0.922	0.799 – 1.000	30-34
Aug-Oct, 2004	1.000	0.866 – 1.000	15-18
Nov-Jan 2003/2004	0.716	0.581 – 0.898	22-31
Nov-Jan 2004/2005	0.719	0.585 – 0.938	3-15

[1] Range of number of fishers at risk during the interval

Table 4.2. Annual (Feb-Feb) Kaplan-Meier survival rates of male and female fishers in eastern Ontario in 2003 and 2004 (n = 59).

Year and sex	Survival	95% CI	n[1]
Male			
2003	0.328	0.179 – 0.600	5-20
2004	0.445	0.238 – 0.832	0-7
Female			
2003	0.633	0.467 – 0.857	12-29
2004	0.809	0.719 – 0.911	3-23

[1] Range of number of fishers at risk in the interval

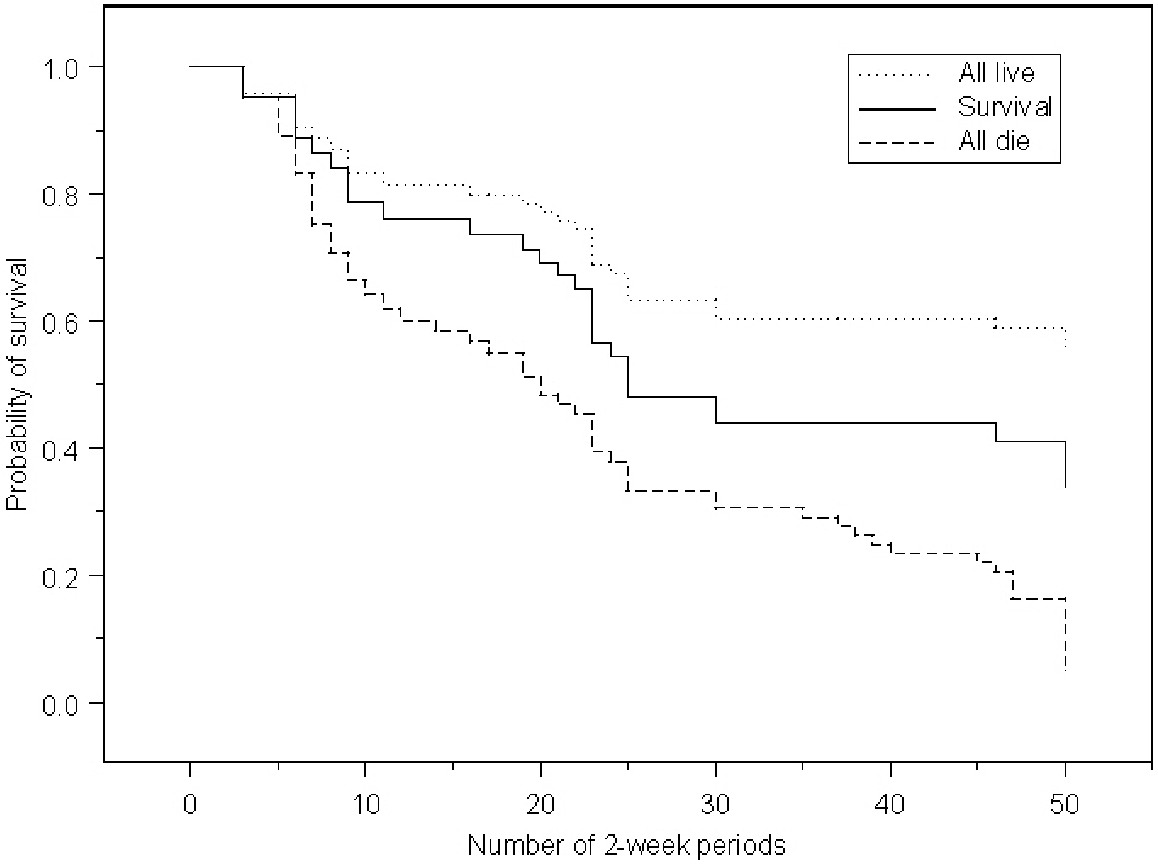

Figure 4.3. Two-year Kaplan-Meier survivorship for fishers between February 2003 and December 2004 (n = 59). The measured survival curve for the period (survival) is compared to survival curves when all censored fishers are presumed to have lived to the end of the study (all live) and all censored fishers are presumed to have died in the interval that they were censored, except those that lived beyond the study period (all die).

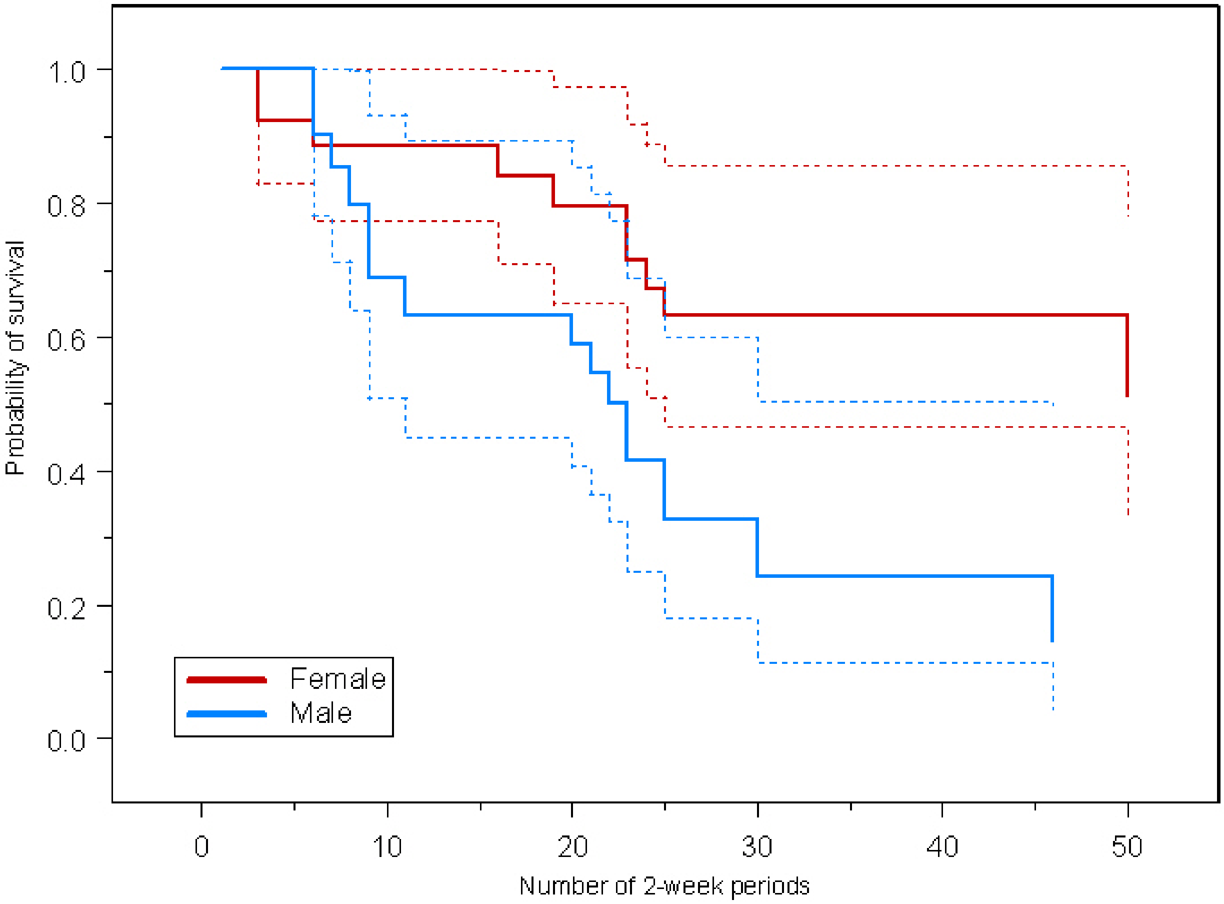

Figure 4.4. Two-year Kaplan-Meier survivorship and 95% confidence intervals (dotted lines) for male and female fishers from February 2003 until December 2004 (n = 59).

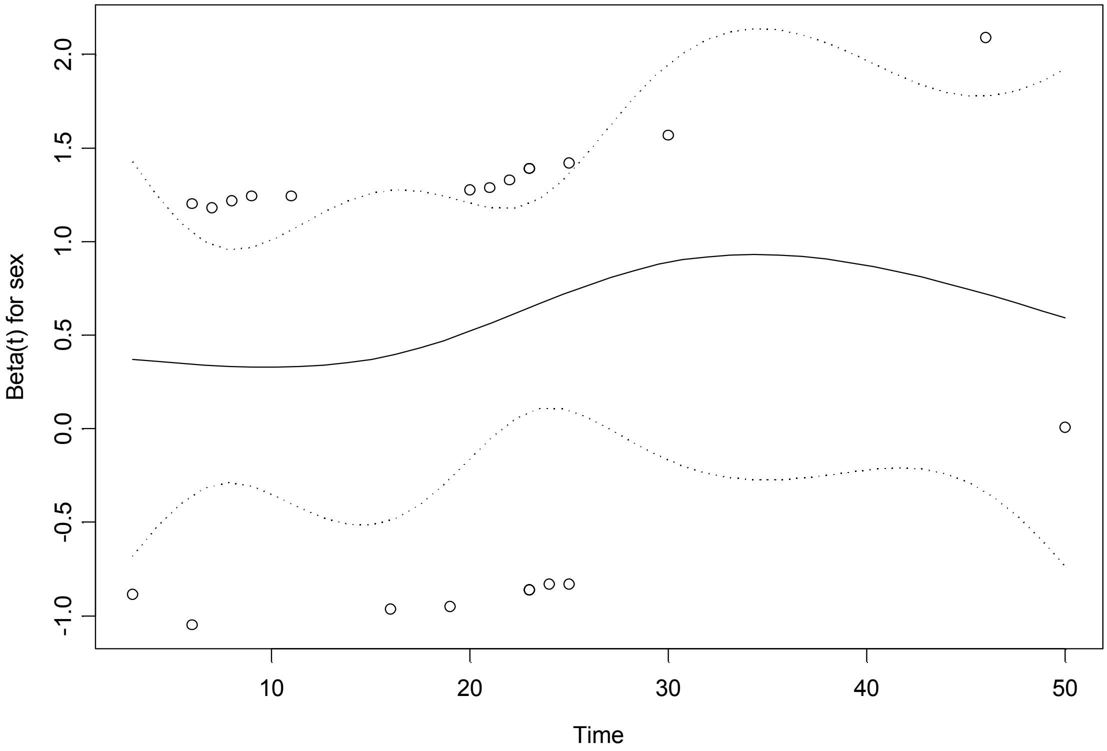

Figure 4.5. Scaled Schoenfeld residuals for the Cox proportional hazards model of fisher survival (over 2 years), with sex as a covariate. The solid line is a smoothing spline and the dotted lines are ± 2 standard error. Departures from a horizontal line are indicative of non-proportional hazards.

Table 4.3. Number of fisher deaths due to specific causes in eastern Ontario, presented by season and sex, between February, 2003 and January, 2005. Years are pooled and grouped by season. Fifty-nine fishers were at risk (35 females, 24 males).

	Number of Deaths								
	Feb-Apr		May-Jul		Aug-Oct		Nov-Jan		
Cause of Death	♂	♀	♂	♀	♂	♀	♂	♀	Total
Human-caused									
Trapped (nuisance)	2	1	1	0	0	0	1	1	6
Trapped (fur)	0	0	0	0	0	0	2	1	3
Other[1]	0	0	0	0	1	0	1	1	3
Natural									
Injuries resulting from attack	1	0	0	0	0	0	0	1	2
Other[2]	1	0	1	0	0	1	2	1	6
Unknown	0	2	3	0	0	1	1	1	8
Total	4	3	5	0	1	2	7	6	28

[1] Other includes road kill and suspected insecticide poisoning.

[2] Other includes emaciation, canine distemper virus, bronchopneumonia, septic peritonitis (likely pancreatitis), and possible drowning.

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Management Implications

Currently in eastern Ontario, fur managers set fisher quotas at 1 fisher per trapper, and additional quotas are allotted at 1 fisher per 1.62 km^2 (400 acres) of registered land. These quotas are then adjusted annually according to Strickland and Douglas (1981), where age and sex ratios in the harvested sample are used as indices of population change. Strickland and Douglas (1981) found that, in the Algonquin region of Ontario, ratios of > 4 juveniles per adult female fisher in the harvest coincided with an increasing population and < 4 with a decreasing population. If indeed these ratios can be used to distinguish declining from increasing populations in eastern Ontario, ratios of juvenile to adult female fishers for Leeds and Grenville County, obtained from fisher teeth voluntarily submitted by trappers (OMNR, unpublished data), would suggest that from 2000 to 2003, fisher populations have declined (Table 5.1), consistent with catch per unit effort data from the Rabies Research and Development's live trapping program in eastern Ontario (Figure 5.1; Bowman et al. unpublished data). However, these data would appear to contradict reports from fur trappers and local residents, many of whom maintain that fisher populations are healthy, indeed, perhaps too healthy. With this in mind, I will here discuss (1) whether the current harvest is sustainable according to guidelines from the literature; (2) the recruitment necessary for a stable population, given annual mortalities estimated in the current study; and (3) whether, in light of (1) and (2), the current quota system is likely to be effective in managing fisher populations in eastern Ontario.

Is the current harvest sustainable?

The issue of the sustainability of the current harvest revolves around the balance (or lack thereof) between recruitment and survival, and, in particular, the contribution of harvesting mortality to total mortality. Douglas and Strickland (1987) and Paragi et al. (1994) estimated that harvesting > 25% of the pre-harvest population (the population size prior to the trapping season) resulted in a population decline the following year in Ontario and Maine, respectively. Similarly, Douglas and Strickland (1987) summarized unpublished data from Minnesota and concluded that harvests of 15-20% of the pre-trapping fisher population resulted in a stable population, while harvests of > 20% led to a population decline.

With current available data, two different estimates of harvest mortality may be obtained. The first, and simplest, is just the proportion of radio-collared fishers harvested: in the 2003/04 and 2004/05 trapping seasons, 6.5% and 6.7%, respectively, of *adult* radio-collared fishers were harvested. However, in the harvests studied by both Krohn et al. (1994) and Strickland and Douglas (1981), the proportion of juveniles in the harvest sample greatly exceeded adults, suggesting that perhaps juveniles were more susceptible. In Leeds and Grenville County, there was no significant difference between the number of adults and juveniles in the harvest between 2000 and 2003 (Table 5.1; t = 1.226, df = 3, p = 0.308), although the power to detect a difference of the reported magnitude was low (power = 0.19). Because the proportion of adult trapping mortalities may not be representative of the true impact of trapping, an estimate that also incorporates juvenile mortality is appropriate.

A second estimate can be obtained by dividing the number of fishers harvested (from FUR Management Information System (FURMIS) data) by the estimated number of individuals in the area (calculated from population density estimates). The FURMIS data are broken down into regions; the boundaries of the Brockville region are the same as those of Leeds and Grenville County (Figure 1.1, Chapter 1). If we assume that the resident *adult* fisher population density estimated for the 894 km^2 study area (32.6/100 km^2) within Grenville County (Figure 1.2, Chapter 1) is representative of all of Leeds and Grenville County (3,390 km^2), then there were approximately 1,105 resident adult fishers in Leeds and Grenville County in both 2003 and 2004. As the pre-harvest population contains both resident adult and transient juvenile fishers, estimates of the size of the pre-harvest population can be adjusted to account for these transient juvenile fishers (Arthur et al. 1989). Paragi et al. (1994) estimated that between 0.8 and 1.4 fisher kits per adult female survive from birth until the trapping season in Maine. Assuming this applies to eastern Ontario, in addition to the 1,105 resident fishers in Leeds and Grenville County in 2003 and 2004, there were also between 646 and 1,130 transient juvenile fishers in the pre-harvest population. Based on the FURMIS data, 398 fishers (adult and juvenile) were harvested in Leeds and Grenville County during the 2003/04 trapping season. Assuming that fisher skulls turned in for analysis represents a random sample of the harvested population, we can use the adult: juvenile ratio in the former (1:1.11, n = 55; Table 5.1) to predict the latter. This yields an estimated 189 adult and 209 juvenile fishers in the trapped population, which represents 17.1% and between 18.5 and 32.3% of the adult and juvenile pre-trapping population, respectively. Overall, between 17.8 and 22.7% of the 1751-2235 fishers in the pre-harvest population (all ages) were harvested in the 2003/04 trapping season in Leeds and Grenville County. This calculation was repeated for the 2004/05 trapping season, using the same population density and adult: juvenile sex ratio as in the former calculation: overall, between 19.9 and 25.4% of the pre-harvest population was harvested in 2004/05 (Table 5.2). If the estimates of Douglas and Strickland (1987) and Paragi et al. (1994) also apply here, the harvest in Leeds and Grenville County would then be close to the threshold for population decline.

Of course, the estimated threshold for population decline obtained by Douglas and Strickland (1987) and Paragi et al. (1994) need not apply in Leeds and Grenville County. This threshold will depend on both recruitment and non-harvest related mortality, and we have no estimate of the former for this area. We do know, however, that non-harvest mortality in Leeds and Grenville County is high relative to estimates in Maine (Krohn et al. 1994, Paragi et al. 1994; see Chapter 4). This implies that unless recruitment in Leeds and Grenville is substantially higher than in Maine, the harvest threshold in Leeds and Grenville County will be *less* than the 25% calculated for Maine.

Estimates of trapping mortality require estimates of both population size and the number of fishers trapped. In our case, the population size estimate for resident adults is based on territory mapping and is, therefore, open to considerable uncertainty. Furthermore, the estimate of juvenile population size is based on estimates of recruitment from fishers in Maine. The number of fishers recorded as harvested is less than or equal to the total number of fishers actually trapped due to high-grading, the incidence of which is unknown. High-grading also biases the trapped sample towards females since female pelts are more valuable (Obbard 1987), influencing estimates of the sex ratio in the harvest.

Recruitment and a stable population

Henny et al. (1970) developed an equation to estimate annual population change that has been applied to many species, including river otters (Tabor and Wight 1977), martens (Hodgman et al. 1994) and fishers (Paragi et al. 1994). If a population is stable (i.e. production equals mortality), then:

$$1 = m_1S_0 + m_2S_0S_1 + m_3S_0S_1S_2 + \ldots \text{ (Henny et al. 1970),}$$

where m_x is the age specific recruitment rate and S_x is the age specific survival rate. This equation was simplified by Henny (1970) and Paragi et al. (1994) to give:

$$1 = mS_0S + S, \tag{1}$$

where m is the recruitment rate, or the number of female offspring per adult female (≥ 2 years old) that survive from birth until the anniversary date to which all estimations are relative (Henny et al. 1970). The recruitment rate is assumed to be the same for all females ≥ 2 years old (fishers reproduce for the first time in their second year (Wright and Coulter 1967)). S_0 is the annual survival rate of juvenile (< 1 year old) females, and S is the annual survival rate of all female fishers ≥ 1 year old. Given observed survival rates, the production necessary to have a stable population can be calculated by solving for m in (1).

Equation 1 was applied to the fisher population in my study, with an anniversary date of February 1. Annual female survival (S) from February-February was 0.633 for 2003 and 0.809 for 2004 (see Chapter 4). As juvenile survival was not measured in this study, a range (0.05 – 0.85) of possible juvenile survival rates were used, assuming that juvenile survival is less than adult survival (Krohn et al. 1994). Depending on juvenile survival, the annual female recruitment necessary to maintain a stable population ranges from 0.682 - 11.6 in 2003 and 0.278 - 4.72 in 2004 (Figure 5.2). In Maine, Paragi et al. (1994) estimated annual juvenile survival to be 0.27. If juvenile survival in this study is assumed to be similar (0.25), then in 2003 and 2004, 2.32 and 0.944 female kits per female, respectively, would have to survive the first year if the population is to remain stable. Sex ratios for fisher kits is close to 1:1 (Douglas and Strickland 1987), which means that at least 4.64 and 1.89 kits ($2m$) per adult female (2003 and 2004, respectively) must be born and survive the first year for the population to be stable. Mean litter size for fishers ranges between 2 and 3 (Powell 1993, Mead 1994, Paragi et al. 1994), so even if every female fisher whelped kits (estimated denning rate is less than 100%: 81% (Leonard 1986), 63% (Paragi et al. 1994), 75% (Arthur and Krohn 1991)), these calculations strongly suggest a declining population, at least in 2003.

Henny et al.'s (1970) equation assumes that, for a stable population, total mortality equals total production; it does not consider immigration or emigration. The recolonization of eastern Ontario by fishers is comparatively recent (deVos 1964). Carr et al. (unpublished data) used DNA microsatellite analysis to determine that fisher populations in the study region are recent migrants from the Adirondack region of New York State. It is possible that immigration from this area is supplementing the fisher population in Leeds and Grenville County. However, catch per unit effort data from the Rabies Research and Development Unit's live trapping program also indicate that fisher populations are declining (Figure 5.1; Bowman et al. unpublished data).

Trapping quotas and compensatory mortality

There is pressure from the trapping community and local residents in eastern Ontario to raise fisher quotas. The question is whether the population can sustain a larger harvest. Since non-trapping mortality in adults is currently relatively high in this area (see Chapter 4), the question becomes one of compensatory versus additive mortality.

Populations in which harvests are compensatory exhibit density-dependent mortality and natality: at high densities, there is less food and space per individual, inducing higher rates of mortality and lower productivity than in less dense populations (Boyce et al. 1999). Disease and cannibalism at high densities can also affect mortality (Boyce et al. 1999). Hunting and trapping serve to reduce the density of the population, thus reducing density dependent mortality and increasing natality following the harvest. Thus, harvested animals take the place of animals that would otherwise die of density-dependent causes, so that the size of the population remains stable whether the animals are harvested or not. Hunting and trapping mortalities can also be additive, such that harvest mortality is in addition to non-harvest mortality and population size decreases as a function of the level of harvest pressure (Boyce et al. 1999, Poysa et al. 2004).

At present, we have little information on whether harvesting is compensatory in fisher populations generally (Douglas and Strickland 1987), and even less information concerning the study area. Strickland and Douglas (1981) found that when population densities were low following a heavy harvest, there was a higher proportion of unbred females, suggesting that fisher populations may not exhibit density-dependent natality, a characteristic of populations with compensatory harvest mortality (Boyce et al. 1999). On the other hand, Krohn et al. (1994) suggest that fisher trapping is compensatory since trapping removes resident fishers, leaving vacant territory for transient juveniles to occupy, thus increasing juvenile survival. In neither case, however, was there substantive evidence one way or the other.

Recommendations

This study is one of the most intensive and complete studies describing fisher home range, population density, habitat preference, and survival in Ontario to date. Although I have highlighted weaknesses in the data and analysis throughout my thesis, these weaknesses are inherent to most studies of this type. My estimates are, thus, the best estimates possible, given the limitations of the data.

I have shown that, although fisher population density was relatively high in 2003 and 2004 compared to other geographical regions in the fisher range, it is unlikely that recruitment can sustain the estimated mortality rates. This assessment is corroborated by independent age and sex ratio data from the harvest, and catch per unit effort data from the Rabies Research and Development Unit, all of which suggest a declining fisher population in Leeds and Grenville County.

Whether decreasing fisher quotas would mitigate the postulated fisher decline is unclear. As fur trapping only constitutes a small proportion of total adult mortality (6.5 – 6.7%), decreasing fisher quotas will not greatly reduce total adult mortality. However, although I do not know what proportion of the pre-trapping population can be harvested sustainably, the threshold is likely less than 25% proposed by Paragi et al. (1994). The 2003/04 harvest took between 17.8 and 22.7% of the pre-trapping population and could be close to this (unknown) threshold; if so, a significant reduction in trapping mortality could well make the difference between future population increase or decrease. Conservative management should assume that harvest mortality is additive, since there is currently no evidence to suggest otherwise. The weight of evidence suggests that fisher harvest quotas should not be increased due to the risk of increasing total adult mortality, thus exacerbating the current (substantial) risk of future population decline.

Tables and Figures

Table 5.1. Ratio of juvenile (J) to adult female (AF) or Adult (A) fishers in the harvested population in Leeds and Grenville County, Ontario.

Year	Ratio (J:AF)	Ratio (J:A)	n
2000/01	1.83 : 1	0.76 : 1	51
2001/02	1.56 : 1	0.91 : 1	111
2002/03	0.78 : 1	0.47 : 1	144
2003/04	1.81 : 1	1.11 : 1	55
2004/05	1.0 : 0	1 : 0	1

Data obtained from a sample of fisher teeth voluntarily submitted by trappers (OMNR, unpublished data).

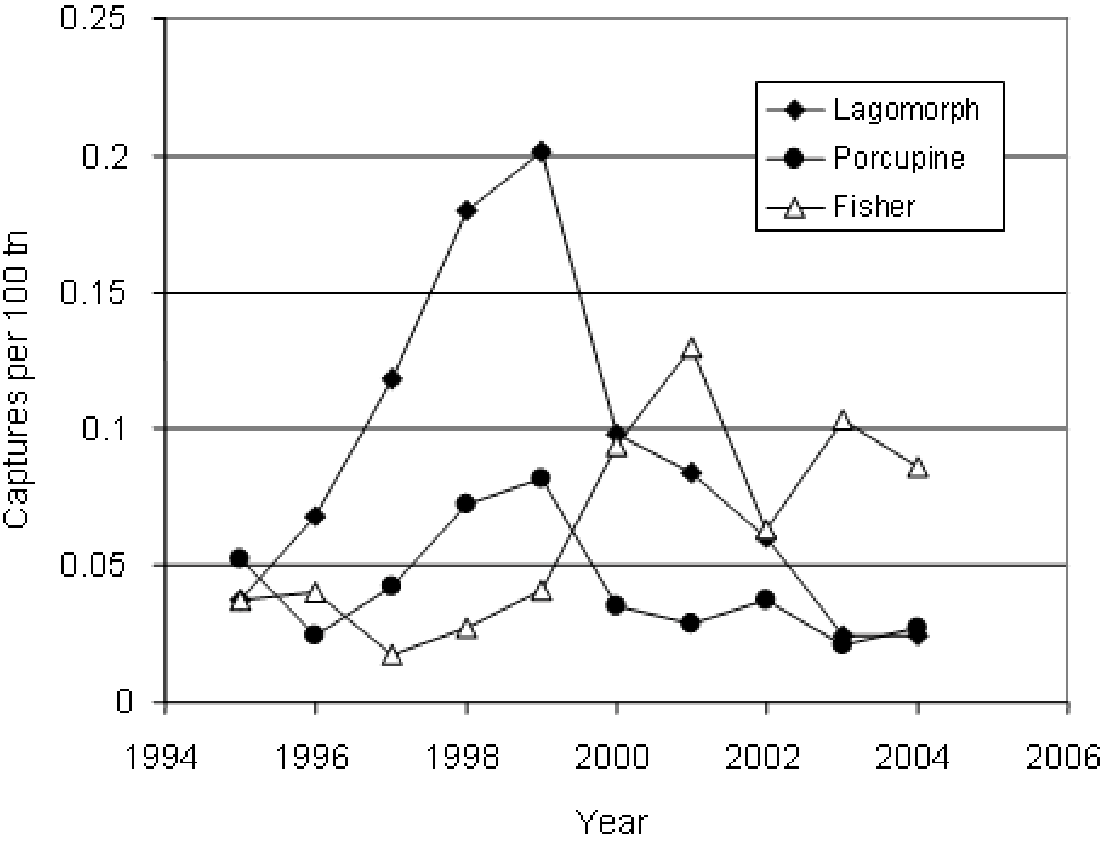

Figure 5.1. Number of fishers (*Martes pennanti*), porcupines (*Erethizon dorsatum*) and lagomorphs (*Lepus americanus* and *Sylvilagus floridanus*) caught per 100 trap nights in eastern Ontario between 1995 and 2004 by the Rabies Research and Development Unit (reproduced with permission from Jeff Bowman et al. (unpublished data)).

Table 5.2. Estimates of the percent of the pre-trapping population harvested in Leeds and Grenville County, ON, for adult (A), juvenile (J) and total fishers.

Season	Population size[1]			Number harvested[2]			% harvested		
	A	J	Total	A	J	Total	A	J	Total
2003/04	1,105	646 - 1,130	1751 - 2235	189	209	398	17.1	18.5 – 32.3	17.8 – 22.7
2004/05				211	234	445	19.1	20.7 – 36.2	19.9 – 25.4

[1] Estimated number of fishers in Leeds and Grenville County based on estimated population density

[2] Total number harvested is from FURMIS data, proportions by sex estimated from sex ratio in the harvest of 2003/04

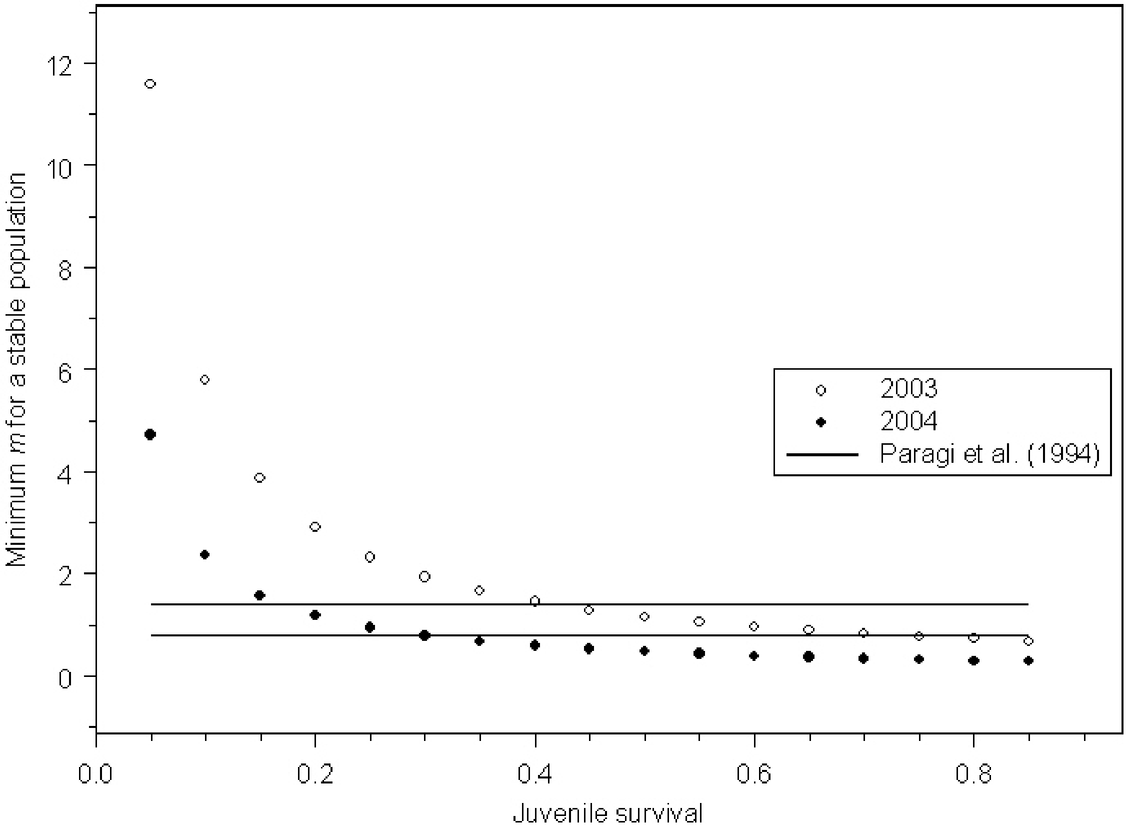

Figure 5.2. Estimates of annual female recruitment per adult female fisher (m) necessary to maintain a stable population, given annual adult female survival rates of 0.633 for 2003 and 0.809 for 2004, for varying estimates of juvenile survival (S_0), using equation (1). The horizontal lines are the range of estimates of m from Paragi et al. (1994). Dots above the lines show the conditions for which mortality is greater than recruitment, resulting in population decline in Leeds and Grenville County, given observed survival rates in 2003/04.

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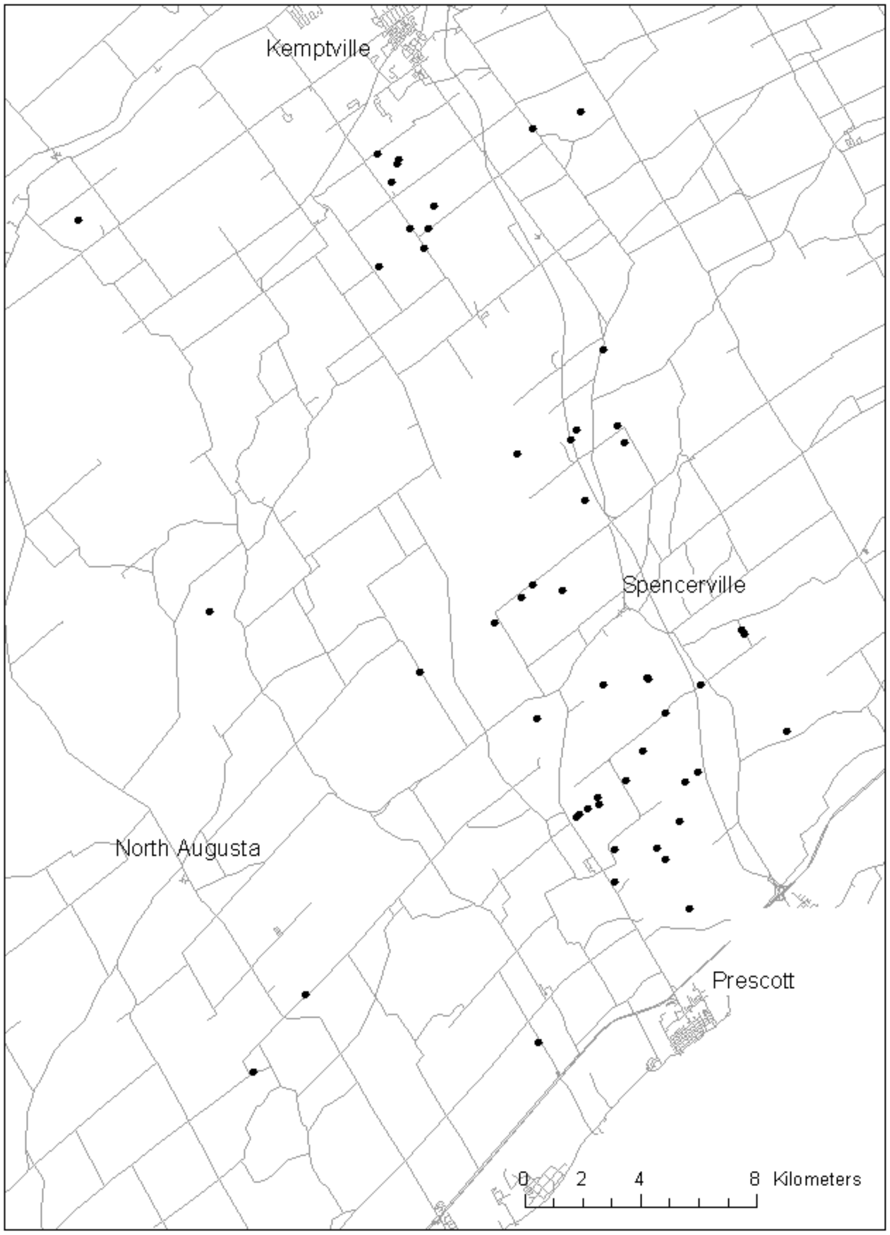
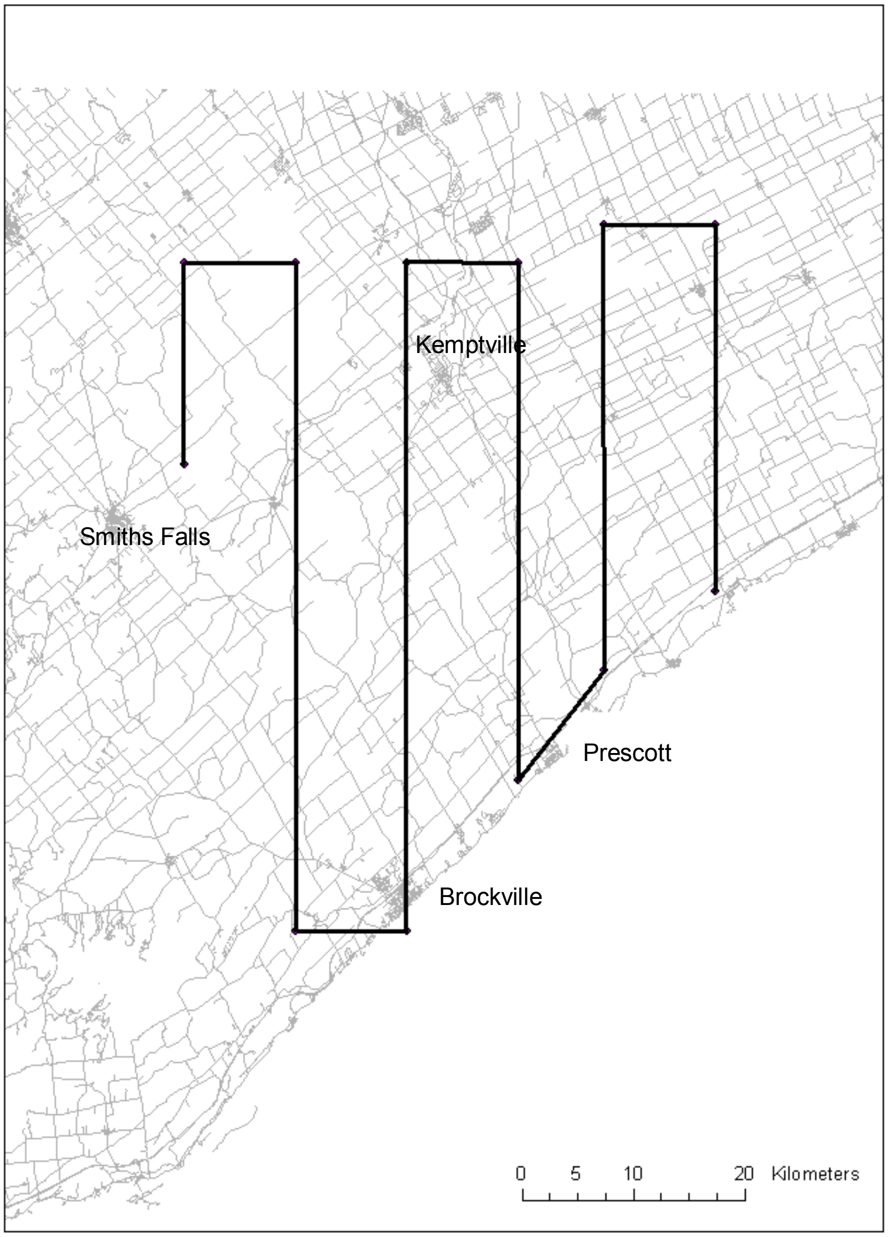
Appendices

Appendix 1. Locations of 61 successful fisher traps between February and October 2003, in Grenville County, Ontario.

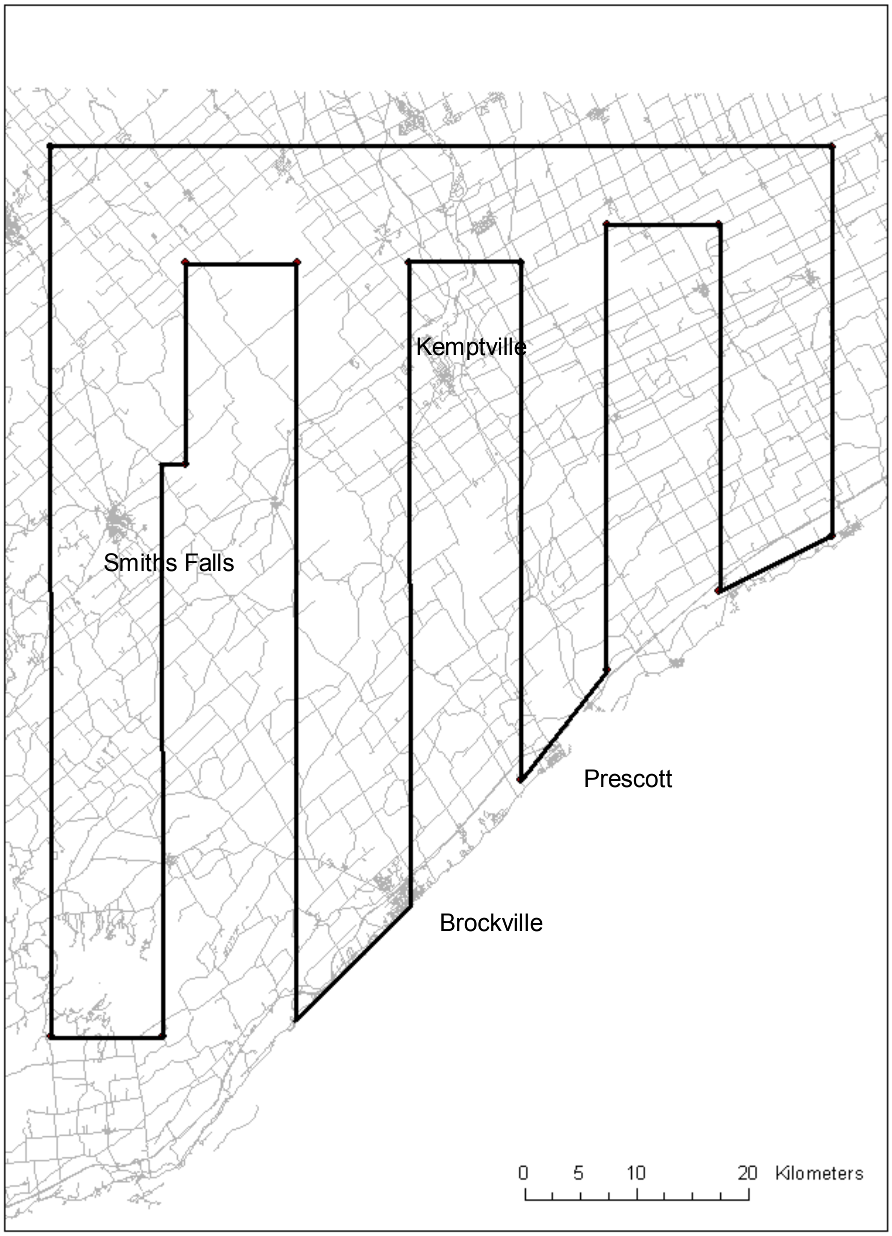


Appendix 2. Transect grids flown between July 7, 2003 and June 30, 2004.

a) Transect grid flown across study area on July 7, 2003 at 950 m. Fishers 155.119, 155.260, 155.919, 155.700, 155.160, 155.290, 155.179 and 155.420 were not found on this flight.

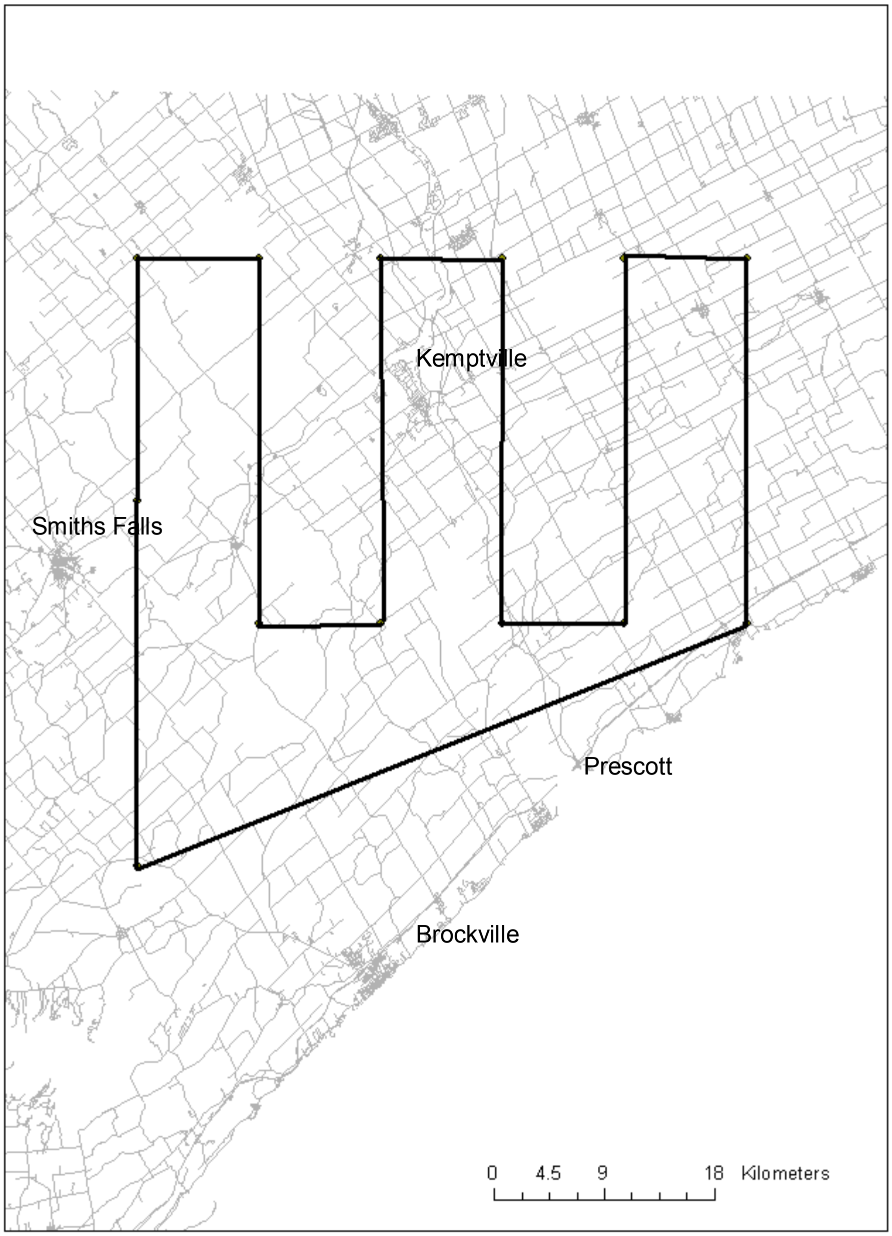


Appendix 2b. Transect grid flown across study area on July 14, 2003 at 950 m. Fishers 155.420, 155.179, 155.919, 155.260 and 155.160 were not found on this flight.

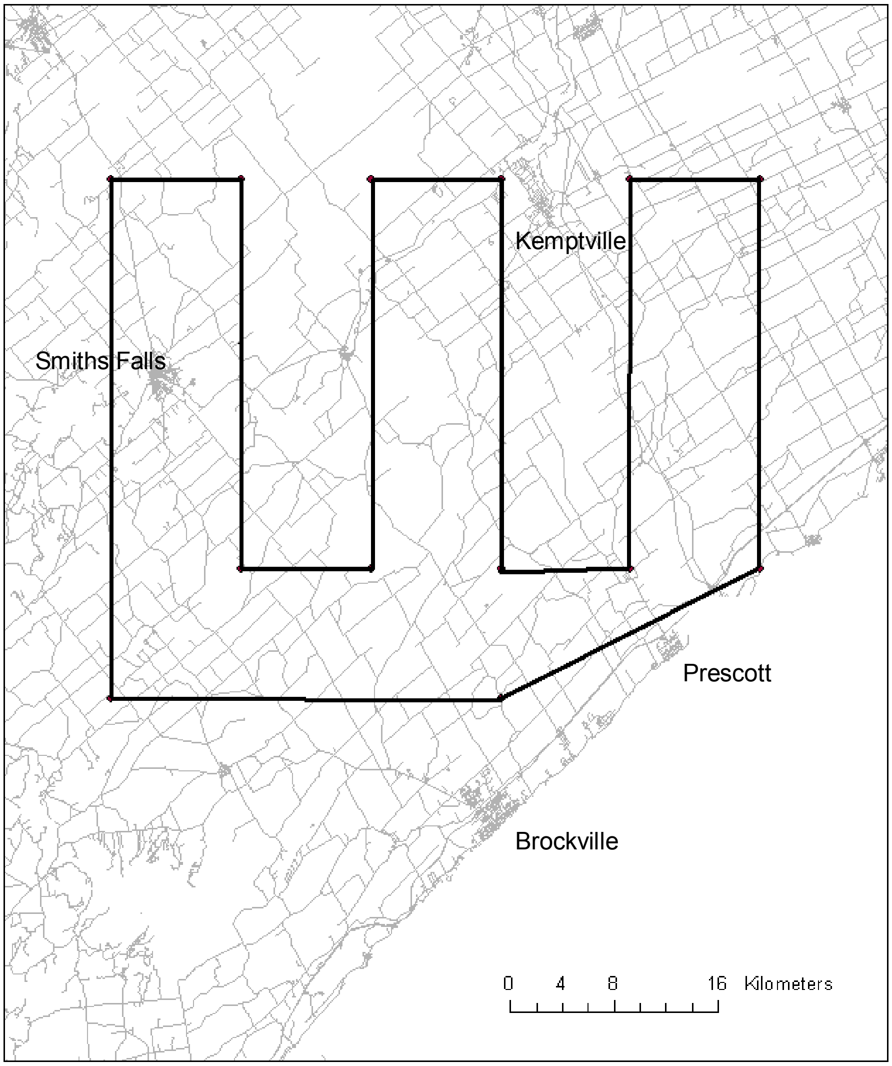


Appendix 2c. Transect grid flown across study area on October 9, 2003 at 750 m. Fishers 155.620, 155.179, 155.420, 155.290, 155.119 were not found on this flight.

Appendix 2d. Transect grid flown across study area on May 31, 2004 at 750 m. Fishers 155.260, 155.899, 155.231, 155.620 or 155.919 were not found on this flight.



Appendix 2e. Transect grid flown across study area on June 30, 2004 at 750 m. Fisher 155.460 was not found on this flight.



Appendix 3. Identification information for 61 fishers radio-collared in eastern Ontario.

Frequency	Sex	Age			Mass	Date[4]	Location[5]		Eartag	
		SC[1]	Tooth[2]	Age[3]	(kg)		Northing	Easting	Left	Right
155.440	M	A	/		6.5	30-Jan-03	4959675	456804	627250	627249
155.840	F	A	/		2	3-Feb-03	4958407	455111	627248	627247
155.399	M	A	A		5	5-Feb-03	4980306	448778	627246	627247
155.480	M	A	A	1.5	5	5-Feb-03	4977397	448343	627243	627242
155.680	F	A	A		2.5	5-Feb-03	4958478	455198	627241	627240
155.760	F	A	A		2	5-Feb-03	4955243	459008	627239	627238
155.420	F	J	J		2	5-Feb-03	4950627	453827	627237	627236
155.519	F	J	J		3.5	5-Feb-03	4959620	458838	627230	627234
155.599	M	A	A	1.5	4.5	5-Feb-03	4956943	458179	627228	627227
155.500	M	A	A	3.5	4	5-Feb-03	4958251	458641	605524	627235
155.581	F	J	A		2	5-Feb-03	4960677	457415	627251	627252
155.659	F	A	A	2*	2.5	6-Feb-03	4971308	456746	627275	245110
155.639	F	A	A		2.5	6-Feb-03	4971385	454930	627224	627231
155.560	M	A	A		4.5	6-Feb-03	4956182	456435	627272	627273
155.139	F	A	J		2.5	6-Feb-03	4958407	455111	627274	627223
155.360	F	A	A		2.5	7-Feb-03	4969329	455408	627203	627232
155.179	F	J	J		2.5	7-Feb-03	4981261	448298	627229	627259
155.038	F	J			2.5	7-Feb-03	4957350	457900	303685	/
155.160	F	J	J		2.5	9-Feb-03	4965093	452312	627269	627268
155.260	F	A	A		3	9-Feb-03	4965954	453229	627267	627266
155.100	M	J	A		4.5	9-Feb-03	4974530	456056	627221	627220
155.000	M	A	U		4.5	12-Feb-03	4979000	438000	627257	627262
155.119	M	A	A		4.5	13-Feb-03	4964723	460888	627219	627218
155.020	M	A	A		4.5	13-Feb-03	4958829	455895	627217	627216
155.620	M	A			3	15-Feb-03	4965500	442500	627208	627207
155.779	M	A			4	16-Feb-03	4980923	448972	627298	627297
155.700	M	J	A	1.5***	4.5	17-Feb-03	4971750	455133	245130	627210
155.800	F	J	A		2.5	17-Feb-03	4949597	443986	627280	627279
155.820	F	J	A	1.5***	2.5	18-Feb-03	4952279	445821	627205	169928
155.860	F	A?	A		2.5	18-Feb-03	4958858	455872	627296	627295
155.880	M	A	A	3.5**	6.5	19-Feb-03	4962966	459377	627278	627277
155.740	F	A	J		2	19-Feb-03	4959620	458838	627213	627212
155.540	F	J	J		2.5	20-Feb-03	4957274	456412	627294	627293
155.059	M	J	J		3.5	21-Feb-03	4981073	448992	627258	627209
155.380	F	A	J		2.1	22-Feb-03	4971917	456527	627265	627264
155.231	F	A	J		2	22-Feb-03	4971917	456527	627254	627253
155.300	F	A	A		3	22-Feb-03	4961994	458181	627282	627281
155.980	F	A	A		2.5	22-Feb-03	4963139	457570	627290	627289
155.919	F	J	J		2.5	22-Feb-03	4963139	457570	627292	627291
155.080	F	A	A		2	22-Feb-03	4964872	460798	627215	627214
155.959	F	J			2.5	22-Feb-03	4958407	455111	627255	/
155.211	M	A	A	1.5*	5.5	22-Feb-03	4962976	456022	627256	/
155.460	M	A	A	2*	6.5	26-Feb-03	4978014	449887	627261	627263
155.940	F	A	A		2.5	26-Feb-03	4963218	457522	627286	627285
155.899	F	J	J		2.25	28-Feb-03	4971385	454930	627284	627283
155.339	M	A	A	2**	4	28-Feb-03	4959067	455826	246889	627211
155.999	M	A	A		5	1-Mar-03	4959955	459281	627288	627287
155.290	M	A	A		5.5	2-Mar-03	4958478	455198	627300	627299
155.720	M	J	A		3.5	2-Mar-03	4978696	449384	627260	627222
155.320	F	A	A	1.5***	2.5	2-Mar-03	4961349	462338	240406	627204

155.840b	F		J		2	4-Jun-03	4982151	453596	245357	245356
155.519b	F				2.5	4-Jun-03	4961781	453774	245361	245362
155.339b	F		A	1*	1.5	4-Jun-03	4978706	450041	245362	245363
155.020b	F	A			2.5	10-Jun-03	4982695	455287	245359	245364
155.959b	F	A	A		2.5	10-Jun-03	4982151	453596	245365	245369
155.599b	M	A	A		5	11-Jun-03	4963418	449746	627233	245371
155.440b	M	A	A	1*	4	12-Jun-04	4979481	450221	245355	245373
155.779b	M	A	A		7	2-Oct-03	4958711	455501	245368	vrg032076
155.659b	F	A	J		4	10-Oct-03	4966212	454610	245372	vrg032027
155.300b	M	A	A		6	10-Oct-03	4966429	453631	245367	vrg032027
155.100b	F	A	A	1*	3.5	17-Oct-03	4970912	453084	245375	307007

[1] Age estimate from relative size of sagittal crest; A is adult (≥1 year old), J is juvenile (<1 year old)

[2] Age estimate from first premolar; size of pulp cavity; A is adult (≥1 year old), J is juvenile (<1 year old)

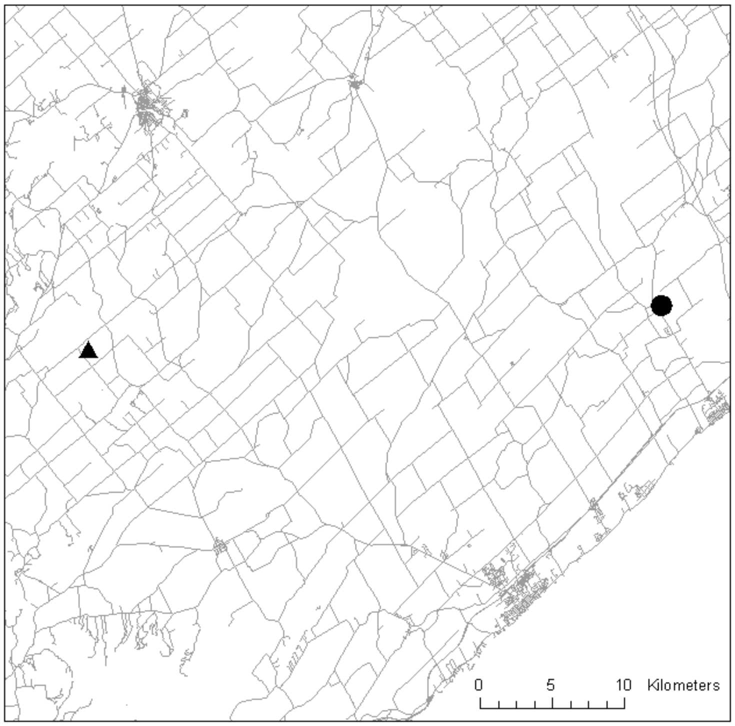
[3] Age, in years, at time of capture, as determined by cementum annuli; * beside age indicates the minimum age, ** indicates most probable age, *** beside age is minimum age based on previous capture by the Rabies Research Unit

[4] Date of first capture

[5] Location of first capture, in UTM, NAD 83 zone 18

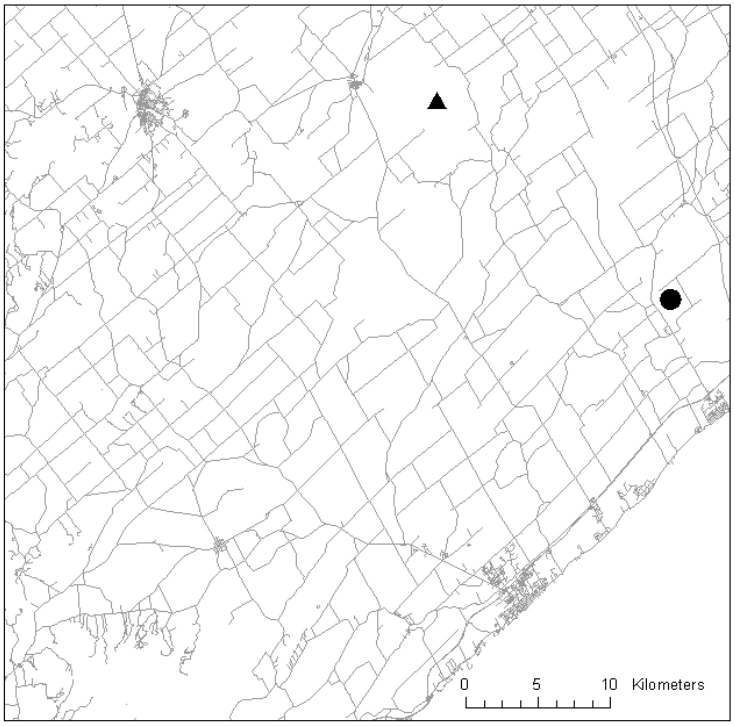
Appendix 4. Initial trap locations (●) and final locations (▲) of radio-collared fishers that dispersed. The scale of each map is the same. a) male 155.290, b) female 155.860, c) female 155.380, d) male 155.399, e) male 155.700, f) male 155.720, g) male 155.779.

a)

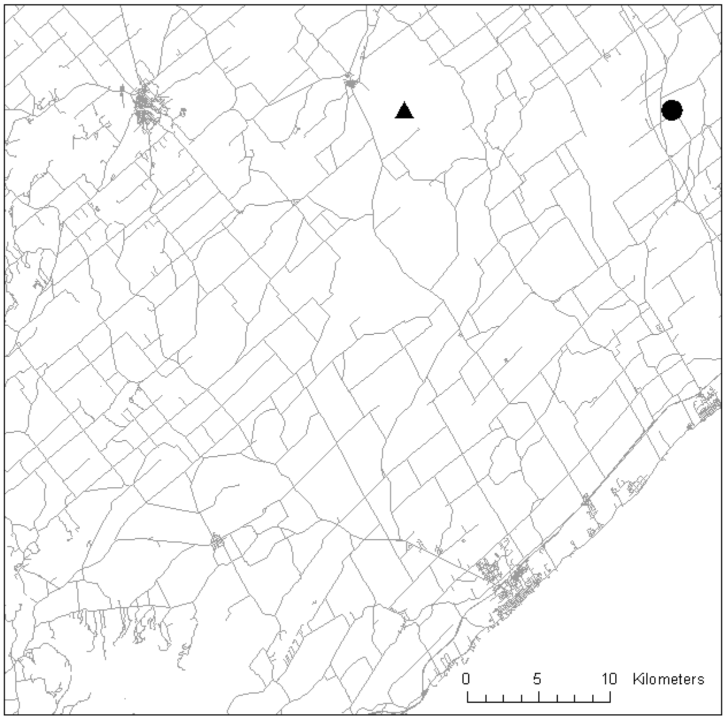


Appendix 4 (continued)

b)

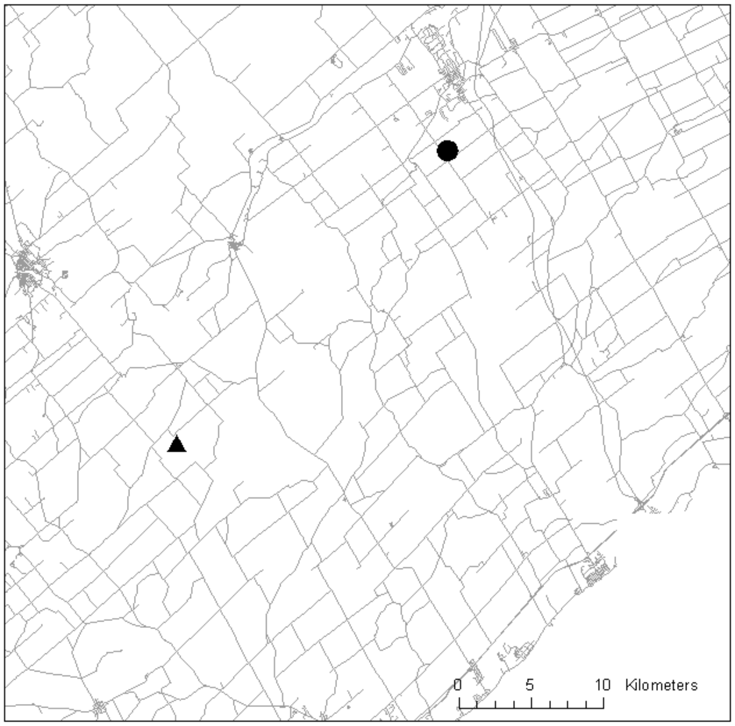


c)

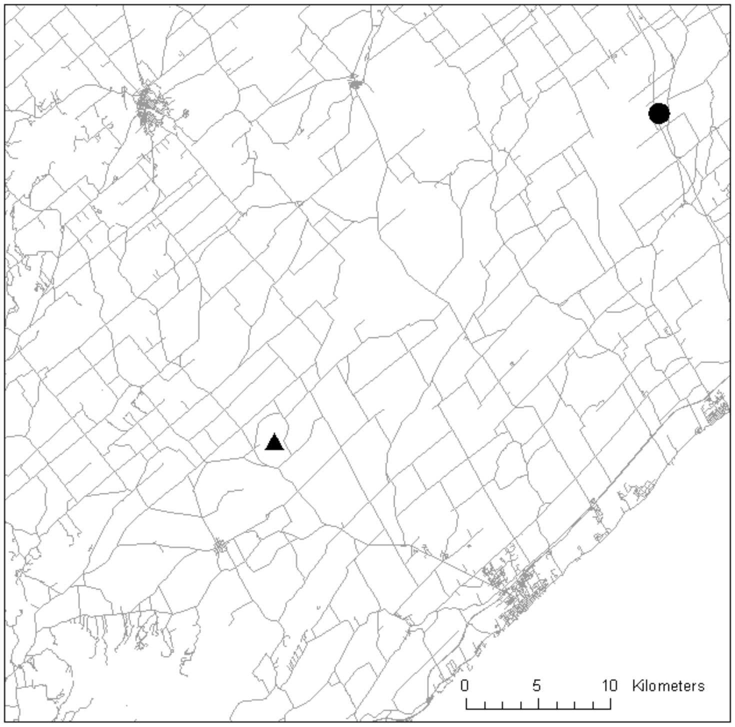


Appendix 4 (continued)

d)

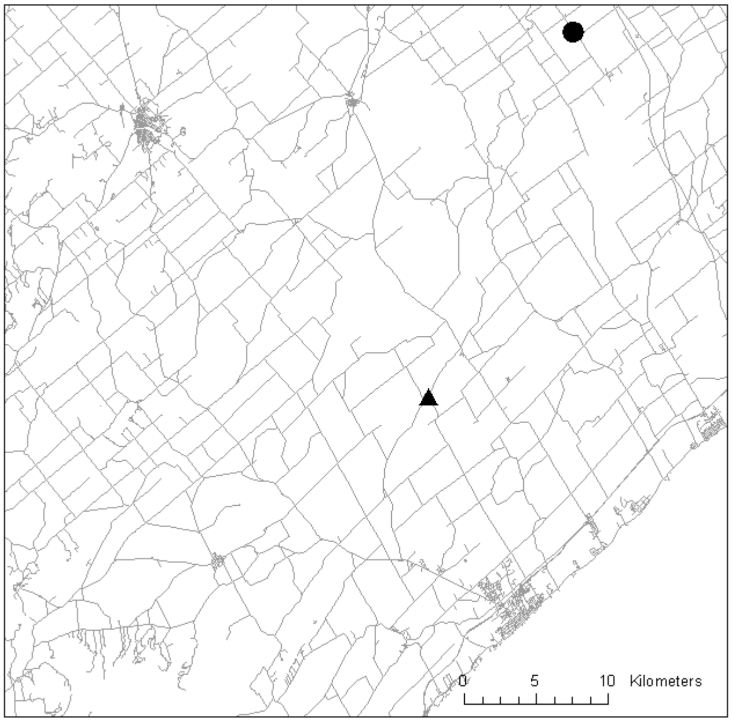


e)

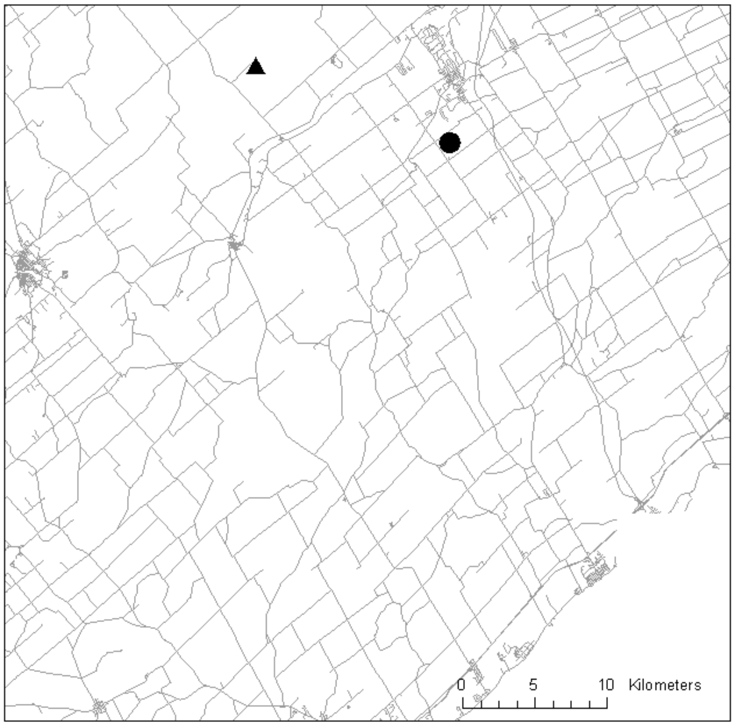


Appendix 4 (continued)

f)

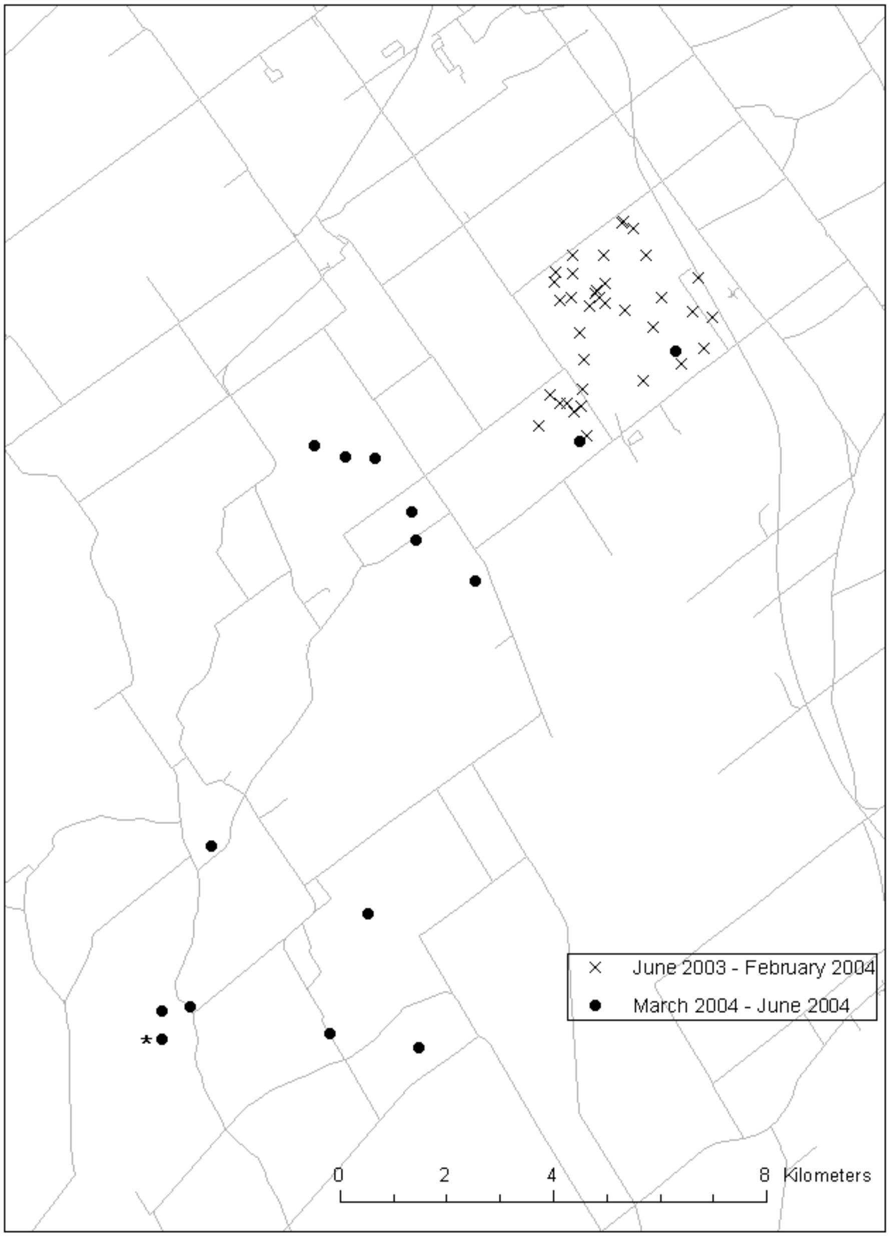


g)

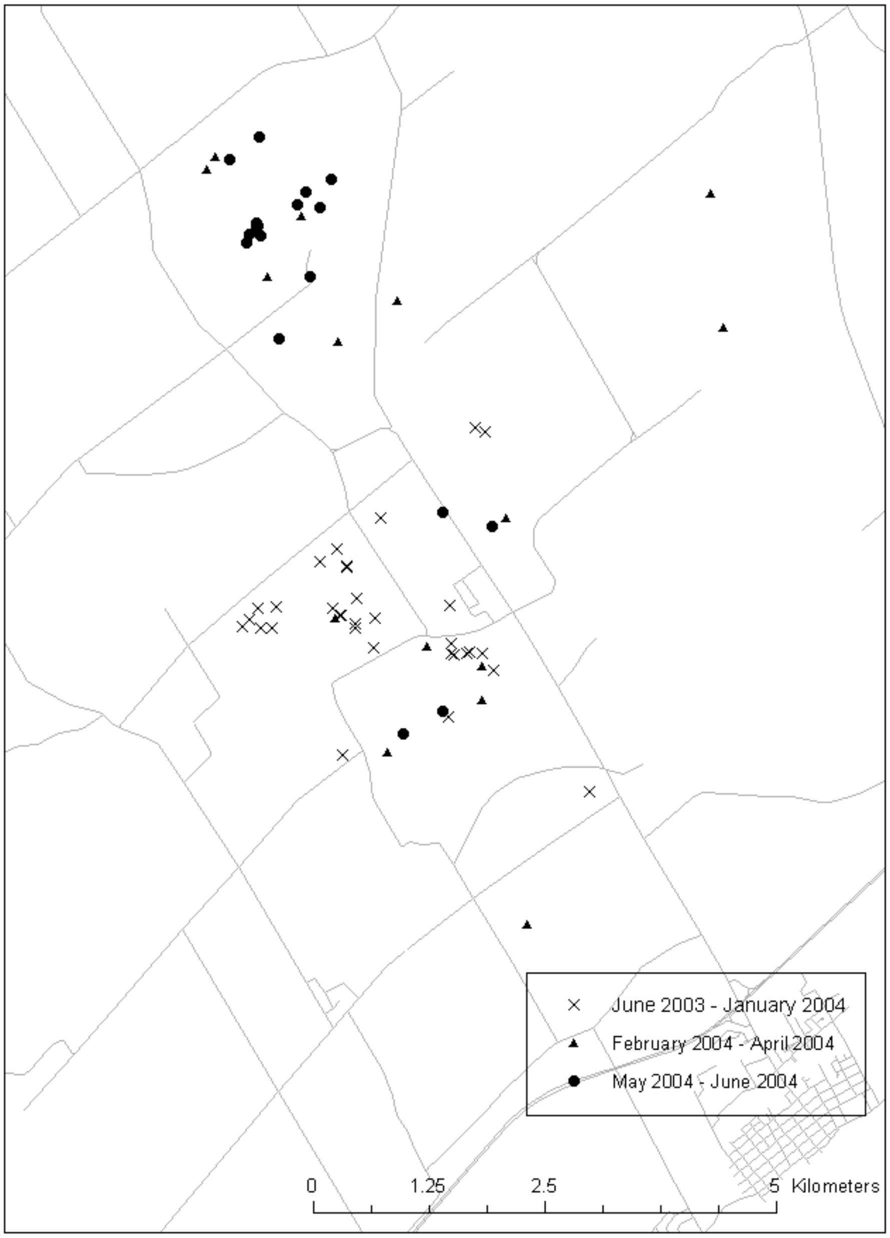


Appendix 5. Breeding season movements and home range shifts of fishers 155.460, 155.560, 155.020b and 155.980.

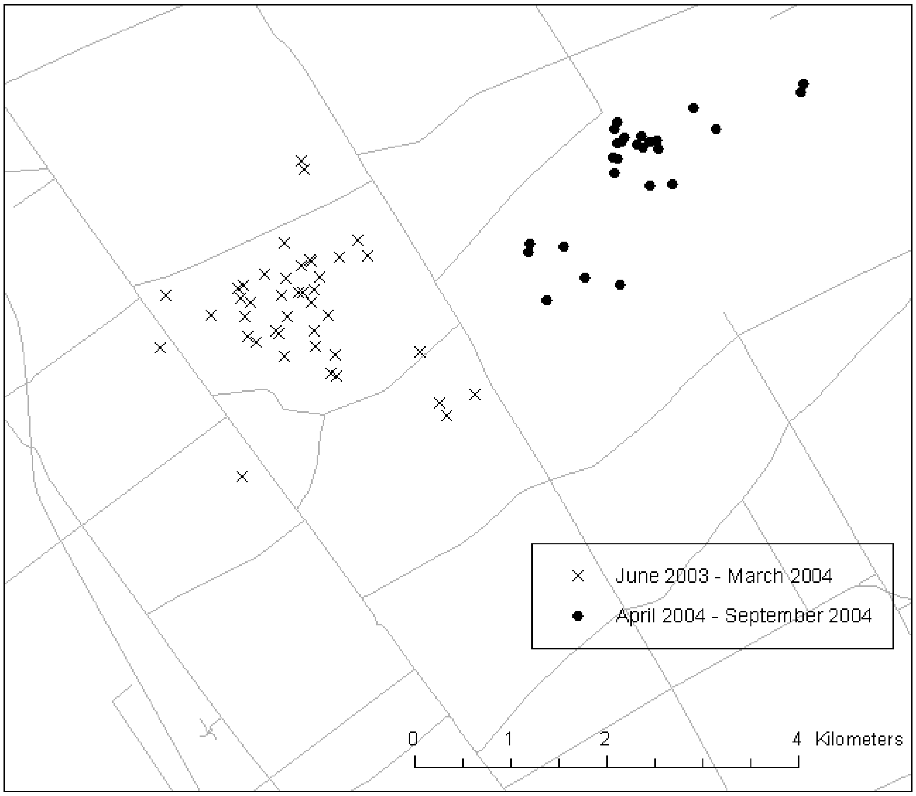
a) Locations for male fisher 155.460 between June 2003 and February 2004 (non-breeding season) and March and June 2004 (breeding season). Contact with 155.460 was lost after June 15, 2004; * denotes last observed location.



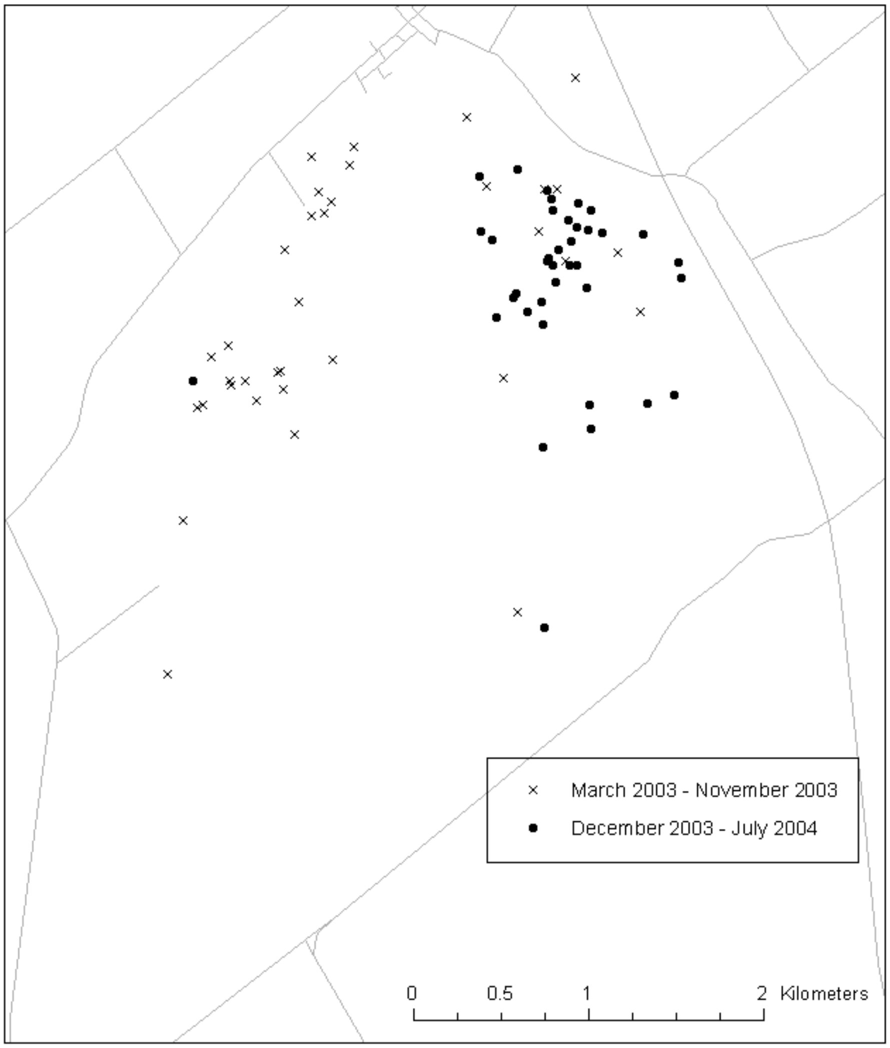
Appendix 5b. Locations for male fisher 155.560 between June 2003 and January 2004 (non-breeding season) and February and June 2004 (breeding season). Contact with 155.560 was lost after July 14, 2004.



Appendix 5c. Locations of female fisher 155.020b, depicting a shift in home range in April, 2004.

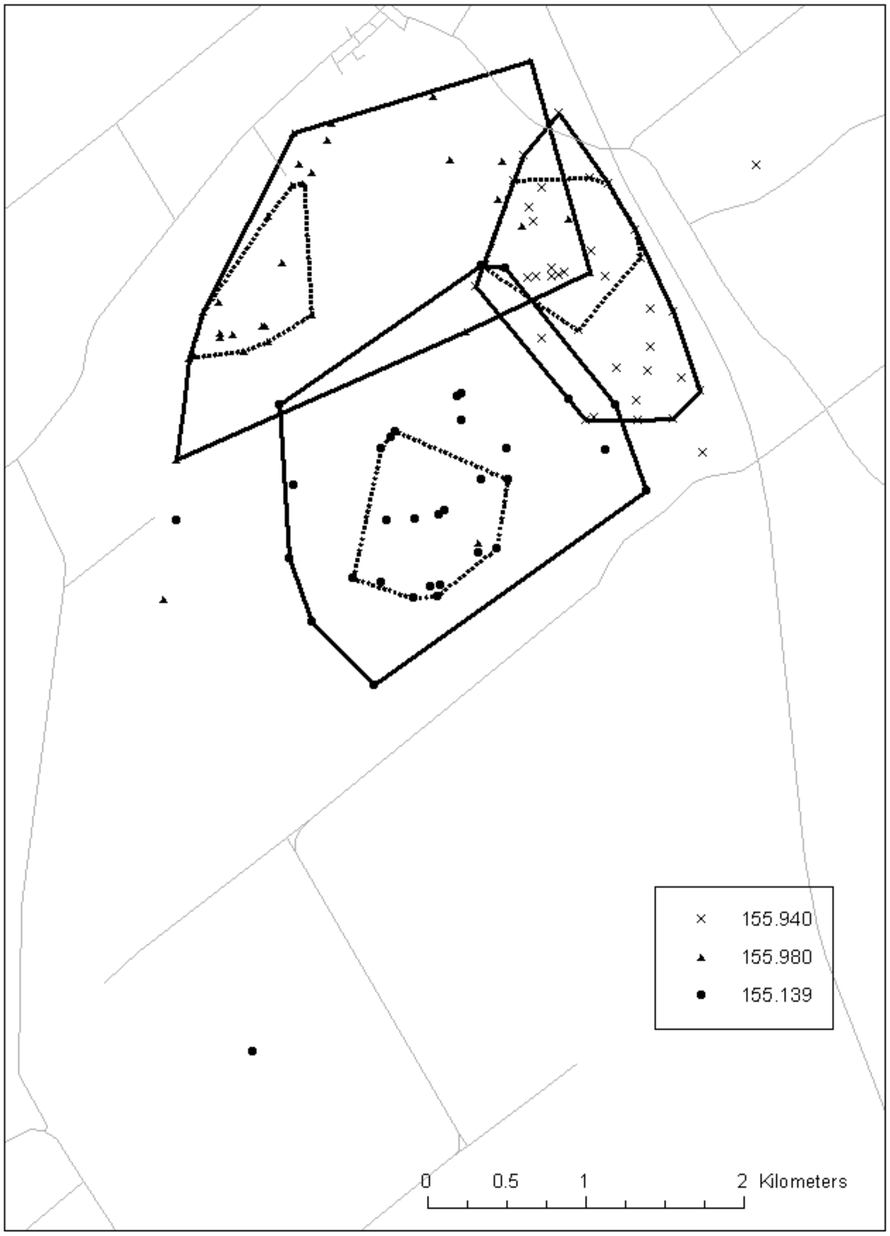


Appendix 5d. Locations of female fisher 155.980, depicting a shift in home range.

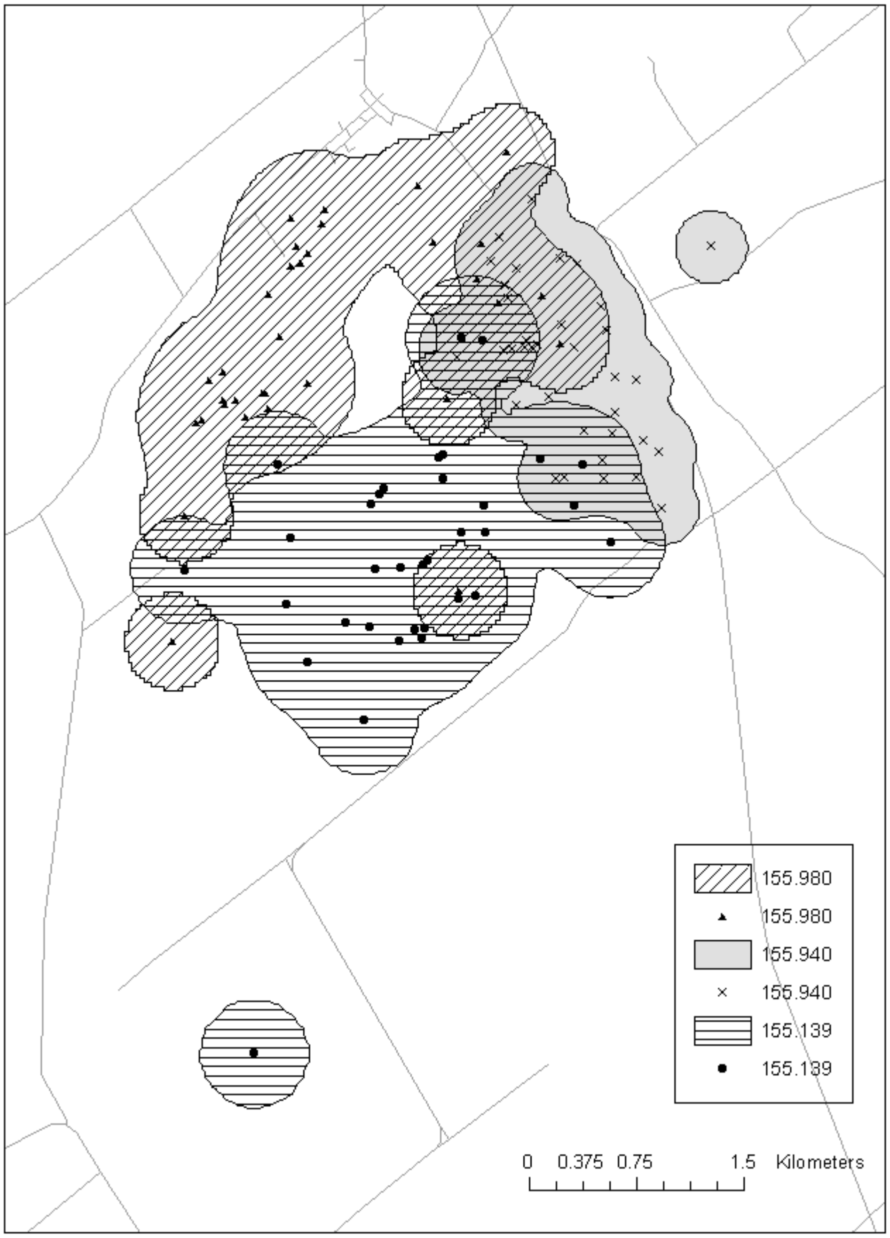


Appendix 6. Intrasexual overlap of home range and core area (MCP and kernel).

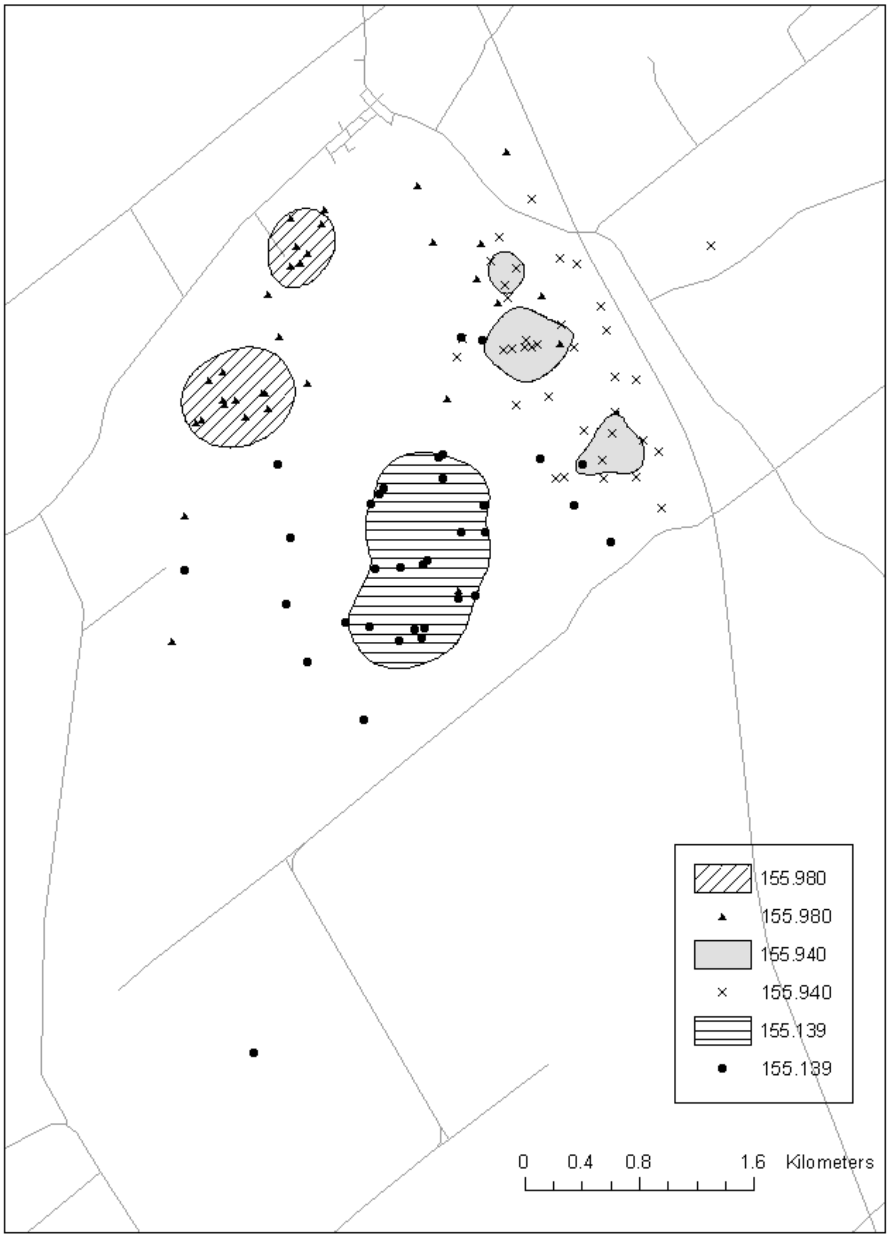
a1) Overlapping 95% (solid lines) and 50% (dashed lines) MCP home ranges of 3 female fishers (155.940, 155.980, 155.139) between March and November, 2003; a period when all 3 fishers were radio-tracked and before 155.980 shifted her home range.



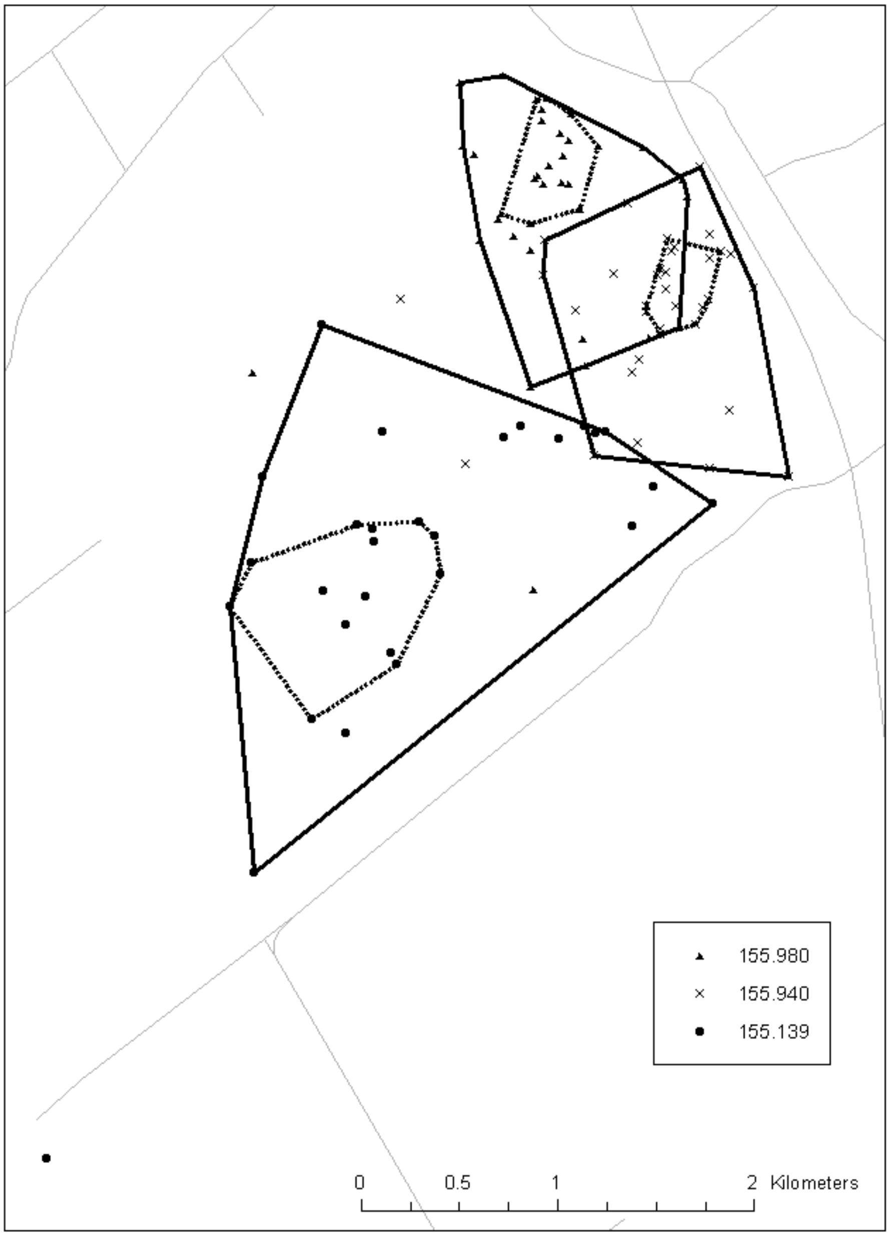
Appendix 6a2. Overlapping 95% kernel home ranges of 3 female fishers (155.980, 155.9408, 155.139) between March and November, 2003; a period when all 3 fishers were radio-tracked and before 155.980 shifted her home range.



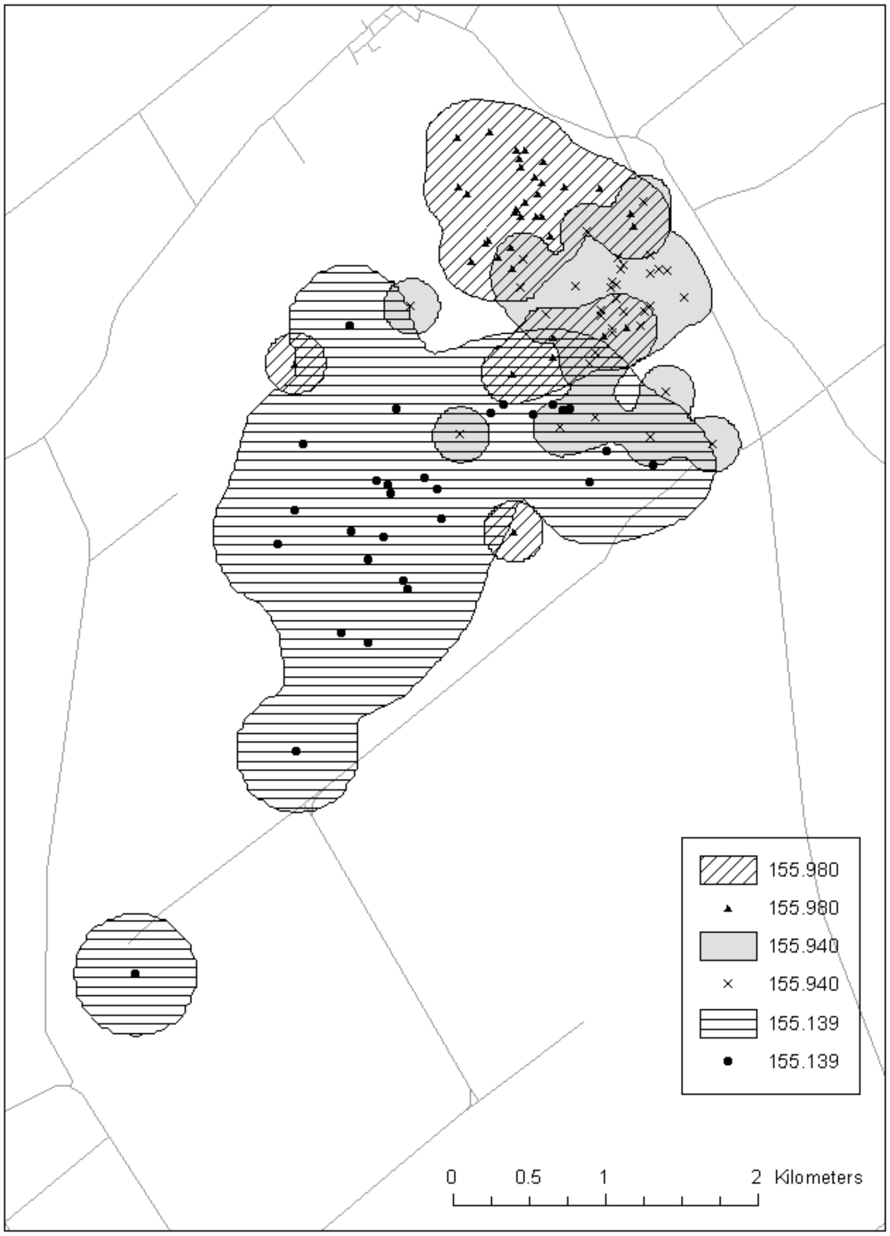
Appendix 6a3. 50% kernel home ranges of 3 female fishers (155.980, 155.940, 155.139) between March and November, 2003; a period when all 3 fishers were radio-tracked and before 155.980 shifted her home range.



Appendix 6b1. Overlapping 95% (solid lines) and 50% (dashed lines) MCP home ranges of 3 female fishers (155.980, 155.940, 155.139) between November, 2003 and June, 2004; a period when all 3 fishers were radio-tracked and after 155.980 shifted her home range.



Appendix 6b2. Overlapping 95% kernel home ranges of 3 female fishers (155.980, 155.9408, 155.139) between November, 2003 and June, 2004; a period when all 3 fishers were radio-tracked and after 155.980 shifted her home range.

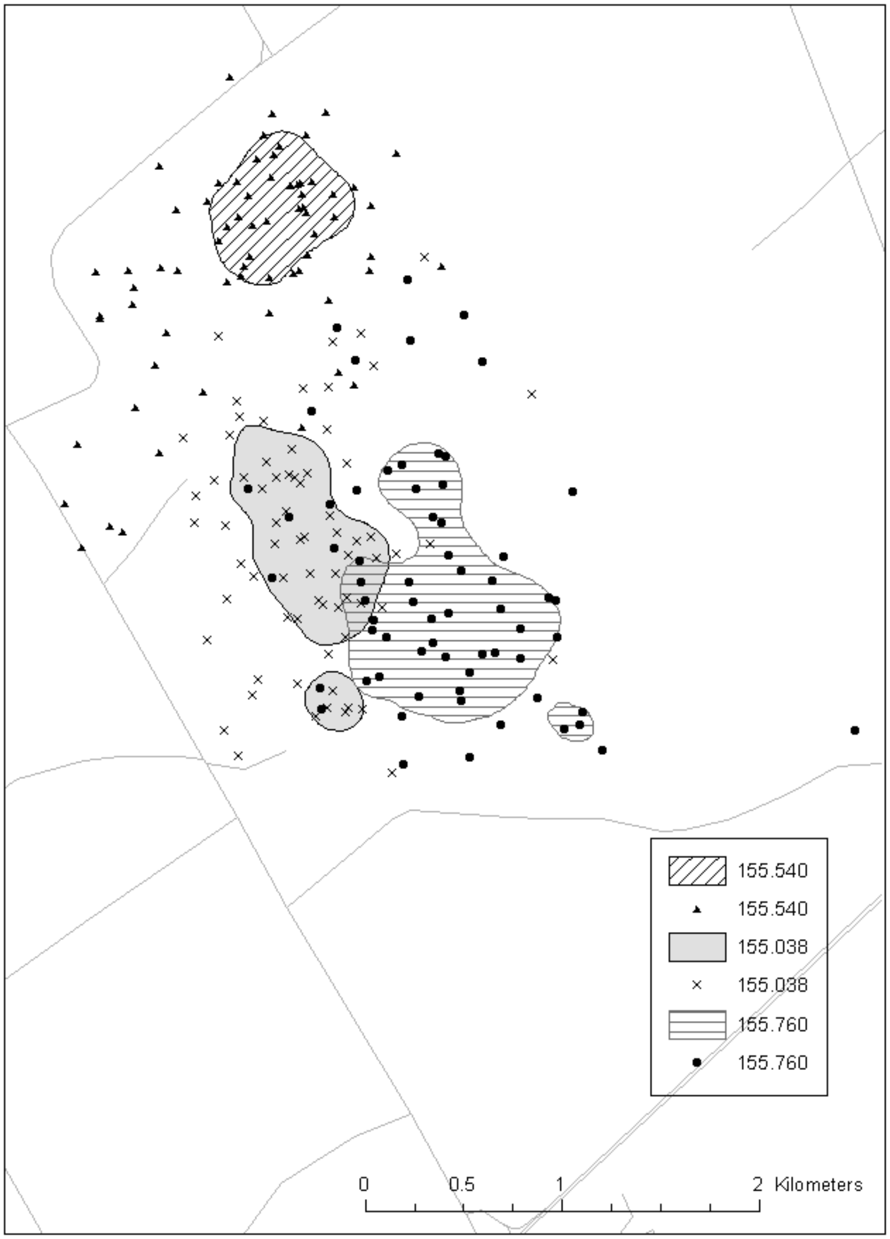


Appendix 6b3. 50% kernel home ranges of 3 female fishers (155.980, 155.940, 155.139) between November, 2003 and June, 2004; a period when all 3 fishers were radio-tracked and after 155.980 shifted her home range.

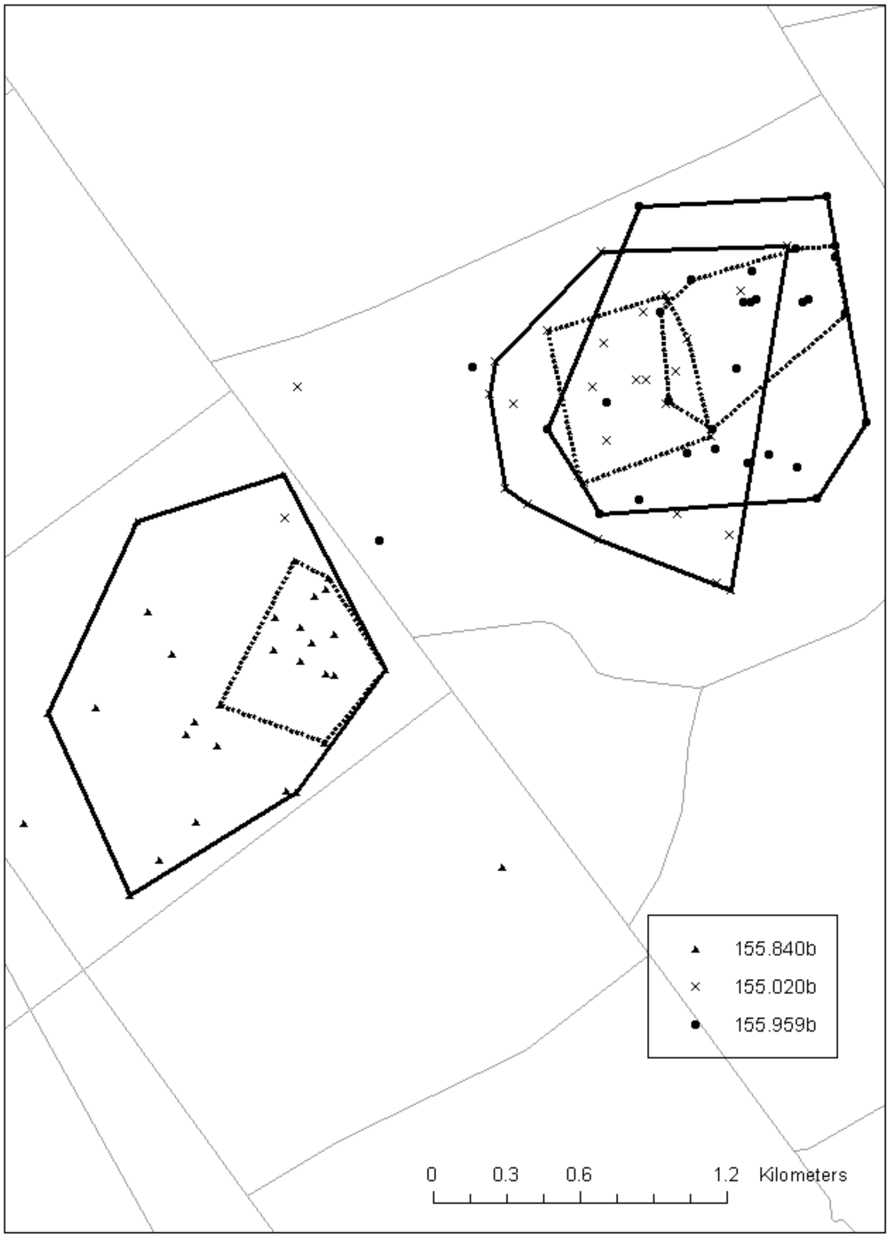
Appendix 6c1. Overlapping 95% (solid lines) and 50% (dashed lines) MCP home ranges of 3 female fishers (155.540, 155.038, 155.760) between February, 2003 and June, 2004; a period when all 3 fishers were radio-tracked.

Appendix 6c2. Overlapping 95% kernel home ranges of 3 female fishers (155.540, 155.038, 155.760) between February, 2003 and June, 2004; a period when all 3 fishers were radio-tracked.

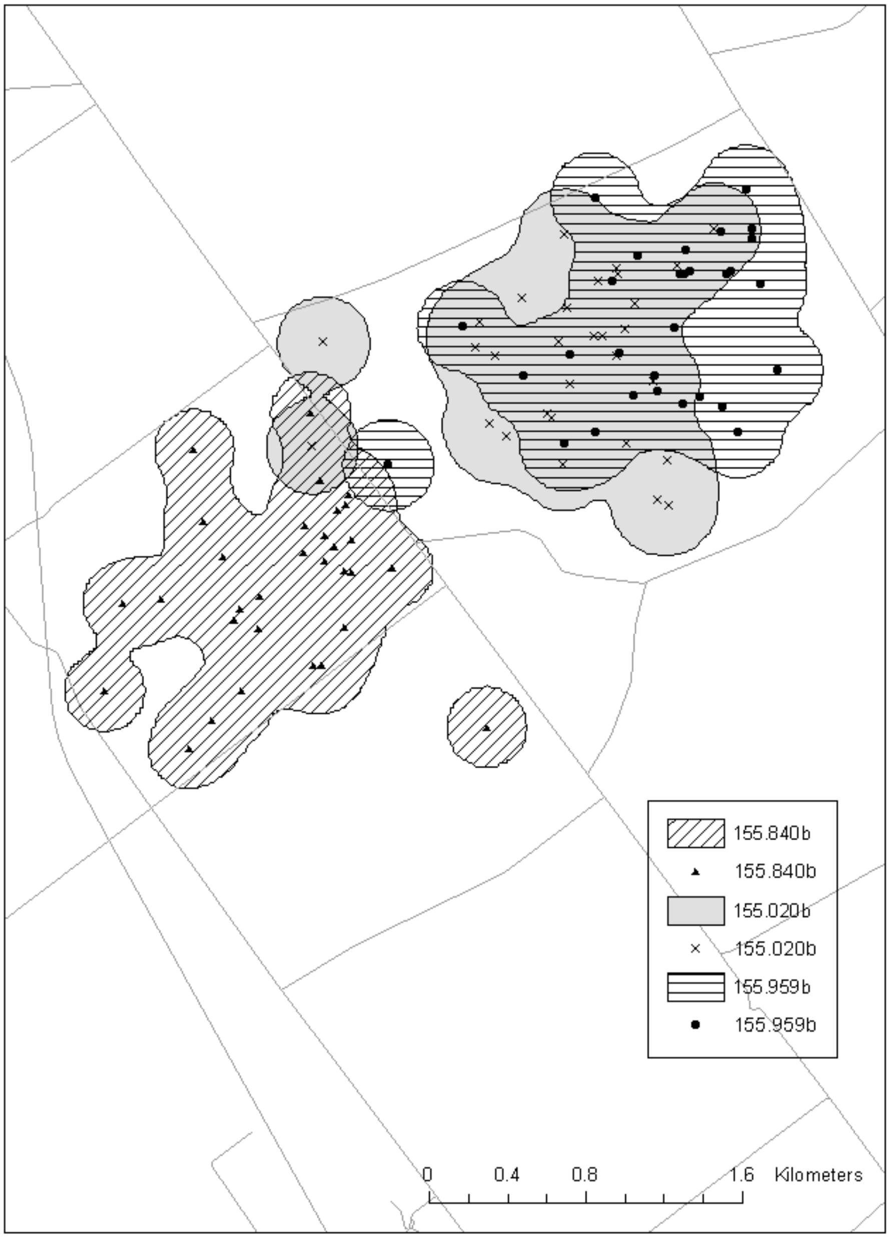
Appendix 6c3. Overlapping 50% kernel home ranges of 3 female fishers (155.540, 155.038, 155.760) between February, 2003 and June, 2004; a period when all 3 fishers were radio-tracked.



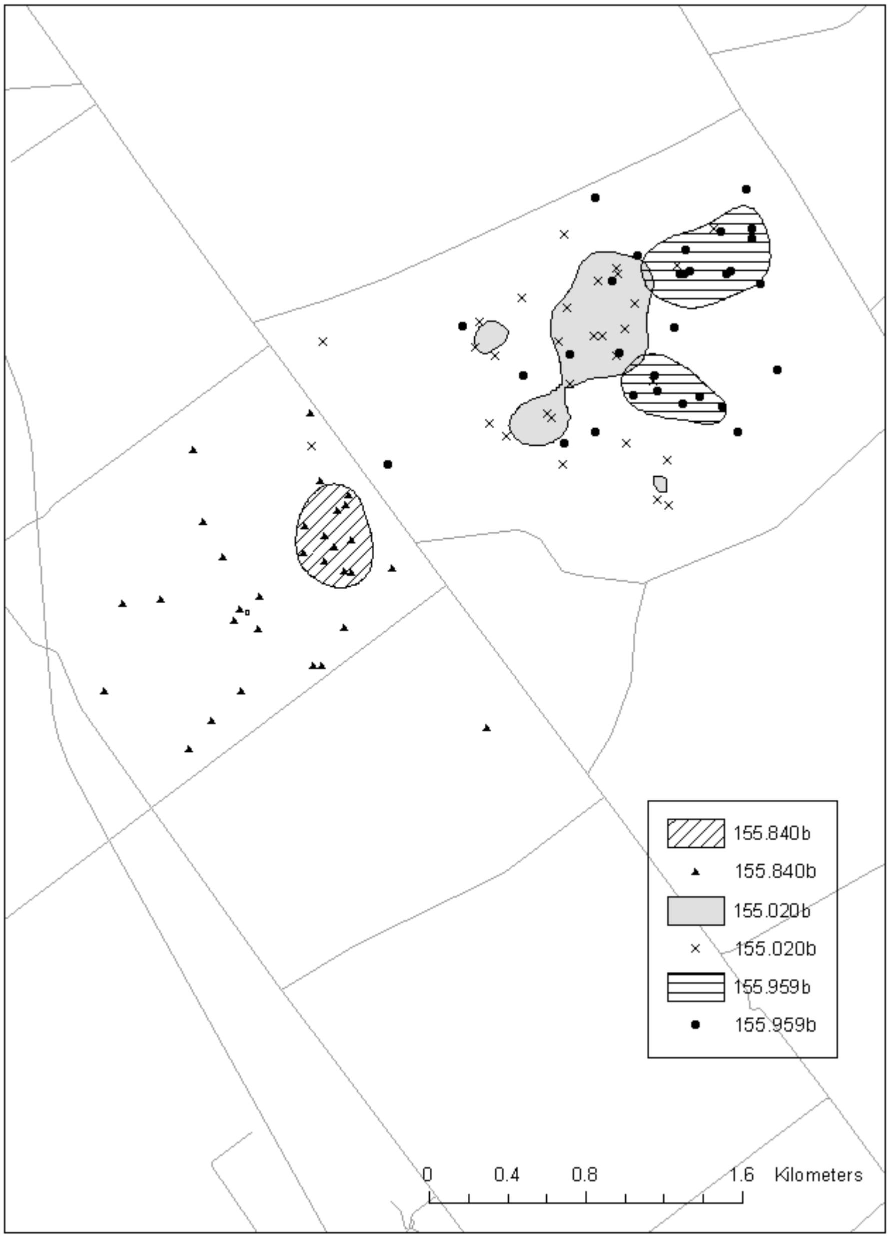
Appendix 6d1. Overlapping 95% (solid lines) and 50% (dashed lines) MCP home ranges of 3 female fishers (155.840b, 155.020b, 155.959b) between June and November, 2003; a period when all 3 fishers were radio-tracked.



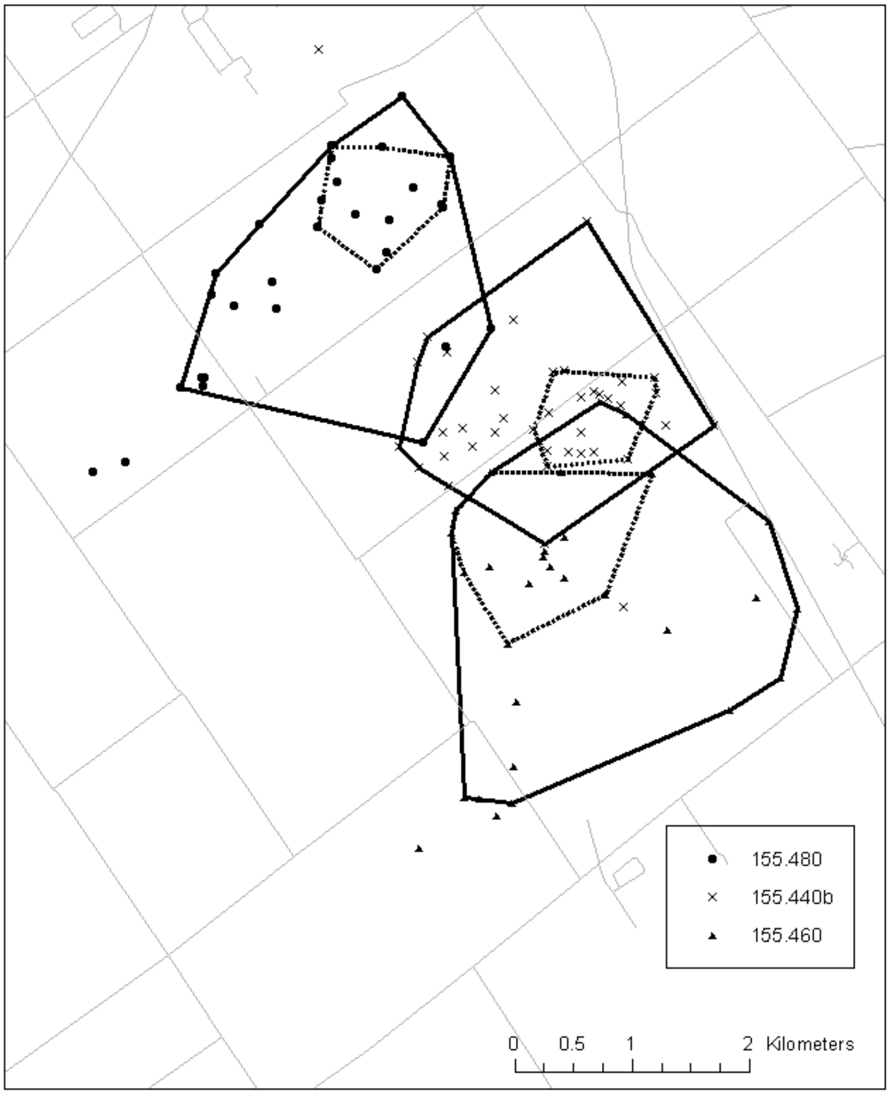
Appendix 6d2. Overlapping 95% kernel home ranges of 3 female fishers (155.840b, 155.020b, 155.959b) between June and November, 2003; a period when all 3 fishers were radio-tracked.



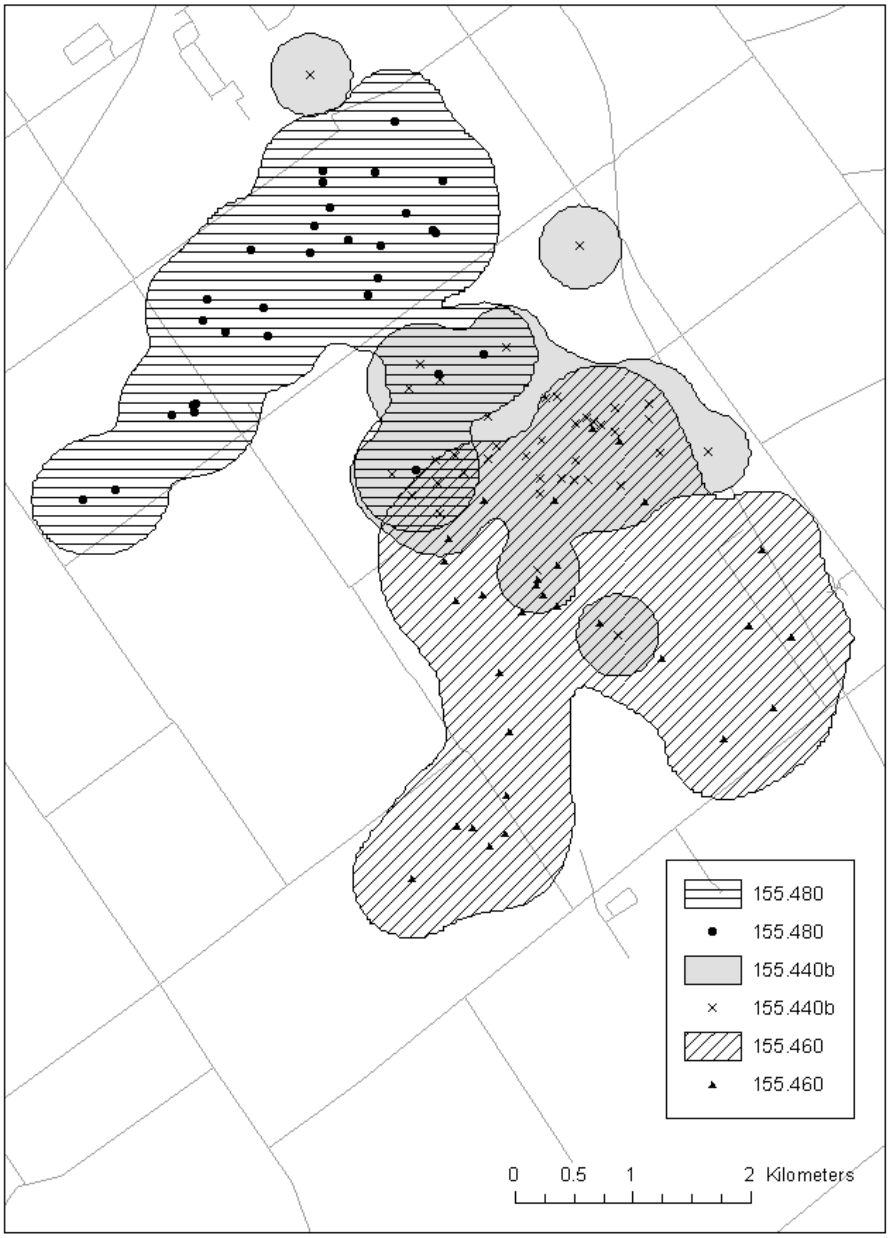
Appendix 6d3. Overlapping 50% kernel home ranges of 3 female fishers (155.840b, 155.020b, 155.959b) between June and November, 2003; a period when all 3 fishers were radio-tracked.



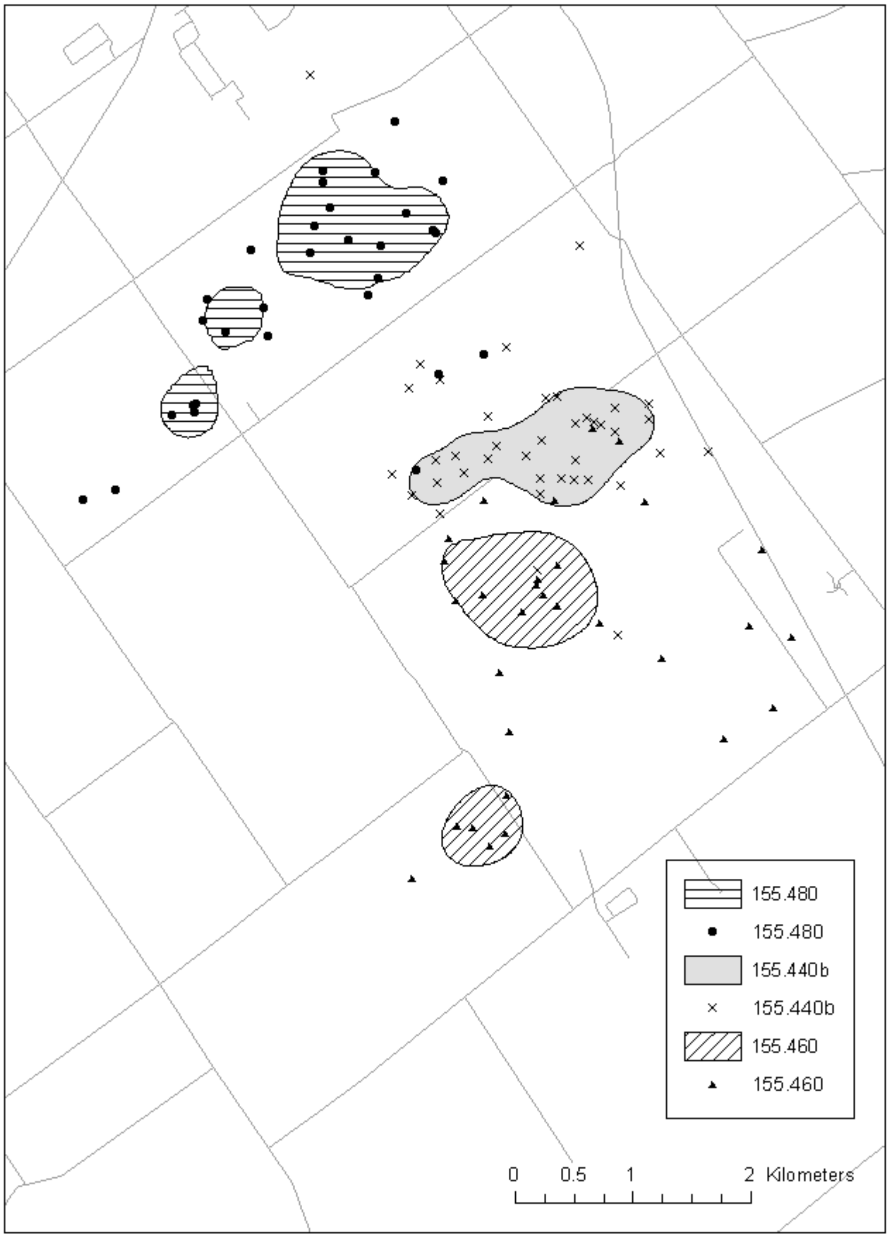
Appendix 6e1. Overlapping 95% (solid lines) and 50% (dashed lines) MCP home ranges of 3 male fishers (155.480, 155.440b, 155.460) between June, 2003 and February, 2004; a period when all 3 fishers were radio-tracked.



Appendix 6e2. Overlapping 95% kernel home ranges of 3 male fishers (155.480, 155.440b, 155.460) between June, 2003 and February, 2004; a period when all 3 fishers were radio-tracked.

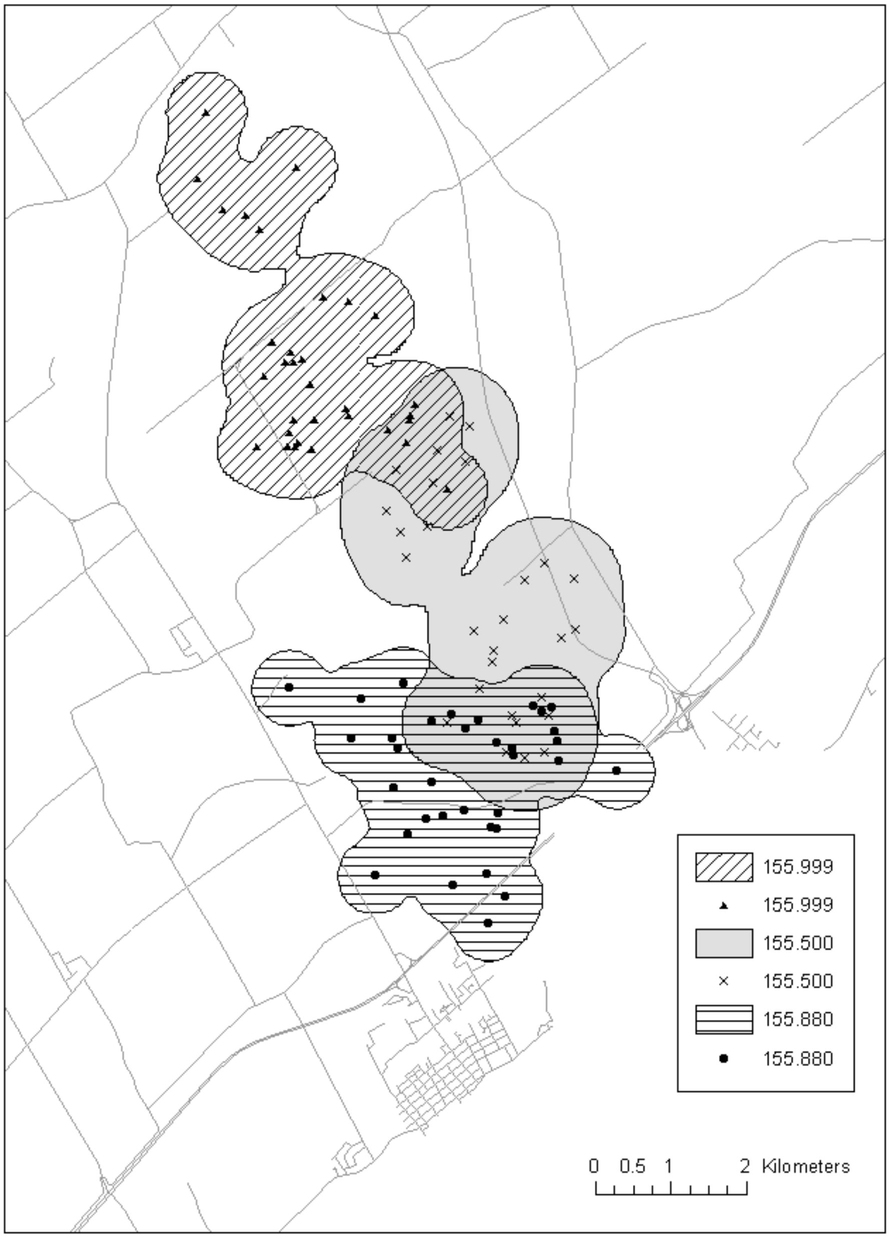


Appendix 6e3. 50% kernel home ranges of 3 male fishers (155.480, 155.440b, 155.460) between June, 2003 and February, 2004; a period when all 3 fishers were radio-tracked.



Appendix 6f1. Overlapping 95% (solid lines) and 50% (dashed lines) MCP home ranges of 3 male fishers (155.500, 155.880, 155.999) between June and December, 2003; a period when all 3 fishers were radio-tracked.

Appendix 6f2. Overlapping 95% kernel home ranges of 3 male fishers (155.500, 155.880, 155.999) between June and December, 2003; a period when all 3 fishers were radio-tracked.



Appendix 6f3. Overlapping 50% kernel home ranges of 3 male fishers (155.500, 155.880, 155.999) between June and December, 2003; a period when all 3 fishers were radio-tracked.

Appendix 7: Power of repeated measures ANOVA for detecting difference in home range size for annual home ranges.

The post-hoc power analysis for the repeated measures ANOVA, with method of home range estimation (95% MCP, 95% kernel, 50% MCP, 50% kernel) as the repeated factor and sex as the between subjects factor was completed using the "other F-tests" function in G*Power (Faul and Erdfelder 1992). The following formulae are from Buchner et al. (1997).

Between-subjects effects

a = number of levels of factor A (sex)
= 2

b = number of levels of (repeated) factor B (method)
= 4

$$\lambda = N * \left(\frac{m}{1 + (m-1) * rho} \right) * f^2$$

N = total sample size
= 15

m = number of levels of the repeated factor
= 4

rho = population correlation between the individual levels of the repeated factor
= 0.505

Since the sphericity assumption of the RM ANOVA requires that the correlation between levels of the repeated factor be the same for all pairs, the average correlation coefficient for the sample, 0.505 (± 0.304 SD) was used for the estimate of *rho*.

Note: the sphericity assumption is assumed to hold since the within-subjects p-values and Huynh-Feldt corrected p-values are similar (Systat 1996).

$$f^2 = (\text{effect size (between subjects)})^2$$

$$= \left(\frac{\text{effect variance}}{\text{error variance}} \right)^2$$

$$= \left(\frac{0.006}{3.330} \right)^2$$

$$= (1.802 \times 10^{-3})^2$$

$$= 3.246 \times 10^{-6}$$

Numerator df = a -1
= 1

Denominator df = N - a
= 13

G*Power calculates the power, (1 - β) as 0.050. Thus, at α = 0.05, the null hypothesis is accepted when, in fact, it is false, 95% of the time. The sample size is not large enough to detect the observed difference in home range overlap between sexes.

Within-subjects effects

There was enough power to detect a difference between methods.

Interaction of between and within-subjects effects

$$\lambda = \frac{N * m * f^2}{1 - rho}$$

f^2 = (effect size (within subjects))^2
= 0.1880

Numerator df = (a -1) * (b -1)
= 3

Denominator df = (N - a) * (b - 1)
= 39

G*Power calculates the power, (1 - β) as 0.237. Thus, at α = 0.05, the null hypothesis is accepted when, in fact, it is false, 76.3% of the time. The sample size is not large enough to detect an interaction between sex and methods.

Appendix 7 (continued)

References

Buchner, A., Erdfelder, E., & Faul, F. (1997). How to Use G*Power [WWW document]. URL http://www.psycho.uni duesseldorf.de/aap/projects/gpower/how_to_use_gpower.html

Faul, F. and Erdfelder, E. 1992. GPower: A priori, post-hoc, and compromise power analyses for MS-DOS (Computer program). Bonn, FRG: Bonn University, Department of Psychology.

Systat. 1996. Systat 6.0 for Windows: Statistics (manual). USA.

Appendix 8a. List of tree species recorded in forest resource inventory (FRI) data.

Black Ash
White Ash
Balsam Fir
Basswood
Beech
Grey Birch
Yellow Birch
White Birch
Red Cedar
White Cedar
Black Cherry
Black Walnut
Butternut
Elm
Hemlock
Hickory
Ironwood
Larch
Hard Maple
Soft Maple
Red Oak
White Oak
Jack Pine
Red Pine
Scots Pine
White Pine
Balsam Poplar
Poplar
Black Spruce
Red Spruce
White Spruce
Willow
Other Hardwood
All Spruce

Appendix 8b. Working groups and codes for FRI data.

A working group label is assigned to the tree species making up at least 60% of the stand. In mixed-species forest stands where a single species does not make up at least 60% of the stand, the label is applied to the “grouped” working group that comprises the highest percentage of the species composition (i.e. other conifers). If one of the members of this grouped working group comprises an equal or higher percentage of the composition than any other individual species, the label is assigned to the working group of that member.

Ash		A
	Black Ash	
	White Ash	
Balsam Fir		B
Other Hardwood		OH
	Other Hardwood	
	Basswood	
	Beech	
	Black Cherry	
	Butternut	
	Elm	
	Hickory	
	Ironwood	
	Willow	
	Black Walnut	
Grey Birch		GB
Yellow Birch		YB
White Birch		WB
Hemlock		HE
Cedar		CE
	Red Cedar	
	White Cedar	
Larch		L
Other Confier		OC
	Cedar	
	Larch	
Hard Maple		MH
Soft Maple		MS
Maple All		M
	Hard Maple	
	Soft Maple	
Oak		OR
	Red Oak	
	White Oak	
Jack Pine		PJ
Red Pine		PR

Appendix 8b (continued).

Scots Pine	PS
White Pine	PW
Balsam Poplar	PB
Poplar	PO
Black Spruce	SB
White Spruce	SW
White Spruce	
Red Spruce	
Spruce All	S
Black Spruce	
Red Spruce	
White Spruce	

Appendix 8c. Condensed FRI data. Wetland layer is from NRVIS (composed of swamp, bog, fen and marsh).

- Coniferous
  - Balsam Fir
  - Hemlock
  - Red Cedar
  - White Cedar
  - Larch
  - Jack Pine
  - Red Pine
  - Scots Pine
  - White Pine
  - Black Spruce
  - Red Spruce
  - White Spruce
- Deciduous
  - Black Ash
  - White Ash
  - Other Hardwood
  - Basswood
  - Beech
  - Black Cherry
  - Butternut
  - Elm
  - Hickory
  - Ironwood
  - Willow
  - Black Walnut
  - Grey Birch
  - Yellow Birch
  - White Birch
  - Hard Map
  - Soft Maple
  - Red Oak
  - White Oak
  - Poplar

Appendix 9. 15 and 28-class provincial-scale Ontario land cover (Landsat TM).

15 class	28 class
Water	Water
Marshes	Intertidal marsh
	Supertidal marsh
	Freshwater coastal marsh/inland marsh
Open wetlands	Open fen
	Open bog
Treed wetlands	Treed fen
	Treed bog
	Deciduous swamp
	Coniferous swamp
Tundra heath	Tundra heath
Dense deciduous forest	Dense deciduous forest (at least 80% deciduous)
Dense coniferous	Dense coniferous forest (at least 80% coniferous)
	Coniferous plantation
Mixed forest	Mixed forest, mainly deciduous (more than 50% deciduous)
	Mixed forest, mainly coniferous (more than 50% coniferous)
Sparse forest	Sparse coniferous forest (30-40% canopy closure, 80% coniferous)
	Sparse deciduous forest (30-40% canopy closure, 80% deciduous)
Early successional forest	Recent cutovers (clear cut areas less than 10 years of age)
	Recent burns (10 years of age)
Successional forest	Old cuts and burns (>10 years of age)
Mine tailings, quarries, bedrock outcrops, mudflats	Mine tailings, quarries and bedrock outcrops
	Coastal mudflats
Settlement and developed land	Settlement and developed land
Agriculture	Pasture and abandoned fields
	Cropland
	Alvar
Unclassified	Unclassified

Appendix 10. 95% kernel home ranges of 16 female and 10 male fishers and the positions of correctly and incorrectly classified ground truth locations (from FRI data).

Appendix 11. Documentation of fisher kits found on May 20, 2004 (offspring of female fisher 155.980).

Description of cavity tree

Location: 4964340N, 457626E, NAD 83, Zone 18

Tree species: Black ash (*Fraxinus nigra*)

Diameter at breast height: 53 cm

Condition of tree: standing live

Decay class: 1(alive) (Maser et al. 1979)

Location of cavity in tree: middle/top, 275 cm from ground

Aspect of cavity: 240°, facing up

Dimensions of cavity: 9 cm high, 10 cm wide, 35 cm deep

Type of cavity: decayed branch-hole

Surrounding forest: mostly deciduous

Description of kits

2 fisher kits were found weighing 600 g each. They had a thin layer of grey fur, their eyes were starting to open and they were unable to walk. Based on weight, these kits were likely 50 days old (Powell 1993). Therefore, they were born around April 1st, 2004. The sex of the kits is unknown.

References

Maser, C., Anderson, R.G., Cromack, K. Jr. Williams, J.T. and Martin, R.E. 1979. Dead and down woody material. *In* Wildlife habitats in managed forests of the blue mountains of Oregon and Washington. Ed. Thomas, J.W. US Department of Agriculture, Forest Service Handbook 553. Pg 78-95.

Powell, R.A. 1993. The fisher: life history, ecology and behavior, second edition. University of Minnesota Press, Minneapolis, Minnesota, USA, 237 pp.

Appendix 12. Critical values for the exponential distribution.

α	0.05	0.1	0.15	0.2	0.25	0.3	0.35	0. 4	0.45	0.5
value	2.996	2.302	1.897	1.609	1.386	1.204	1.050	0.916	0.798	1.693

The cumulative distribution is the probability that the variable takes a value less than or equal to x; it is the area under the probability density function at the value x. The standard exponential distribution has the equation

$$f(x) = e^{-x}$$

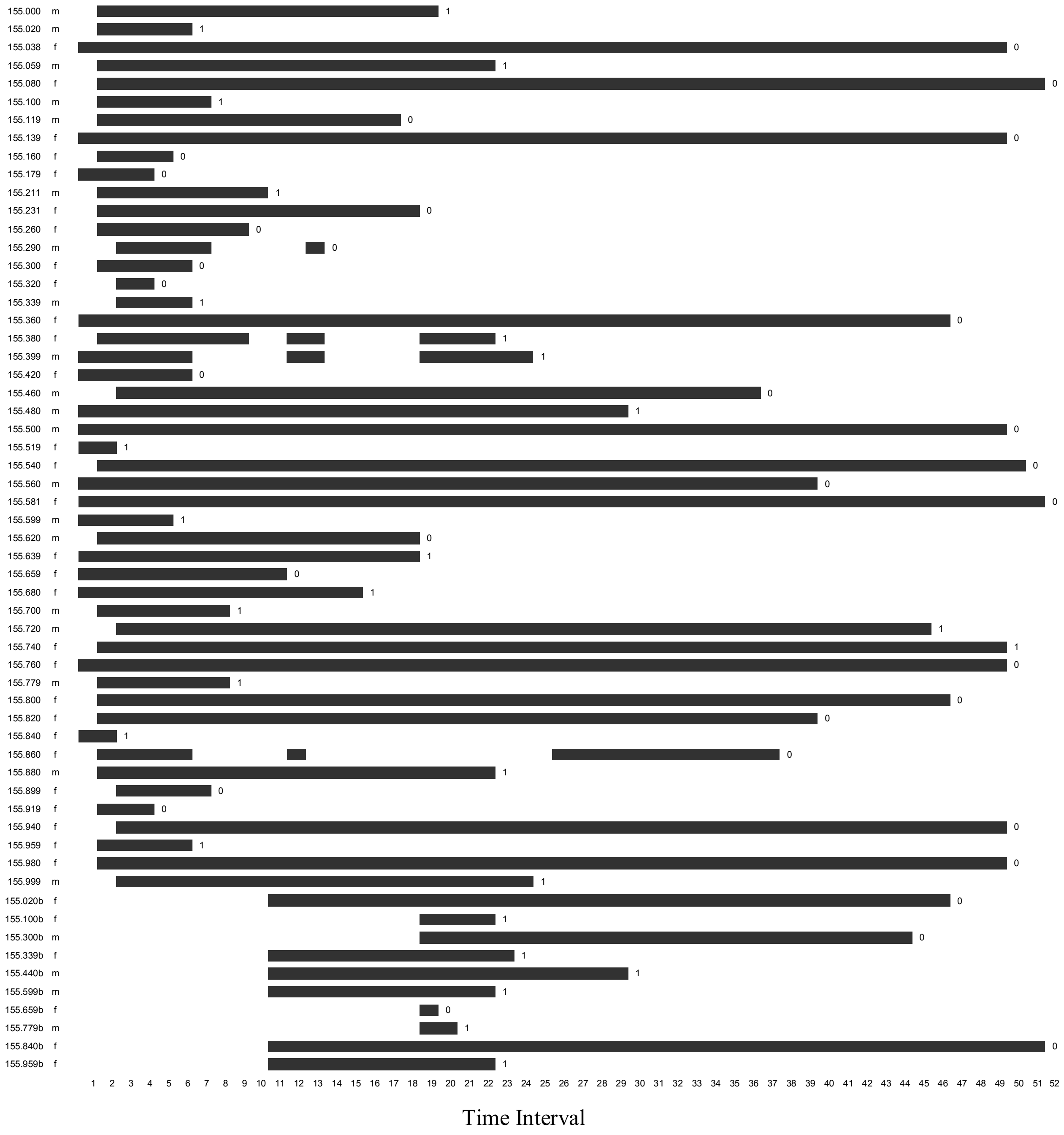
and the cumulative distribution function for the standard exponential distribution is

$$F(x) = 1 - e^{-x}.$$

To create the table of critical values for the exponential distribution, we are interested in those values of x for which some percentage of the area under the exponential curve is contained. For example, to calculate the value for which 95% of the area under the exponential curve is contained,

$$\begin{aligned} F(x) &= 1 - e^{-x} \\ 0.95 &= 1 - e^{-x} \\ \ln 0.05 &= -x \\ x &= 2.996. \end{aligned}$$

Appendix 13. Time until death (1) or censoring (0) of 59 fishers in eastern Ontario from February 2003 (interval 1) until January 2005 (interval 51). Each time interval represents 2 weeks.



Appendix 14. S-Plus code for various survival functions

S code for Kaplan-Meier estimate:

```
fit1<-coxph(Surv(enter,exit,event)~1,data =
survivalanalysis1)
> summary(survfit(fit1))
Call: survfit(formula = fit1)
```

S code for testing proportional hazards:

```
Fit2 <- coxph(Surv(enter, exit, event) ~ sex,
data=survivalanalysis1, x=T)
> temp <- cox.zph(fit2)
> print(temp)
```

S code for Cox proportional hazards model with sex as a covariate:

```
>fit3<-coxph(Surv(enter,exit,event)~sex+age,data =
survivalanalysis1)
> summary(fit3)
Call:
coxph(formula = Surv(enter, exit, event) ~ sex+age, data =
survivalanalysis1)
```

S code for Cox proportional hazards model, stratified by sex:

```
> fit4<-coxph(Surv(enter,exit,event)~strata(sex),data =
survivalanalysis1)
> summary(survfit(fit4))
Call: survfit(formula = fit4)
```