ILLINOIS NATURAL HISTORY SURVEY

Lake-wide mark and recapture investigation of Lake Michigan yellow perch: evaluation of interstate movements, spawning site fidelity, spawning population abundance, and sources of mortality.

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2005 Project Completion Report[1]

Lake-wide mark and recapture investigation of Lake Michigan yellow perch: evaluation of interstate movements, spawning site fidelity, spawning population abundance, and sources of mortality

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INTRODUCTION:

In Lake Michigan, yellow perch *Perca flavescens* have suffered from poor recruitment since 1989 (Francis et al. 1996). However, the mechanisms driving poor recruitment still have not been identified. Much recent research has focused on survival during the first year of life (e.g., Bremigan et al. 2003; Dettmers et al. 2003; Fitzgerald et al. 2004; Graeb et al. 2004); less attention has been given to the adult population. Until a mechanism(s) affecting recruitment can be identified, it is essential to properly manage the existing adult populations. Delineation of yellow perch stocks is an integral element for the management of the yellow perch fishery. Specifically, this information is important to determine allocation of catch between competing fisheries, identify spawning areas for potential protection, and for development of optimal harvest goals and monitoring strategies (Kutkuhn 1981; Moring 1999).

Recent research has indicated that at least three genetically distinct groups of yellow perch exist in Lake Michigan: Green Bay, northern basin, and southern basin (Miller 2003). However, other work suggests that individual stocks may be present within these three genetic groups. For example, previous tagging studies in Lake Michigan indicated that yellow perch tended to return to the same spawning site each spring (Marsden et al. 1993; Hirethota et al. 1997). This could be a mechanism by which groups of spawning yellow perch become spatially isolated, a key for genetic separation (Adkinson 1995). Another study in southern Lake Michigan examined the spatial and temporal patterns in size-at-age for three year classes of yellow perch, as well as their condition (Horns 2001). Differences occurred among sympatric groups in mean size-at-age and condition of adult yellow perch, implying spatial segregation. Thus, yellow perch in southern Lake Michigan may not be a single population but may consist of two or more distinguishable subpopulations or stocks. Further genetic analyses with current technology and techniques would most likely not yield the desired resolution, given the low genetic variability of yellow perch (Leary and Booke 1982; Billington 1996; Miller 2003). Thus, other methods to identify and discriminate between stocks must be used.

We used recaptures from a lake-wide yellow perch tagging project to provide information about stock delineation in Lake Michigan. We quantitatively evaluated movement patterns using methods similar to those used to determine seasonal dispersal distance, directional preference, and spawning site fidelity for lake trout in Lake Michigan (Schmalz et al. 2002). Our results explored the range and pattern of movement, degree of mixing among populations, movements across jurisdictional boundaries (Ihssen et al. 1981) and degree of spawning site fidelity (MacLean et al. 1981). In addition, we evaluated the potential of using appropriate mark-recapture models to estimate several population dynamic parameters using maximum likelihood estimation techniques; such as, local population abundance, movement probabilities between locations, survival, and emigration through time at a variety of spatial scales. All of this information is critical to successful management of yellow perch in Lake Michigan and will provide useful information for future large scale tagging efforts in the Great Lakes.

OBJECTIVES:

Objective 1. Complete development of the lake-wide yellow perch mark-recapture database housed at the Illinois Natural History Survey Lake Michigan Biological Station.

METHODS:

To conduct rigorous analyses of the lake-wide yellow perch mark-recapture data, we first created a standardized database formatted to correspond with other Great Lakes tagging databases. We consulted with other Great Lakes project personnel that have experience with tagging databases to ensure maximum efficiency of the database. Also, McLaughlin et al. (2001) provided excellent recommendations and cautions regarding development of databases that integrate data from a variety of sources. We used this document throughout all phases of the work described. Following completion of the database, all tagging and recapture data from 1996-2001 was entered and proofed.

RESULTS / DISCUSSION:

A lake-wide yellow perch mark-recapture database was created using Microsoft Access, which corresponds to other Great Lakes tagging databases. The database was used for data entry, proofing, and analyses. Utilizing features available in Access such as a master directory board and data entry forms with explicit explanations built-in, the database is extremely user-friendly. In the past, response letters were sent to anglers that supplied recapture information for the tagged fish harvested. These letters included release and recapture information of their tagged yellow perch as well as an overview of the tagging project. Although this was intended to keep the public interested in the project and keep return rates high, it took a considerable amount of time to gather the necessary information. The current database now generates these response letters via mail merge with Microsoft Word in the click of a button, significantly decreasing response time. Additionally, the database contains a table devoted entirely to contact information for each of the tagging agencies, facilitating inter-agency communication. We have also incorporated simple calculations that are commonly used with mark-recapture data, such as time-at-liberty, growth, as well as distance and direction traveled. In our experience, the database has proven more efficient than using other software packages such as Microsoft Excel and we highly recommend its use in future Great Lakes tagging efforts.

Plans are currently being finalized to deliver the database for posting and access on the GLFC web page.

Objective 2. Describe the interstate movements of Lake Michigan yellow perch, and develop a quantitative movement matrix for this species that can be used in population modeling efforts.

METHODS:

Yellow perch capture, marking, and recapture. Adult yellow perch were captured, marked, and released concurrently with annual assessments of spawning adults at nine locations in the southern basin of Lake Michigan and one location in the southern portion of Green Bay (Table 1). Tagging was conducted during May and June from 1996 to 2000 by the Illinois Natural History Survey (INHS) in Illinois and from 1997 to 1999 by the Indiana Department of Natural Resources (IN DNR), the Michigan Department of Natural Resources (MDNR), and the Wisconsin Department of Natural Resources (WDNR), in Indiana, Michigan and Wisconsin waters, respectively. WDNR was also responsible for tagging yellow perch in Green Bay during April from 1997 to 1999.

Yellow perch were collected with a variety of gear types, depending on the agency. INHS used 1.2 x 1.8-m double-ended, double-throated fyke nets with 38-mm stretched mesh and a 30.5-m lead in depths ranging from 5 to 17 m. Throughout the study, INHS tagged fish at five sites in Illinois, including Waukegan Wiremill (IL-1), Lake Bluff (IL-2), North Lake Forest (IL-3), South Lake Forest (IL-4), and Fort Sheridan (IL-5; Table 1; Figure 1). IN DNR captured yellow perch at Mt. Baldy (IN-1) at depths ranging from 6.5 to 9 m using double-ended, double-throated fyke nets with dimensions similar to those used by INHS (Table 1; Figure 1). MDNR used trap nets consisting of pots with 38-mm stretched mesh and 60 to 90-m leads having 51 to 76-mm stretched mesh set between 5 and 10-m deep at North St. Joseph (MI-1) and South St. Joseph (MI-2; Table 1; Figure 1). WDNR set several gear types at Green Can Reef (WI-1; Table 1; Figure 1) at depths ranging from 6 to 18 m, including 1.2 x 1.8-m double-ended, double-throated fyke nets with 19-mm stretched mesh and a 15.2-m lead (1997); 1.2 x 1.8-m double-ended fyke nets with 57-mm stretched mesh and a 152.4-m lead (1998); 1.2 x 30.5-m monofilament gill net panels that consisted of 57-mm, 64-mm, or 70-mm stretched mesh (1999). Little Tail Point (GB-1; Table 1; Figure 1) was sampled at a depth of 4 m with 1.2 x 1.8-m double-ended, double-throated fyke nets with 19-mm stretched mesh and a 30.5-m lead. Every attempt was made to set and recover all gear types within 24 hours. Occasionally, unfavorable weather prevented their recovery for up to 8 d (3 of 363 sets).

All yellow perch were measured to the nearest mm (total length); sex was determined by expression of milt or eggs. Fish over 150 mm long (120 mm in Green Bay) and in good condition were tagged and immediately released. Individually-numbered Floy® FD-94 anchor tags were inserted on the left side, above the lateral line and below the soft rays of the dorsal fin. The INHS address and phone number was also imprinted on all tags to facilitate returns.

Tagged fish were recovered by commercial fishermen, recreational anglers, or recaptured by state fisheries agencies from 1996 to 2001. The tag number, date, recapture location, sex, and total length (nearest mm) were recorded for each yellow perch recaptured (fish recaptured alive and subsequently released). For yellow perch recovered (i.e., harvested) by commercial and recreational sources, information was requested for the tag number, date, and location. No rewards were offered for providing information on recovered yellow perch. However, details about the project and information specific to the recovered yellow perch (e.g., length, sex, date and location tagged) were sent to anglers that supplied contact information. To inform fishermen, details of the project were included in press releases, harvest regulation booklets, on DNR web-sites, and were posted at harbors and bait shops. Occasionally, inconsistencies were found with reported tag recoveries. Unless inconsistencies could be verified with the original tag, these recoveries were omitted from analyses.

Mark-recapture models (*Green Bay*). To estimate survival (Objective 5) and local population abundance (Objective 4) for Green Bay, we modeled agency recaptures under Pollock's robust design model structure (Kendall et al. 1995; Kendall et al. 1997). Green Bay was analyzed separately from Lake Michigan because previous tagging studies in Green Bay documented that the majority of yellow perch were recovered within 32 km of their release site and no perch ventured into the main lake (Mraz 1951). Only a single yellow perch tagged in waters of Lake Michigan was recaptured in Green Bay throughout the current tagging study. The Green Bay model only included GB-1 (Table 1; Figure 1), where tagging was conducted for three years (i.e., primary sessions). A series of secondary trapping sessions were completed within each year (6, 8, and 10, respectively). All secondary sessions were completed within three weeks and were conducted during the closed yellow perch fishing season. The most significant advantage to using Pollock's robust design with a single location is that the model allows for the estimation of emigration and thus the relaxation of assumptions on animal movement (Kendall et al. 1995; Kendall et al. 1997). Parameters and their definitions for Pollock's robust model estimated by the program MARK are as follows (Kendall et al. 1995; Kendall et al. 1997; White and Burnham 1999):

Parameters estimated from secondary sessions (within primary sessions):

c_{ij} = the probability a fish at risk of first capture in the j^{th} secondary sampling session during the i^{th} primary session is captured.

p_{ij} = the probability of recapture in the j^{th} secondary sampling session during the i^{th} primary session is recaptured, conditional on the tagged fish being alive and available for recapture.

N_i = abundance, (n_i/p^*_i), where n_i is the total number of individuals detected during the i^{th} primary session, and p^*_i is the pooled detection probability for secondary sessions during the i^{th} primary session.

Parameters estimated for intervals between primary sessions:

S_i = the probability a fish alive during the i^{th} primary session is alive during $i + 1$.

γ''_i = the probability of emigration from the trapping area during the i^{th} primary session given that the fish was available for capture during $i - 1$.

γ'_i = the probability of a fish permanently emigrating from the trapping area during the $i + 1$ primary session given that the animal has left the trapping area during the i^{th} primary session.

Parameter estimates were obtained by the maximum likelihood function derived from the probability of each unique encounter history using the logit link (White and Burnham 1999). Please refer to Kendall et al. (1995) and Kendall et al. (1997) for the maximum likelihood estimators and the corresponding variance components of these parameters.

Model testing (*Green Bay*). In the analysis of mark-recapture data, proper parameterization of the encounter probabilities (e.g., probability of first capture, recapture and recovery) are the foundation for precise and unbiased estimates of population size and survival (Seber 1982; White et al. 1982; Lebreton et al. 1992). Thus, following the general strategy for analyzing mark-recapture data (Lebreton et al. 1992), the most parsimonious model was determined based on probability of first capture and probability of recapture, followed by emigration, survival, and finally abundance. We first ran the general time-dependent model in which parameter estimates were obtained for each time period (Model 1; Table 2). Model 1 was also designed to separately estimate probability of first capture and recapture, which allows for a behavioral response induced by the trapping experience (e.g., trap happy or trap shy; White et al. 1982; Kendall et al. 1995; Kendall et al. 1997). To test if a behavioral response did occur, these parameters were set equal for each trapping session (Model 2; Table 2). We also tested whether the probability of capture and recapture were equal across time for secondary sessions (Model 3; Table 2) and among primary sessions (Model 4; Table 2). Following recommendations from Kendall et al. (1995) and Kendall et al. (1997), we set the emigration parameters equal to zero (Model 5; Table 2) to determine if emigration was occurring from GB-1. To determine if emigration was a random process, we set γ''_i and γ'_i equal to each other (Model 6; Table 2). We also set the probability of emigrating from the study area equal across time to determine if emigration rates changed between 1997 and 1998 (Model 7; Table 2). Survival was set equal across time to test its temporal stability over the relatively short time period (Model 8; Table 2). Lastly, abundance was set equal among years to determine if it has remained similar throughout the study (Model 9; Table 2).

Mark-recapture models (*Lake Michigan*). We estimated survival (Objective 5) and movement rates among management units in the southern basin of Lake Michigan. Including abundance as a parameter along with the other required parameters required to estimate abundance would greatly increase the number of parameters estimated and reduce the overall precision. However, we obtained independent estimates of local spawning abundance using a Schnabel census (Seber 1982; see Objective 4). Thus, for waters of Lake Michigan, a multistate live-dead model was used (Brownie et al. 1993; Burnham 1993). This model utilizes live recaptures (i.e., agency) and dead recoveries (i.e., sport harvested) to estimate survival and movement rates among predetermined localities or strata. Given the extremely large number of parameters that would be estimated, combined with a relatively low probability of capture if all sites were utilized in the model, all tagging sites were pooled within a state (i.e., Wisconsin, Illinois, Indiana, and Michigan). The model parameters for the multistate live-dead model estimated in the program MARK are as follows (Brownie et al. 1993; Burnham 1993; White and Burnham 1999):

p_i^s = the probability that a tagged fish alive in the destination stratum s during the i^{th} year is recaptured.

r_i^s = the probability that a tagged fish alive in the destination stratum s during the i^{th} year is recovered.

S_i^{rs} = the probability that a fish alive in stratum of origin r during the i^{th} primary session is alive in the destination stratum s in the primary session i+1.

ψ_i^{rs} = the probability that a fish in the stratum of origin r during the i^{th} primary session is in the destination stratum s in primary session $i+1$, given that the animal is alive in the primary session $i+1$.

Parameter estimates are obtained by the maximum likelihood function derived from the probability of each unique encounter history (White and Burnham 1999). Separate multinomial logit links were used for transition rates from each location to ensure that the combined probabilities summed to one. For all other parameters, a single logit link was used (White and Burnham 1999). Please refer to Brownie et al. (1993) and Burnham (1993) for the maximum likelihood estimators and the corresponding variance components of these parameters.

Model testing (*Lake Michigan*). Before building a set of candidate models for Lake Michigan, we fixed several parameters equal to zero because they could not have occurred, which effectively removes them from the estimation procedure (White and Burnham 1999). Specifically, the probability of recapture for 2000 was fixed to zero for Wisconsin, Michigan, and Indiana because tagging operations were not conducted in these waters during this year (Table 1). We also set all transition probabilities from these states to all other states zero for the 1996 to 1997 interval because no fish were tagged in these areas during 1996.

Similar to the methodology used for Green Bay, the most parsimonious model was determined by sequentially manipulating recovery, transition, and survival probabilities. However, all models with a time-dependent component failed to converge. After the best model was determined for a particular set of parameters, it then became the baseline for the next set of manipulations. Hence, we began with a model in which all parameters were set equal across time (Model 10; Table 2). I set recovery probabilities equal between Illinois and Indiana because harvest regulations were similar between these states (Model 11; Table 2). Equal recovery rates among all states were not tested due to the varying harvest regulations among states. Further, recapture probabilities were not tested for similarity among states due to the varying gear types used among agencies and varying sampling duration. No further manipulations were made to transition rates because they were already reduced to the fewest possible parameters. To evaluate if survival probabilities were similar among states, we set survival probabilities equal between adjacent states (i.e., Illinois and Indiana [Model 12; Table 2], Illinois and Wisconsin [Model 13; Table 2], Indiana and Michigan [Model 14; Table 2]). In addition, to determine if the southern basin was panmictic with respect to total survival we ran a model in which all survivals were set equal among all states (Model 15; Table 2).

Mark-recapture models (*Illinois*). We used data from Illinois waters to evaluate survival and movement at a smaller scale using the same model structure as for Lake Michigan (i.e., multistate live-dead model) because the largest and most complete data set was available for Illinois waters(e.g., most tagging sites, years, yellow perch tagged, and number of recaptures). For the Illinois model, we employed six strata, including the IL-1, IL-2, IL-3, IL-4, and IL-5 tagging sites and another stratum that included recaptures and recoveries from anywhere in the lake (hereafter, LM). Included this additional stratum not only preserves as much information as possible, but also allows us to determine the probability of returning to Illinois sites after emigrating into the LM stratum.

For all models in Illinois, we set survival equal to zero during the 1996 to 1997 interval for IL-1, IL-4, and LM because fish were not tagged in these areas during 1996. Probabilities of recapture also were fixed to zero for IL-2 (1997-2000), and IL-4 (2000). The probability of transition from LM to all other strata was set to zero for 1996 because no tagged fish were available to disperse from this stratum. No fish tagged in IL were recaptured in LM during 2000 and the recapture rate for LM was therefore fixed to zero.

Model Testing (*Illinois*). The general strategy for determining the most parsimonious model was similar to the Green Bay and Lake Michigan model testing. Due to continued convergence failure with time-dependent models, we began with the model in which all parameters were held equal among years and sites (Model 16; Table 2). We then manipulated the recovery probabilities to be time-dependent (Model 17; Table 2), different among sites (Model 18; Table 2), and both different among sites and time-dependent (Model 19; Table 2). To evaluate if the change in the harvest regulations from 1996 to 1997 (from 25 fish per day in 1996 to 15 fish per day with an 8 to 10-inch keeper slot limit in 1997) affected probability of recovery, we divided recovery probabilities into pre- (1996) and post- (1997 to 2000) regulation change for Illinois sites (Model 20; Table 2). The recovery probability for the LM area was estimated separately, and set equal over time, because this stratum included harvest regulations from several states. Using the best model based on recovery probabilities as a baseline, we then built upon the model by manipulating recapture probabilities. We did not expect that recapture probabilities varied substantially among sites (within Illinois) because the same gear type was used; hence this effect was not tested. We set recapture probabilities to be time dependent to determine if they varied among years (Model 21; Table 2). After determining the best model based on recapture rates, it was then used as a baseline for manipulating transition rates. We did not expect that transition probabilities would drastically change over time, but did recognize they might be quite different among sites. Therefore, we set transition probabilities different among sites while keeping them temporally constrained (Model 22; Table 2). To evaluate if the probability of transition among sites was a function of proximity, we used distance to destination sites as a covariate (Model 23; Table 2). However, distance to the LM stratum could be an infinite number of values, and therefore we did not include it as part of the covariate. Rather, we set all transition probabilities to and from the LM stratum equal, which were estimated separately from all other sites. To determine if survival was affected by the change in harvest regulations after 1996, we split it between pre- and post- time periods, similar to treatment of the recovery probability (Model 24; Table 2). We also evaluated if survival varied across time but was equal across sites (Model 25; Table 2), if survival varied across sites, but was not time variant (Model 26; Table 2), and if survival varied across both time and space (Model 27; Table 2).

Model selection (*Green Bay, Lake Michigan, Illinois*). After a set of candidate models were developed, models were ranked by AIC_c (i.e., second-order Akaike's information criterion corrected for small sample size) which was calculated by:

$$\mathrm{AIC_c} = -2\log(L(\hat{\theta} \mid x, g)) + 2K\left(\frac{n}{n-K-1}\right),$$

where $L(\hat{\theta} \mid x, g)$ is the likelihood function of the model parameters $\hat{\theta}$, given the data x and the model g; n is the sample size, and K is the number of estimable parameters in model g (Hurvich and Tsai 1989). We used ΔAIC_c (i.e., the difference between a particular model's AIC_c value and the AIC_c value from the model with the lowest AIC_c value) to select the best model to draw inferences. (Burnham and Anderson 2002) recommended that all models having ΔAIC_c of 1-2 be considered as having substantial support, models having 3-7 considerably less support, and models greater than 10 essentially no support. Further, AIC_c weight (w_i) was determined for the set of candidate models, and was calculated as:

$$w_i = \frac{\exp\left(-\frac{1}{2}(\Delta AIC_c)_i\right)}{\sum_{r=1}^{R}\exp\left(\frac{1}{2}(\Delta AIC_c)_r\right)},$$

where R is the total number of candidate models (Burnham and Anderson 2002). The proportion of AIC_c weights between two models is the evidence ratio (i.e., how many times a particular model is more likely to be the best model; Burnham and Anderson 2002).

Dispersal distance, directional movement, growth rate. In addition to analyzing movement with the mark-recapture models, we determined dispersal distance and directional preference for each of the tagging locations, using methods similar to those used for lake trout in Lake Michigan (Schmalz et al. 2002). To describe the range and pattern of adult yellow perch movements, the spatial distribution of sport recoveries was analyzed from each release site. Sport recoveries were used because they exhibited greater spatial coverage and temporal variation than agency recaptures. The ban on commercial fishing in Lake Michigan proper during most of the tagging study limited the utility of commercial recoveries. However, all sources of recapture and recovery were used later in spawning site fidelity analyses (see Objective 3). To account for spatially disproportionate angling effort that can bias movement analyses (Hilborn 1990), estimates of directed angler effort (h) for yellow perch were incorporated. Annual creel surveys are conducted for each state and monthly estimates are typically pooled by Lake Michigan management units for each fishery type (i.e., boat, charter, shore and stream anglers; Lockwood 1999; Lockwood et al. 1999; Palla 2003; Peterson and Eggold 2003; Brofka and Dettmers 2004). To account for spatial differences in angler effort at the smallest scale possible, we obtained estimates by port for Illinois (Figure 1; W. Brofka, INHS, unpublished data), Indiana (Figure 1; J. Palla, IN DNR, unpublished data), and Michigan (Figure 1; S. Thayer, MDNR, unpublished data), and by county for Wisconsin (Figure 1; B. Eggold, WDNR, unpublished data) for each year of the tagging study (1996 – 2001). However, Michigan did not separate effort directed at a specific species until 1997. To obtain angling effort directed at yellow perch for 1996 in Michigan, we multiplied total effort directed at all species in 1996 by the ratio (0.14) between effort directed at yellow perch and total effort directed at all species for 1997 to 2001. Creel surveys were conducted in Wisconsin from March to October (March and April, and September and October were combined into two single estimates), but no county was consistently surveyed during March and April throughout the tagging study. All ports in Illinois were surveyed from April to September each year. Ports in Indiana and Michigan were sampled from April to October, with the exception of St. Joseph and New Buffalo (Figure 1), which were also sampled in March. In 1998, Hammond (Figure 1) was not surveyed during April due to personnel shortages. The method of recovery (i.e., fishery type) was not specified for the majority of recoveries. Therefore, all fishery types were pooled into an estimate of total directed angler effort for yellow perch. For analyses concerning fish released in Illinois, angler effort was summed across 1996 to 2001 into three recovery periods; summer (June – August), non-summer (March – May and September – October), and total (March – October). For all other areas, angler effort was summed from 1997 to 2001 into the same recovery periods because tagging commenced one year after tagging began in Illinois.

Each recovered yellow perch was assigned the amount of estimated angler effort from their respective creel unit. Creel units for each port were derived by drawing straight lines from the midpoint between ports to their respective state line. Existing county lines were used to delineate creel units in Wisconsin waters of Lake Michigan and Green Bay. Distance moved was calculated as the straight-line distance from the tagging location to the closest possible location of reported recovery (e.g., street names of cities and towns, harbors, piers, beaches, power plants, water filtration plants, etc.). To determine the direction moved, we assigned directional movement for each fish as follows: north (315° - 45°), south

(135° - 225°), east (45° - 135°), or west (225° - 315°). Fish recovered at the original tagging location were not assigned a direction and were thus omitted from the analyses. However, this was an extremely rare occurrence for sport recoveries.

Dispersal distance, which we defined as the distance within which 90% of the recoveries per effort (RPE) occured (Schmalz et al. 2002), was used as an index of home range for a group of individuals. Number of recoveries at a particular location were summed for each reported recovery location. The RPE for each recovery location was calculated as the number of recoveries per 10,000 angler hr directed at yellow perch. We assumed that angler effort was uniformly distributed within each port and county, which allowed me to assign several recovery locations within each area with identical estimates of angler effort. The cumulative proportion of RPE (y) was fit to an exponential sigmoid function of distance:

$$y = \frac{\alpha}{(1+\beta e^{Kx})},$$

where α is the maximum cumulative proportion of RPE that can be obtained (theoretically 1.0 or 100%), β is a parameter that scales the function toward zero, and K is the rate at which RPE increases with distance (x). The modified Gauss-Newton iterative method that relies on exact derivatives was used to determine the parameters that produce the lowest residual sum of squares for each tagging location (SAS Institute 1999). Using the derived parameters, the distance (x) at which the cumulative proportion of RPE (y) was equal to 0.90 (90%) was estimated for each tagging location and recovery period. We applied this analysis to recoveries during summer and to total recoveries.

ArcView, version 3.2 (ESRI 1998) was used to develop a linear scale for shoreline distance in the southern basin of Lake Michigan to assess dispersal across management boundaries and overlap among fish released from the various tagging locations (shoreline data provided by E. Marshall, University of Michigan). Shoreline distance was used because all recoveries occurred nearshore throughout the study except in Green Bay. The shoreline distance scale began at the northern border of Ozaukee County, Wisconsin (43°32.528′ N, 87°47.607′ W; Figure 1) and continued counter-clockwise around the southern basin, ending at the northern border of the Muskegon unit, Michigan (43°30.233′ N, 86°26.714′ W; Figure 1). Dispersal was expressed in terms of shoreline distance by creating 90% dispersal buffers around each tagging location for each recovery period and determining the point at which the buffer crossed the shoreline. All recoveries, tagging sites, and management boundaries were similarly translated into shoreline distance.

To describe directional movement from release locations between two recovery periods, five separate weighted analyses of variance (ANOVA) were used. Each model represented a major area where tagging occurred, i.e. Green Bay, Wisconsin, Illinois, Indiana, and Michigan. The form of the generalized linear model used was:

$$y_{ijkl} = \mu + T_i + R_j + TR_{ij} + D(R)_{jk} + TD(R)_{ijk} + L_{ijkl} + \varepsilon_{ijkl},$$

where

y_{ijkl} is the l^{th} observed distance traveled (km) by a recovered yellow perch tagged at site i, recovered in period j, at k direction from the tagging location weighted by the reciprocal of angler effort in hours,

μ is the grand mean of all observations,

T_i is the effect of tagging location i,

R_j is the effect of recovery period j (summer and non-summer recovery periods),

$D(R)_{jk}$ is the effect of direction k nested within recovery period j (north, south, east, or west nested within the summer or non-summer recovery period),

TR_{ij} is the interaction between tagging location i and recovery period j,

$TD(R)_{ijk}$ is the interaction effect between tagging location i and direction k nested within recovery period j,

L_{ijkl} is the random effect of a covariable, time-at-liberty, and

ε_{ijk} is the residual error from each observation.

We did not include the effect of T_i and its interactions for models that had only one tagging location (i.e., Green Bay, Wisconsin and Indiana). T_i was included in the Illinois and Michigan models to evaluate whether similar trends occurred at closely spaced tagging locations. Directions in which few recoveries (three or less) occurred in the specific recovery period were omitted from the analysis.

To minimize the type-I experiment-wise error, post-hoc one-tailed t-tests that were performed at the $\alpha = 0.025$ level tested whether 1) there was directional preference from each tagging location by testing whether the largest directional mean distance traveled was statistically larger than all other directions, 2) the magnitude of directional movement was greater in summer as compared to non-summer periods when the preferred movement was similar in direction between periods, and 3) mean movement was greater during the non-summer recovery period. One-tailed t-tests were not performed unless the F-tests indicated significance at $\alpha = 0.05$.

We used the increment in growth from time of release to recapture as an index of the growth rate (mm·year^{-1}) for adult yellow perch in the southern basin of Lake Michigan and the southern portion of Green Bay. Only agency recaptures were used in this analysis because they provided the most reliable source for accurate measurement of yellow perch. Further, only fish recaptured at least one year subsequent to tagging were used. An ANOVA was used to discern if the mean growth rate differed among fish released from each of the ten tagging sites nested within each area (i.e., Green Bay, Wisconsin, Illinois, Indiana, and Michigan) and among areas. Initial size was included as a covariate to correct for size-dependent growth. Growth was analyzed with Proc Mixed in SAS version 8.0 (SAS Institute 1999). The general linear model used was:

$$y_{ijk} = \mu + S_i + T(S)_{j(i)} + I_k + \varepsilon_{ijk},$$

where

y_{ijk} is the growth rate (mm·year^{-1}) of the k^{th} recaptured yellow perch tagged at site j within state i,

μ is the grand mean of all observations,

S_i is state i in which tagging occurred,

$T(S)_{j(i)}$ is tagging site j nested within state i,

I_k is the initial size for the k^{th} fish, and

ε_{ijk} is the residual error from each observation.

Residuals were assessed for normality; extreme outliers (N=11) were deleted to meet the assumptions of ANOVA. The Pdiff option was used for all possible comparisons between tagging areas, a total of 10 comparisons. Therefore, we used a Bonferroni adjustment to decrease the type-I experiment-wise error (Littell et al. 2002).

RESULTS:

Yellow perch capture, marking, and recapture. A total of 63,948 adult yellow perch were tagged at the ten tagging locations between 1996 and 2000 (Table 1). A total of 5,025 tagged yellow perch were recaptured between 1996 and 2001, which represented a 7.9% combined recapture/ recovery rate (Table 1). Agency recaptures, sport recoveries, and commercial recoveries accounted for 75.6%, 18.8% and 5.6% of all recapture/ recoveries, respectively. Mean time-at-liberty was 224 d and ranged from 0 to 2004 d (5.5 years), with 92% of all recaptures occurring within 2 years of release.

Mark-recapture models (*Green Bay*). We have little confidence in the estimates of survival and local spawning abundance produced as a result of poor fitting for the Green Bay model. For example, the model selection process for Green Bay was ambiguous because AIC_c was equivalent for Models 1, 6, 7, 8 and 9 (Table 3), indicating insufficient data for the models tested (Burnham and Anderson 2002). Also, the number of estimable parameters observed was less than the number of theoretically estimable parameters for all models (i.e., the total number of parameter estimates attempted; Table 3), another indication of insufficient data to estimate specific parameters (Burnham and Anderson 2002). Further, Model 3 failed to reach numerical convergence. Moreover, estimates from Model 1 (Table 4; note that estimates were similar among Models 1, 6, 8, and 9) indicate probabilities of survival from 1997 to 1998 and from 1998 to 1999 were either unrealistically high or low. The very high SE (extending beyond the theoretical threshold from 1998 to 1999 [i.e., greater than one]) and confidence intervals at the extreme boundaries of the estimates also indicate problems with the estimates of survival rates (Burnham and Anderson 2002). Similarly unreliable results occurred for all emigration parameters (i.e., $\gamma''_{1997-1998}$, $\gamma''_{1998-1999}$, γ'_{1998}). In addition, the estimated population abundance for GB-1 closely resembled the number of adult yellow perch tagged during each year (within 20 fish). The SE for abundance estimates were zero, resulting in confidence intervals equal to that of the estimate. Probabilities of first capture and recapture varied greatly within each year, but also had suspect SE and confidence intervals. The poor foundation of first capture and recapture probabilities are likely responsible for the unfavorable estimates of all other parameters (Seber 1982; White et al. 1982; Lebreton et al. 1992).

Mark-recapture models (*Lake Michigan*). The model in which all parameters were different among states and similar through time (Model 10) was the most parsimonious model for Lake Michigan (Table 3). Results from this model indicate that survival differed between Illinois and Indiana (Model 12), Illinois and Wisconsin (Model 13) and among all states (Model 15) from 1996 to 2001 (Table 3). However, the attempts to set survival equal between Indiana and Michigan (Model 14) as well as setting recovery probabilities equal between Illinois and Indiana (Model 11) were unsuccessful, also due to convergence failure. In all cases, the number of estimable parameters was less than the number that could theoretically be estimated (Table 3), once again implicating too few data (Burnham and Anderson 2002). According to parameter estimates obtained from Model 10, survival varied greatly among states (Table 5). Yet, we have little confidence in the estimates of the best model because recapture and recovery rates were very low for each state and recovery was inestimable for Illinois. In addition, transition probabilities were inestimable to a large extent, likely a result of the poor recapture and recovery rates.

Mark-recapture models (*Illinois*). The model fitting procedure for Illinois was more successful than for Lake Michigan. Yet, tests for survival were mostly unsuccessful due to convergence failure (Models 24, 26 and 27; Table 3). The only model to achieve convergence with respect to survival was the time dependent survival rate (Model 25), which had essentially no support compared to Model 22 (Table 3), indicating survival was equal across time. Based on the parameters derived from our best model for Illinois (Model 22), the annual survival rate for fish released from Illinois was extremely low (Table 6). However, both recapture and recovery rates were often 1 or 0 and had suspect SE, drawing into question the reliability of the estimates derived from this model and potential inferences drawn from between model testing (Burnham and Anderson 2002).

Dispersal distance (*Summer*). Summer angling effort was generally greater than non-summer effort for each creel location during each year of recovery (Table 7). No area (i.e., state) consistently received the most angler effort for all years, but the majority of angler effort directed at a particular creel location within each area was fairly consistent (Table 7). For Green Bay, Door County received most of the angling effort directed toward yellow perch in all years (Table 7). Kenosha and Milwaukee counties received the most effort in Wisconsin. In Illinois, Montrose Harbor consistently received the most fishing effort (Table 7). Fishing effort was variable among creel locations in Indiana and was not consistent through time (Table 7). Within Michigan, South Haven, St. Joseph, and Grand Haven received the majority of angler effort directed at yellow perch in most years.

The proportion of total variability explained (R^2) for the dispersal distance models ranged from 0.940 to 0.997, indicating that describing the cumulative proportion of RPE as an exponential sigmoid function of distance fit the data very well for both summer and total recovery periods (Table 8). During summer, 90% dispersal distance from GB-1 was 28.7 km, which remained within the Wisconsin waters of Green Bay (Table 8). In the southern basin, summer dispersal distance averaged 60.4 km but was quite variable, ranging from 12.8 to 101.4 km (Table 8). Ninety percent of the recoveries from WI-1 stayed within the Wisconsin waters of Lake Michigan (Figure 2). However, dispersal distances from four out of five of the Illinois tagging sites crossed the Illinois border into Wisconsin waters (Figure 2). The 90% dispersal distance from IL-3 also crossed into Indiana waters (Figure 2). Fish released from IN-1 were recovered within 44.3 km 90% of the time (Table 8), resulting in overlap into Michigan waters (Figure 2). Dispersal from MI-2 crossed into Indiana waters, whereas dispersal from MI-1 extended 101.4 km, well into Illinois waters (Table 8; Figure 2).

Considerable amounts of mixing within and among states occurred during summer (Figure 2). However, mixing among states occurred mostly between adjacent states with the exception of Michigan and Illinois (Figure 2). Dispersers within Illinois and Michigan had considerable amounts of overlap among sites, many being completely enclosed within the dispersal area of another site, indicating little potential for isolation by distance within states (Figure 2).

Dispersal distance (*Total*). Dispersal distances for the total recovery period exceeded those of summer for all sites except for fish from IL-3 and MI-1 (Table 8). Dispersal distance could not be estimated for IL-1 and MI-2 because the estimated maximum cumulative proportion of RPE (α) was below 0.90. Although the dispersal distance increased when considering the entire time period, movement from GB-1 and WI-1 remained within the local management jurisdiction. Dispersal from four sites in Illinois crossed into Wisconsin waters (Figure 3). Dispersal from two sites in Illinois crossed into Indiana (Figure 3). Ninety percent of the fish from IN-1 were recovered within 58.8 km (Table 8), resulting in movement into both Illinois and Michigan waters (Figure 3). Dispersal from MI-1 extended 80.5 km (Table 8), crossing into Indiana waters (Figure 3).

Mixing among dispersers during the total time period was limited to mixing among fish released from adjacent states (Figure 3). However, as a result of increased dispersal distances during the total time period, the mixing that occurred between Illinois and Wisconsin and Illinois and Indiana increased slightly compared to summer (Figure 3). Dispersal among all sites within Illinois was completely overlapping (Figure 3).

Directional movement (*Green Bay*). Time-at-liberty did not affect the variability of distance traveled from GB-1 ($F = 0.64$; df = 1, 40; $P = 0.43$) and was omitted from the Green Bay model.

Recovery period also did not affect the mean distance traveled ($F = 0.87$; df = 1, 41; $P = 0.36$), likely due to the geographical constraints on movement. Distance traveled depended on direction nested within recovery period ($F = 11.01$; df = 3, 41; $P < 0.001$). During non-summer months, yellow perch traveled farther east than south ($t = 5.56$; df = 41; $P < 0.001$; Table 9), in part because southward movement was geographically constrained. During summer, there was no directional movement preference ($t < 0.41$; df = 41; $P > 0.08$; Table 9).

Directional movement (*Wisconsin*). Variability of distance traveled was not affected by time-at-liberty for fish recovered from WI-1 and was therefore omitted from the Wisconsin model ($F = 1.36$; df = 1, 43; $P = 0.25$). The mean distance traveled was affected by recovery period (F=16.96; df = 1, 44; $P <$ 0.001), with distance moved during summer (25.9 ± 2.4 km) greater than during other months (6.0 ± 4.2 km; $t = 4.12$; df = 44; $P < 0.001$). Direction nested within recovery period also affected the mean distance traveled ($F = 5.33$; df = 1, 44; $P = 0.03$). Distance traveled southward was greater than northward during summer ($t = 2.31$; df = 44; $P = 0.01$; Table 9), but remained within the rocky habitat in Wisconsin waters. During the non-summer recovery period, only six recoveries were recorded, all of which moved northward (Table 9).

Directional movement (*Illinois*). Time-at-liberty affected the variability of distance traveled from Illinois tagging locations ($F = 28.40$; df = 1, 399; $P < 0.001$). Recovery period also affected distance traveled ($F = 39.63$; df = 1, 399; $P < 0.001$), with mean distance traveled during the non-summer period (38.0 ± 2.0 km) larger than during summer (24.7 ± 1.2 km; $t = 6.30$; df = 399; $P < 0.001$). Mean distance traveled was also affected by tagging location ($F = 5.06$; df = 4, 399; $P < 0.001$), direction nested within recovery period ($F = 68.78$; df = 2, 399; $P < 0.001$), the interaction between tagging location and recovery period ($F = 6.51$; df = 4, 399; $P < 0.001$), as well as the tagging location*direction interaction nested within recovery period ($F = 3.69$; df = 8, 399; $P < 0.001$).

During summer, mean distance traveled southward was greater than northward from four out of the five sites, which was consistent with availability of rocky substrate (IL-2, $t = 5.25$; df = 399; $P <$ 0.001; IL-3, $t = 3.72$; df = 399; $P < 0.001$; IL-4, $t = 2.91$; df = 399; $P < 0.01$; IL-5, $t = 2.95$; df = 399; $P <$ 0.01; Table 9). Distance traveled north or south from IL-1 during summer did not differ ($t = 0.16$; df = 399; $P = 0.44$; Table 9). The distance traveled southward was greater than northward for all Illinois sites during non-summer months ($t > 1.89$; df = 399; $P < 0.03$; Table 9). The magnitude of southward movement between recovery periods increased for IL-2 ($t = 5.67$; df = 399; $P < 0.001$), IL-3 (t = 2.25; df = 399; $P = 0.01$), and IL-5 ($t = 4.60$; df = 399; $P < 0.01$), but did not differ for IL-4 ($t = 0.92$; df = 399; P = 0.18).

Directional movement (*Indiana*). The variability of distance traveled from IN-1 was affected by time-at-liberty ($F = 7.69$; df = 1, 82; $P < 0.001$). The effect of recovery period and direction traveled within period affected distance traveled ($F = 161.68$; df = 1, 82; $P < 0.001$; $F = 34.72$; df = 2, 82; $P <$ 0.001, respectively). The mean distance traveled during the non-summer period (49.0 ± 1.8 km) was larger than during summer (17.2 ± 2.1 km; $t = 12.72$; df = 82; $P < 0.001$). Movement westward toward cobble substrate during summer was larger than the movement eastward or southward ($t > 6.97$; df = 82; $P < 0.001$; Table 9). Southward movement was limited by the shoreline. During non-summer months, yellow perch only moved westward, for greater distances than during summer ($t = 2.28$; df = 82; $P = 0.01$; Table 9).

Directional movement (*Michigan*). Time-at-liberty affected the variability of the distance traveled and remained in the Michigan model ($F = 20.62$; df = 1, 268; $P < 0.001$). Mean distance traveled was similar between tagging locations ($F = 3.82$; df = 1, 268; $P = 0.05$) but differed for tagging locations between recovery periods ($F = 3.94$; df = 1, 268; $P = 0.05$). Also, direction moved within recovery period did not differ between tagging locations ($F = 9.67$; df = 1, 268; $P < 0.01$), but recovery period did affect the distance traveled ($F = 53.29$; df = 1, 268; $P < 0.001$) with movement during non-summer months (52.7 ± 4.6 km) greater than that of summer months (19.3 ± 3.7 km; $t = 7.30$; df = 268; $P < 0.001$).

Only northward movement from MI-1 and MI-2 was observed during the non-summer recovery period (Table 9). Although random movement was expected during summer due to the predominance of sandy substrate on the eastern shoreline (Powers and Robertson 1968), the mean distance traveled northward was greater than southward from MI-1 ($t = 6.39$; df = 268; $P < 0.001$; Table 9). This movement northward was greater during non-summer months when compared to summer ($t = 5.21$; df = 268; $P < 0.001$). During summer, movement was random between north and south from MI-2 ($t = 0.08$; df = 268; $P = 0.47$; Table 9).

Growth Rate. Growth of recaptured yellow perch depended on the area in which it was tagged ($F = 0.45$; df = 1289; $P < 0.001$), but was similar between tagging sites within each state ($F = 0.94$; df = 1289; $P < 0.001$). Growth was also affected by the initial length ($F = 385.97$; df = 1289; $P < 0.001$), which negatively affected growth rate (slope = -0.18). This was not surprising given the general decrease in growth efficiency experienced by larger fish (Kitchell et al. 1977; Hewett and Kraft 1993). The lowest growth rate was observed for fish released from Michigan (3 mm·year^{-1}), which differed from all other areas ($t > 6.47$; df = 1289; $P < 0.001$; Figure 4). Green Bay had the highest growth rate (19 mm·year^{-1}), but was not different from any other area ($t < 2.19$; df = 1289; $P > 0.29$; Figure 4). Growth rates from Wisconsin (15 mm·year^{-1}), Illinois (16 mm·year^{-1}), and Indiana (13 mm·year^{-1}) were all similar ($t < 1.80$; df = 1289; $P > 0.72$; Figure 4).

DISCUSSION:

The Green Bay, Lake Michigan, and Illinois mark-recapture models fit the data poorly, which was evident from many inestimable parameters, standard errors near the boundaries of theoretical thresholds (extending beyond in some cases), and the inability to achieve numerical convergence for many models. Parameter inestimability is due either to parameters that are confounded with each other, such as survival and probability of recapture for the last time interval in the Cormak-Jolly-Seber model (Seber 1982), or to insufficient recaptures necessary to estimate the various parameters (Burnham and Anderson 2002). All parameters within Pollock's robust design are theoretically estimable (Kendall et al. 1995; Kendall et al. 1997), as are those for the multistate live-dead model (Brownie et al. 1993; Burnham 1993; White and Burnham 1999); therefore the inability to estimate parameters in this case can only be due to the scarcity of data.

The Lake Michigan model and Illinois model derived very different survival estimates for Illinois. Neither of the estimates derived from these models were close to those from recent catch-at-age model analyses, which estimated the annual survival rate of Illinois at around 40% from 1986-1997 (Wilberg et al. in press). The poor fit and ambiguity of these mark-recapture models draws into question the reliability of the estimates produced. This poor fit is likely due to not only the lack of recaptures/ recoveries, but also to bias caused by a number of assumption violations (e.g. tag loss, tag induced mortality, behavioral differences between marked and unmarked fish.), which we discuss in more detail in Objectives 4, 5, and 6.

Movement of adult yellow perch seemed to be related to substrate type and availability, yet we were unable to directly test this with our data. Spawning yellow perch select cobble substrate for spawning in Lake Michigan (Robillard and Marsden 2001), but might also select these areas due to higher abundance of preferred prey, such as crayfish, sculpins *Cottus* spp., and alewife *Alosa pseudoharengus* (Powers and Robertson 1968; Janssen and Quinn 1985; Janssen and Luebke 2004). Higher abundances of zebra mussels *Dreissena polymorpha* in rocky areas of Lake Michigan (Fleischer et al. 2001) might also increase abundance of other invertebrates important in the yellow perch diet (Stewart et al. 1998; Kuhns and Berg 1999; Cobb and Watzin 2002). From Waukegan, Illinois to Calumet harbor, Indiana, the substrate consists of cobble, sand and gravel; whereas a large area of fine sand exists immediately north of Waukegan (Foster and Folger 1994). The presence of cobble substrate along and to the south of the tagging sites is consistent with the preferred direction of movement for most fish tagged in Illinois. Substrate along western Indiana (i.e., west of Gary, IN) consists of a mixture of cobble, sand and gravel, whereas the eastern portion is mainly silty-sand (Foster and Folger 1994) and predominantly sand to the north of IN-1 (Powers and Robertson 1968). The availability of cobble substrate to the west of IN-1 may explain why fish tagged there preferred to move west during both time periods. Movement from GB-1, where the substrate is mostly sand, was generally random during both periods, which confirms previous findings (Mraz 1951). Despite the predominance of sandy substrate, approximately 8 hectares of cobble is present at the tagging locations in Michigan (D. Clapp, MDNR, personal communication). However, this did not seem to strongly influence movement of perch in Michigan, yet may have played a role in the fidelity observed at those sites (see Objective 3). Further research should be conducted to directly test the association of perch with different substrate types.

Where cobble substrate is lacking, other important habitat features or factors, biotic and abiotic, might become important for yellow perch movement and/ or habitat selection. For example, despite the lack of cobble substrate on the eastern side of Lake Michigan (Powers and Robertson 1968) fish from MI-2 preferred to travel north during summer, as did those from both Michigan sites during non-summer months. To the north of these sites are a number of streams, rivers, and harbors near or in which many yellow perch were recovered. Tendencies to select these areas could be a function of early yellow perch evolution primarily as a riverine species (Collette et al. 1977) because these areas potentially provide access to backwater spawning areas or food. Use of Great Lakes wetlands by yellow perch has been documented by Chubb and Liston (1986) and Jude and Pappas (1992). Movement of prespawning yellow perch into stream tributaries has been documented in Lakes Michigan (MDOC 1942), Huron (MDOC 1936), and also in southern Lake Superior (Manion 1977).

We observed increased movement during non-summer months compared to summer months for all areas within the southern basin of Lake Michigan. The summer period includes the spawning period during which many fish tend to linger in spawning areas, particularly males (Muncy 1958). The population was largely skewed toward males at the time of tagging,[Wilberg, in review #537; Marsden, 2004 #501] thereby increasing the number of fish staying on or near spawning grounds for a longer time period and increasing the probability of recovering fish close to spawning areas. As fall approaches and the lake begins to cool, yellow perch have been thought to seek warmer water, causing them to travel lakeward toward deeper water (Schaefer 1977). However, our research did not effectively sample offshore areas and angling is directed mainly along the shoreline (Francis et al. 1996). Therefore, offshore migration and mixing cannot be validated with these data.

The 90% dispersal models indicated that yellow perch within the southern basin traversed management boundaries to a high extent during summer, increasing in magnitude when considering the entire study period. According to these models, only fish from Green Bay and Wisconsin stayed within local management boundaries during both time periods. However, considering movement of adult yellow perch based solely on dispersal distances would be a rather conservative management approach, given that yellow perch exhibited directional preference during summer and non-summer time periods. For example, although dispersal of fish from four of five Illinois sites crossed into Wisconsin waters during summer, the preferred directional movement was southward from these sites during this time period, thus decreasing the number of fish likely crossing into Wisconsin as well as the amount of mixing with Wisconsin fish. Therefore, both dispersal distance and directional preference should be considered when evaluating movement of yellow perch. This argument could be extended to dispersal during total recovery periods. Directional preference was similar between the two recovery periods (e.g., IL-2 [S], IL-3 [S], IL-4 [S], IL-5 [S], IN-1 [W] and MI-1 [N]), increasing the likelihood of mixing between Illinois and Indiana fish, but decreasing the likelihood of mixing between Indiana and Michigan.

Horns (2001) documented similarities between Wisconsin and Illinois fish in length and condition of adults and their size at age-1. Fish from Indiana were greater in size at age-1 than all other areas, but smaller and in poorer condition as adults. His results suggested a separation for management between

Indiana and Illinois/ Wisconsin. However, our results suggest that mixing is strong between Illinois and Indiana fish as well as between fish from Illinois and Wisconsin. Growth rates of fish among these three states were also similar in our analysis. These results support the contention that fish from Michigan are different from Indiana fish, at least in growth rates, and are largely spatially segregated (Horns 2001). Thus, managers should carefully consider the delineation of biologically significant management boundaries that not only encompass the directed range of yellow perch movements, but also consider differences in population characteristics.

Objective 3. Determine the extent of spawning site fidelity within local populations of yellow perch in the lake. Compare the results of these analyses with those of previous GLFC/USFWS - funded yellow perch genetic analyses.

METHODS:

Logistic regression was used to determine fidelity (F_{ij}) to tagging sites and to other areas for each tagging site over the entire course of the study:

$$F_{ij} = \frac{A_{ij}}{B_i},$$

where A_{ij} is the number of fish released from tagging location i that were recaptured or recovered at location j during subsequent spawning seasons, B_i is the total number of fish released from tagging site i that were recaptured or recovered anywhere in the lake. Probabilistic estimates were calculated using Proc Logistic in SAS version 8.0 (SAS Institute 1999) and converted to percentages by multiplying by 100. Analysis of recaptures and recoveries was limited to spawning seasons (May – July in Lake Michigan; April – June in Green Bay) at least one year after tagging. Fidelity to the tagging location was determined by setting the recapture/ recovery location j equal to the tagging location i. Setting j different from i allowed us to determine whether fidelity occurred at a larger scale (i.e., nearby tagging locations within the same region, if some areas represented transition zones in which fish were caught en route to more preferred spawning areas), as well as the percentage of fish moving across jurisdictional boundaries. In Illinois, IL-2 and IL-4 were excluded from the fidelity analysis, because no or low effort was expended to recapture yellow perch at these sites in later years. Rather than omitting fish recaptured at IL-2 and IL-4 that were released from other sites, they were pooled into all of Illinois waters (i.e., IL).

RESULTS:

Green Bay. Spawning site fidelity was very high at GB-1, with 80% of the recaptured or recovered yellow perch observed at the release site during the spawning season at least one year after tagging (Figure 5). The remaining 20% stayed in Wisconsin waters of Green Bay (Figure 5). These estimates were computed from only 15 recaptures and recoveries (due to closed fishing seasons for much of the spawning season), resulting in large confidence intervals around these estimates.

Wisconsin. At WI-1, 72% of the 57 recaptures and recoveries remained faithful to this location for spawning (Figure 5). The majority of remaining recoveries (21%) occurred within Wisconsin waters; however a few recaptures occurred at Illinois sites (Figure 5).

Illinois. Spawning-site fidelity varied greatly among tagging locations in Illinois (Figure 5). Fish tagged at IL-1 were very faithful, with 74% returning in subsequent spawning seasons (Figure 5). Many of the fish tagged at IL-3 were recaptured or recovered at IL-3 (55%; Figure 5). High percentages of recaptures and recoveries of fish released from IL-5 also occurred at the IL-3 site (Figure 5). The majority of all fish tagged in Illinois were recaptured or recovered in Illinois waters. However, some did venture across state boundaries into Wisconsin, Indiana, or Michigan waters of Lake Michigan (Figures 5).

Indiana. At IN-1, 55% of the fish recaptured and recovered were faithful to their release site (Figure 5). A large portion of the remaining recoveries remained within Indiana waters (27%; Figure 5). However, some fish from IN-1 did disperse across state boundaries, with recaptures or recoveries occurring in all four states within the southern basin (Figure 5).

Michigan. Fish released from MI-1 and MI-2 displayed higher faithfulness for MI-1 (52% and 35%, respectively; Figure 5). The majority of the remaining recoveries from MI-1 and MI-2 were captured within Michigan waters (39% and 37%, respectively; Figure 5). A limited amount of straying across state boundaries occurred with fish released from MI-1 recovered in Indiana and Illinois, and fish released from MI-2 recovered in Illinois (Figure 5).

DISCUSSION:

Yellow perch select spawning areas based on the substrate makeup in Illinois waters of Lake Michigan, specifically targeting areas with greater proportions of cobble (Robillard and Marsden 2001). Faithfulness to rocky substrate that is patchily distributed throughout the southern basin of Lake Michigan (Powers and Robertson 1968; Foster and Folger 1994) could provide a mechanism to isolate groups of yellow perch for the formation of sympatric stocks. Although we did not directly test for differences among substrate types, we did document high fidelity, particularly at sites with cobble substrate. Yet, fish released from Illinios were recaptured/recovered at all other sites within Illinois, suggesting that yellow perch in these waters were not faithful to an exact spawning location, but remain faithful to a much larger area. This contrasts with studies in smaller systems such as Long Lake, MI, where homing to an exact location of displaced yellow perch was documented (Hodgson et al. 1998).

Despite high fidelity to certain sites, straying occurred from all sites, resulting in mixing among all areas in the southern basin except Michigan and Wisconsin. Because this straying occurred during spawning seasons, it increases the chance for gene flow among areas (Slatkin 1987), supporting the conclusion that a homogenous genetic population exists within the southern basin (Miller 2003). Our results demonstrate that movement of adult yellow perch can contribute to the genetic homogeneity within the southern basin, whereas Miller (2003) speculated the homogeneity to be mainly from larval mixing via ocean-like currents (Beletsky et al. 1999). Also, a single yellow perch from IL-5 was recovered in Green Bay, which we considered an outlier in both directional and dispersal models. This type of large-scale movement by a few individuals can contribute to low genetic variability within Lake Michigan.

Although results of our study support the assignment of a single genetic stock within the southern basin of Lake Michigan, the transfer of a few individuals that homogenizes the population should be of less interest to managers because environmentally-induced phenotypic expressions of local environments will still cause groups to respond differently to exploitation (Pawson and Jennings 1996; Swain and Foote 1999).

Objective 4. Estimate local spawning population sizes (where possible).

METHODS:

In addition to the abundance estimation attempts for Green Bay using mark-recapture modeling (Objective 2), we also estimated the local spawning population size in each of the major tagging areas (i.e., Green Bay, Wisconsin, Illinois, Indiana, and Michigan) for each year tagging was conducted using a

Schnabel census (Seber 1982). Information was pooled into a single site by sampling date for areas with more than one site (i.e., Illinois and Michigan).

A major assumption of mark-recapture studies is that loss of marks is negligible (Seber 1982). Estimates of survival and exploitation rates, along with estimates of population size, can be biased owing to loss or shedding of tags (Wetherall 1996). In fact, (Nelson et al. 1980) showed that tag loss can negatively bias survival estimates up to eighteen percent depending on the rate of tag loss. Floy anchor tags were used in this study and have been shown to be shed a significant amount (>60% in some cases) for other species of fish (e.g., Wilbur and Duckrow 1973; Dunning et al. 1987; Haegele 1990; Muoneke 1992; Fabrizio et al. 1999). Some recaptured fish exhibited sores that apparently resulted from tagging, and such wounds could promote mortality and/ or tag shedding (Stobo 1972). Therefore, it is likely that the lake-wide mark-recapture data violates the negligible tag loss assumption. Tag loss can be estimated by releasing double tagged fish and observing the number of tags present on recaptured fish (Beverton and Holt 1957; Seber 1982; McFarlane et al. 1990; Wetherall 1996). These estimates can then be used to correct parameters derived from mark-recapture studies. However, the original study design did not allow for proper estimation of Floy tag loss for yellow perch. Therefore, a double tagging study was conducted in Illinois waters of Lake Michigan during Spring 2003 – Spring 2005 to estimate the tag shedding rate of Floy tags for yellow perch.

RESULTS / DISCUSSION:

Estimating abundance using the Schnabel census method was much more successful than modeling efforts using the program MARK. Local spawning abundances were highly variable among tagging areas and years (Figure 6). Our estimates were 5 to 186 times lower than those derived by catch-at-age modeling for age-2 yellow perch and older (Wilberg et al. in press). However, lower abundance estimates should be expected given that our estimates are site specific, whereas the Wilberg et al. (In press) estimates pertain to all waters of Wisconsin and Illinois. We did observe a generally decreasing trend in spawner abundance through time in Illinois waters, which was also observed using catch-at-age modeling, but was not as drastic (Wilberg et al. In press). Previous catch-at-age modeling also documented generally stable abundance in Wisconsin waters during the tagging study (Wilberg et al. in press). Our estimates showed a large increase in abundance from 1998 to 1999 (Figure 6). The only other discernable pattern was a slight increase in our estimated abundance in Michigan waters (Figure 6).

In 2004, 1,837 adult yellow perch were double tagged in addition to the 2,772 tagged in 2003. To date, 24 fish tagged in 2004 and 38 tagged in 2003 have been recaptured. Only two fish (both tagged in 2003) have been reported with a single tag, which equates to a 0.05 annual tag shedding rate for fish tagged in 2003. This indicates that tag loss is much lower for yellow perch compared to other species. This is similar to a three and a half month tagging study on yellow perch that documented no signs of tag loss with dart tags (Stobo 1972). We are continuing double tagging efforts in 2005.

Objective 5. Estimate local mortality rates and exploitation (where possible).

METHODS:

Local mortality rates were estimated for each state in the southern basin of Lake Michigan, for each site within Illinois, and at GB-1 for Green bay, using mark-recapture models described above (see Objective 2).

Information regarding the angler reporting rate is necessary to separate total mortality into natural mortality and exploitation (Pollock et al. 2001). Typically, the angler tag-reporting rate is estimated with a high reward tagging program that is implemented in conjunction with the tagging study, by releasing reward tags with standard tags (Pollock et al. 2001). However, the original study design did not implement a high reward tagging program to estimate angler reporting rates. We explored the option of using creel data to estimate the tag reporting rate. To accomplish this, we obtained creel information from each state agency concerning the number of yellow perch observed in the creel, the number of tagged yellow perch observed in the creel, and the estimated number of yellow perch harvested during each year of the study. Using the ratio of tagged yellow perch observed in the creel to the total number of yellow perch examined in the creel, the number of tagged perch that should have been reported out of the total number yellow perch harvested could be estimated. This number can then be compared to what was actually reported by anglers to estimate non-reporting rates (Pollock et al. 2001).

RESULTS / DISCUSSION:

Observations of tagged fish in the creel were extremely low throughout the entire study. In Green Bay, Wisconsin, Illinois, Indiana, and Michigan the total number of tagged fish observed in the creel throughout the duration of the tagging study was 1, 0, 13, 14, and 14, respectively. The number of times that no tagged fish were observed in the creel for a year was very frequent, thus making this a poor method for determining the reporting rate. The best chances for successfully estimating local mortality and exploitation rates likely involve the use of reward tagging such that an unbiased estimate of the reporting rate can be determined.

Objective 6. Evaluate the need and, if justified, develop long-range plans for periodic lake-wide yellow perch tagging studies.

METHODS:

Based on the results from Objectives 1-5, we evaluated the need for periodic mark-recapture studies of Lake Michigan yellow perch. We developed a complete picture of yellow perch movements and spawning site fidelity, and attempted to estimate population size and losses due to natural mortality and exploitation. These results provide the basis for our recommendations to the YPTG, LMTC, and LMC regarding the potential need for periodic mark-recapture studies on the yellow perch population, as well as recommendations concerning how the studies should be conducted. These recommendations include how many "populations" need to be sampled per state, as well as how many fish need to be tagged and recovered to meet different objectives, given a variety of population size and population structure scenarios. Future long-range marking plans will be developed in consultation with those involved in other ongoing percid tagging efforts in the Great Lakes. This consultation will likely take place in the form of a percid tagging workshop, to be held in conjunction with a future Lake Michigan YPTG or LMTC meeting.

RESULTS / DISCUSSION:

Although specific design criteria will depend on the questions desired to be answered, we can recommend several important considerations for future tagging studies:

1) Develop concrete questions to be answered at the project design phase such that appropriate tagging and recovery strategies can be employed to best provide the data to answer the question(s) of greatest importance. This does mean that not all questions will be well answered by a given tagging study.
2) Maximizing recoveries is a critical component of eventually being able to answer the desired question. Although mark-recapture experts would like to see over 90% recovery of tagged individuals, we recognize the near impossibility of achieving that level of recovery in a fish population study in a system as large as Lake Michigan. We suggest that a 30% recovery level

would be a reasonable recovery target when planning future studies. In comparison, for this study, recovery/recaptures were about 8%.

3) Mark-recapture experts typically prefer long-term tag recovery commitments because the data sets improve with each additional year that the study is conducted (Lebreton et al. 1992). Long-term data sets can be more useful than shorter studies because they permit observation of population trends over a long period of time, allowing for relative comparisons through time. The biology and life history of the species being studied needs to be considered, however, to determine the appropriate length of a study. For instance, 92% of all recapture/recoveries occurred within the first two years of release in this study. Thus, focusing on obtaining another 8% of all recoveries in later years might not have added much information. The few fish recaptured after two years at liberty may have resulted from tagging during a time when the age structure of the population consisted primarily of older fish. With a broader (or younger) age distribution it is possible to obtain more recaptures through time, increasing the usefulness of a longer study.
4) Our results indicate tagging over a broader spatial scale is generally better than tagging at smaller scales at the state level. Increasing the spatial coverage of tagging sites is critical to maximizing the information gained for movement analyses. Not only will increasing the spatial coverage of tagging give interesting insight into movement of fish from different habitat types and near management boundaries, but it will also increase the spatial coverage of recaptures that occur during tagging operations. In this study, 75% of all recovery/recaptures were from agency sources. Yet, we were unable to use agency recaptures for movement analyses because of their poor spatial coverage. Future studies should consider tagging and recapturing adult yellow perch on a predetermined grid or intervals along the shoreline to increase the spatial coverage of our most reliable source of recaptures.
5) We used two types of closed population estimation procedures in this study (i.e., Pollock's robust design and a Schnabel census) and neither produced reliable estimates. If local spawning abundance estimation using mark-recapture data is a major goal of the YPTG, we suggest the recapture rate be at least 25% within a year to obtain precise estimates. Because this may not be a realistic goal, managers might consider focusing their efforts toward other questions with mark-recapture data.
6) Increasing the number of recapture/recoveries in years subsequent to tagging is essential to estimating total mortality using mark-recapture models. In this study, 62% of recapture/recoveries (excluding commercial recoveries) occurred within the first year of release, causing these encounters to be useless in the estimation of annual mortality. However, tag-induced morality and tag loss will affect how many recaptures are possible in later years. We feel that this is an important consideration in developing long term plans for future tagging efforts not only to obtain more accurate estimates, but also to compensate for tag shedding and tag-induced mortality by tagging more individuals. Our results suggest that the use of creel data to estimate tag non-reporting rate is ineffective. Instead, high-reward tagging should be considered as a method to estimate non-reporting rate to separate total morality into natural and fishing sources.

Because of the need to prioritize questions of importance, we recommend that tagging workshops be held in the future if the YPTG or other groups are interested in other tagging studies. Ideally, the workshop would include an evaluation of research priorities, design planning, logistical considerations, determination of the best strategies for recovering tags, analysis strategies, and database needs.

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TABLES & FIGURES:

Table 1. Location identifier, latitude (lat.) and longitude (long.) of each tagging location, years sampled, number of adult yellow perch tagged, and number of recaptures/ recoveries by source. Recaptures and recoveries are from 1996 to 2001 for Illinois tagging sites and from 1997 to 2001 for all other sites. GB-1 = Little Tail Point; WI-1 = Green Can Reef; IL-1 = Waukegan Wiremill; IL-2 = Lake Bluff; IL-3 = North Lake Forest; IL-4 = South Lake Forest; IL-5 = Fort Sheridan; IN-1 = Mt. Baldy; MI-1 = North St. Joseph; MI-2 = South St. Joseph.

Location identifier	Tagging location (Lat., Long.)	Years sampled	Number tagged	Agency recaptures	Sport recoveries	Commercial recoveries
GB-1	44°39.384′ N, 87°59.571′ W	1997 - 1999	7,198	413	52	104
WI-1	42°59.035′ N, 87°50.250′ W	1997 - 1999	9,615	543	53	0
IL-1	42°20.244′ N, 87°49.462′ W	1997 - 2000	4,633	410	59	0
IL-2	42°16.772′ N, 87°49.502′ W	1996	4,210	424	124	57
IL-3	42°15.280′ N, 87°49.015′ W	1996 - 2000	11,167	658	83	35
IL-4	42°13.950′ N, 87°48.435′ W	1997 - 1999	1,766	123	25	9
IL-5	42°12.789′ N, 87°47.792′ W	1996 - 2000	10,952	522	142	75
IN-1	41°42.912′ N, 86°56.095′ W	1997 - 1999	6,410	233	115	0
MI-1	42°8.166′ N, 86°28.454′ W	1997 - 1999	5,240	312	187	0
MI-2	42°6.154′ N, 86°30.200′ W	1997 - 1999	2,757	163	104	0
Total			**63,948**	**3,801**	**944**	**280**

Table 2. Model identifier, structure and description of each model tested for Green Bay using Pollock's robust design, and for Lake Michigan and Illinois using a multistate live-dead model in the program MARK. The model structure follows that recommended by Lebreton et al. (1992) and that of Kendall et al. (1995), Kendall et al. (1997), Brownie et al. (1993) and Burnham (1993).

Model identifier	Model structure	Model description
		Green Bay
Model 1	$c_{ij}, p_{ij}, \gamma'_i, \gamma''_i, S_i, N_i$	Probability of first capture, recapture, permanent emigration, temporary emigration, survival and abundance all time dependent.
Model 2	$c_{ij} = p_{ij}, \gamma'_i, \gamma''_i, S_i, N_i$	Model 1 with probability of first capture and recapture set equal among secondary sessions (i.e., no trap response).
Model 3	$c_{i.}, p_{i.}, \gamma'_i, \gamma''_i, S_i, N_i$	Model 1 with no time variation across secondary sessions.
Model 4	$c_{..}, p_{..}, \gamma'_i, \gamma''_i, S_i, N_i$	Model 1 with probability of first capture and recapture equal across time for both secondary and primary sessions.
Model 5	$c_{ij}, p_{ij}, \gamma'_i = \gamma''_i = 0, S_i, N_i$	Model 1 with no temporary emigration or permanent emigration.
Model 6	$c_{ij}, p_{ij}, \gamma'_i = \gamma''_i, S_i, N_i$	Model 1 with random emigration.
Model 7	$c_{ij}, p_{ij}, \gamma'_i, \gamma''_., S_i, N_i$	Model 1 with permanent emigration equal across time.
Model 8	$c_{ij}, p_{ij}, \gamma'_i, \gamma''_i, S_., N_i$	Model 1 with survival equal across time.
Model 9	$c_{ij}, p_{ij}, \gamma'_i, \gamma''_i, S_i, N_.$	Model 1 with abundance equal across time.
		Lake Michigan
Model 10	$r^s_., p^s_., \psi^{rs}_., S^s_.$	Probability of recovery, recapture, transition, and survival different among states and similar through time.
Model 11	$r^{IL=IN,s}_., p^s_., \psi^{rs}_., S^s_.$	Model 10 with recovery probability equal between Illinois and Indiana but different among all other states.
Model 12	$r^s_., p^s_., \psi^{rs}_., S^{IL=IN,s}_.$	Model 10 with survival equal between Illinois and Indiana but different among all other states.
Model 13	$r^s_., p^s_., \psi^{rs}_., S^{IL=WI,s}_.$	Model 10 with survival equal between Illinois and Wisconsin but different among all other states.
Model 14	$r^s_., p^s_., \psi^{rs}_., S^{IN=MI,s}_.$	Model 10 with survival equal between Indiana and Michigan but different among all other states.
Model 15	$r^s_., p^s_., \psi^{rs}_., S_.$	Model 10 with survival equal among all states.

Table 2, cont.

		Illinois
Model 16	$r_{\cdot}, p_{\cdot}, \psi_{\cdot}^{\cdot\cdot}, S_{\cdot}$	Recovery, recapture, transition, and survival probability equal among years and sites.
Model 17	$r_{i}, p_{\cdot}, \psi_{\cdot}^{\cdot\cdot}, S_{\cdot}$	Model 16 with recovery probability time dependent.
Model 18	$r_{\cdot}^{s}, p_{\cdot}, \psi_{\cdot}^{\cdot\cdot}, S_{\cdot}$	Model 16 with recovery probability different among sites.
Model 19	$r_{i}^{s}, p_{\cdot}, \psi_{\cdot}^{\cdot\cdot}, S_{\cdot}$	Model 16 with recovery probability time dependent and different among sites.
Model 20	$r_{\text{regs}}^{\text{IL, LM}}, p_{\cdot}, \psi_{\cdot}^{\cdot\cdot}, S_{\cdot}$	Model 16 with recovery probability different between pre- (1996) and post- (1997-2000) harvest regulation change for Illinois.
Model 21	$r_{i}^{s}, p_{i}, \psi_{\cdot}^{\cdot\cdot}, S_{\cdot}$	Model 19 with recapture probability time dependent and equal across strata.
Model 22	$r_{i}^{s}, p_{\cdot}, \psi_{\cdot}^{rs}, S_{\cdot}$	Model 19 with transition probabilities different among sites and equal across time.
Model 23	$r_{i}^{s}, p_{\cdot}, \psi_{\cdot}^{\cdot\cdot} + \text{distance}, S_{\cdot}$	Model 19 with transition probabilities a function of distance to destination strata.
Model 24	$r_{i}^{s}, p_{i}, \psi_{\cdot}^{rs}, S_{\text{regs}}^{\text{IL, LM}}$	Model 22 with survival probability different between pre- (1996) and post- (1997-2000) regulation change for Illinois.
Model 25	$r_{i}^{s}, p_{\cdot}, \psi_{\cdot}^{rs}, S_{i}$	Model 22 with survival probability time dependent.
Model 26	$r_{i}^{s}, p_{\cdot}, \psi_{\cdot}^{rs}, S_{\cdot}^{s}$	Model 22 with survival probability different among sites.
Model 27	$r_{i}^{s}, p_{\cdot}, \psi_{\cdot}^{rs}, S_{i}^{s}$	Model 22 with survival probability time dependent and different among sites.

Table 3. Model and corresponding AIC_c, Δ AIC_c from the best model, AIC_c weight, model likelihood and the number of estimable parameters observed to the number theoretically estimable (Est. par. / theoretical) for Green Bay, Lake Michigan and Illinois.

Model	AIC_c	Δ AIC_c	AIC_c weight	Model likelihood	Est. par. / theoretical
			Green Bay		
Model 1	-65690.60	0.00	0.20	1.00	37 / 53
Model 6	-65690.60	0.00	0.20	1.00	37 / 52
Model 8	-65690.60	0.00	0.20	1.00	37 / 52
Model 7	-65690.60	0.00	0.20	1.00	37 / 52
Model 9	-65690.60	0.00	0.20	1.00	37 / 52
Model 5	-65683.04	7.56	0.00	0.02	36 / 50
Model 2	-65669.27	21.33	0.00	0.00	20 / 32
Model 4	-61846.89	3843.72	0.00	0.00	8 / 10
Model 3			Failed to converge		
			Lake Michigan		
Model 10	151760.29	0.00	1.00	1.00	14 / 24
Model 12	151783.21	22.92	0.00	0.00	15 / 23
Model 15	463358.37	311598.08	0.00	0.00	15 / 23
Model 13	573944.87	422184.59	0.00	0.00	20 / 23
Model 11			Failed to converge		
Model 14			Failed to converge		
			Illinois		
Model 22	84911.40	0.00	1.00	1.00	39 / 62
Model 19	86951.38	2039.97	0.00	0.00	21 / 33
Model 17	87687.49	2776.09	0.00	0.00	8 / 8
Model 23	87813.81	2902.40	0.00	0.00	22 / 34
Model 20	90300.69	5389.29	0.00	0.00	5 / 6
Model 21	111480.83	26569.43	0.00	0.00	12 / 36
Model 16	115890.98	30979.58	0.00	0.00	4 / 4
Model 18	115995.11	31083.71	0.00	0.00	7 / 9
Model 25	169311.78	84400.37	0.00	0.00	48 / 66
Model 24			Failed to converge		
Model 26			Failed to converge		
Model 27			Failed to converge		

Table 4. Parameter estimates, standard errors (SE), and 95% confidence intervals from Model 1 ($p_{ij}, c_{ij}, N_i, S_i, \gamma'_i, \gamma''_i$; i.e., completely time dependent) for Green Bay (GB-1). Estimates were derived using Pollock's robust design in the Program MARK.

Parameter	Estimate	SE	Lower 95% C.I.	Upper 95% C.I.
N_{1997}	1790.00	0.00	1790.00	1790.00
N_{1998}	2261.00	0.00	2261.00	2261.00
N_{1999}	3003.00	0.00	3003.00	3003.00
$S_{1997\text{-}1998}$	0.01	0.95	0.00	1.00
$S_{1998\text{-}1999}$	0.93	49.47	0.00	1.00
$\gamma''_{1997-1998}$	0.84	21.00	0.00	1.00
$\gamma''_{1998-1999}$	1.00	0.10	0.00	1.00
γ'_{1998}	0.80	33.97	0.00	1.00
$p_{1997,1}$	0.09	0.01	0.07	0.10
$p_{1997,2}$	0.03	0.00	0.02	0.04
$p_{1997,3}$	0.10	0.01	0.09	0.12
$p_{1997,4}$	0.41	0.01	0.39	0.44
$p_{1997,5}$	0.68	0.02	0.65	0.71
$p_{1997,6}$	1.00	0.00	0.00	1.00
$p_{1998,1}$	0.03	0.00	0.02	0.03
$p_{1998,2}$	0.06	0.01	0.05	0.07
$p_{1998,3}$	0.25	0.01	0.23	0.27
$p_{1998,4}$	0.22	0.01	0.20	0.24
$p_{1998,5}$	0.05	0.01	0.04	0.07
$p_{1998,6}$	0.46	0.01	0.43	0.49
$p_{1998,7}$	0.85	0.01	0.82	0.87
$p_{1998,8}$	1.00	0.00	1.00	1.00
$p_{1999,1}$	0.21	0.01	0.20	0.23
$p_{1999,2}$	0.20	0.01	0.18	0.22
$p_{1999,3}$	0.04	0.00	0.04	0.06
$p_{1999,4}$	0.16	0.01	0.15	0.18
$p_{1999,5}$	0.23	0.01	0.21	0.26
$p_{1999,6}$	0.43	0.01	0.40	0.46
$p_{1999,7}$	0.37	0.02	0.33	0.41
$p_{1999,8}$	0.64	0.02	0.59	0.68

Table 4, cont.

$p_{1999,9}$	1.00	0.00	1.00	1.00
$p_{1999,10}$	1.00	1.61	0.00	1.00
$c_{1997,2}$	0.00	0.00	0.00	0.00
$c_{1997,3}$	0.00	0.00	0.00	0.00
$c_{1997,4}$	0.02	0.01	0.01	0.04
$c_{1997,5}$	0.02	0.00	0.02	0.04
$c_{1997,6}$	0.00	0.00	0.00	0.01
$c_{1998,2}$	0.00	0.00	0.00	0.00
$c_{1998,3}$	0.02	0.01	0.01	0.05
$c_{1998,4}$	0.00	0.00	0.00	0.01
$c_{1998,5}$	0.00	0.00	0.00	0.01
$c_{1998,6}$	0.01	0.00	0.00	0.01
$c_{1998,7}$	0.01	0.00	0.00	0.01
$c_{1998,8}$	0.00	0.00	0.00	0.00
$c_{1999,2}$	0.06	0.01	0.04	0.08
$c_{1999,3}$	0.01	0.00	0.01	0.02
$c_{1999,4}$	0.02	0.00	0.01	0.03
$c_{1999,5}$	0.02	0.00	0.02	0.03
$c_{1999,6}$	0.04	0.00	0.03	0.05
$c_{1999,7}$	0.02	0.00	0.02	0.03
$c_{1999,8}$	0.02	0.00	0.02	0.03
$c_{1999,9}$	0.01	0.00	0.01	0.02
$c_{1999,10}$	0.00	0.00	0.00	0.00

Table 5. Parameter estimates, standard errors (SE), and 95% confidence intervals from Model 10 (r^s, p^s, ψ^{rs}, S^s) for Lake Michigan. Estimates were derived using a multistate live-dead model in the Program MARK.

Parameter	Estimate	SE	Lower 95% C.I.	Upper 95% C.I.
S^{IL}	0.99	0.00	0.99	0.99
S^{IN}	0.48	0.06	0.37	0.59
S^{MI}	0.30	0.02	0.25	0.34
S^{WI}	0.59	0.08	0.44	0.73
$\psi^{IL, IN}$	0.04	0.00	0.04	0.05
$\psi^{IL, MI}$	0.02	0.00	0.01	0.03
$\psi^{IL, WI}$	0.07	0.05	0.02	0.25
$\psi^{IN, IL}$	0.00	0.00	0.00	0.00
$\psi^{IN, MI}$	0.00	0.00	0.00	0.00
$\psi^{IN, WI}$	0.02	0.01	0.00	0.06
$\psi^{MI, IL}$	0.48	0.01	0.45	0.51
$\psi^{MI, IN}$	0.00	0.00	0.00	0.00
$\psi^{MI, WI}$	0.02	0.01	0.00	0.08
$\psi^{WI, IL}$	0.00	0.00	0.00	0.00
$\psi^{WI, IN}$	1.00	0.00	1.00	1.00
$\psi^{WI, MI}$	0.00	0.00	0.00	0.00
p^{IL}	0.00	0.00	0.00	0.00
p^{IN}	0.00	0.00	0.00	1.00
p^{MI}	0.00	0.00	0.00	0.00
p^{WI}	0.00	0.00	0.00	1.00
r^{IL}	1.00	0.00	1.00	1.00
r^{IN}	0.02	0.00	0.01	0.02
r^{MI}	0.01	0.00	0.01	0.02
r^{WI}	0.00	0.00	0.00	0.00

Table 6. Parameter estimates, standard errors (SE), and 95% confidence intervals from Model 22 ($r_i^s, p_\cdot, \psi_\cdot^{rs}, S_\cdot$) for Illinois. Estimates were derived using a multistate live-dead model in the Program MARK. IL-1 = Waukegan Wiremill; IL-2 = Lake Bluff; IL-3 = North Lake Forest; IL-4 = South Lake Forest; IL-5 = Fort Sheridan; LM = all other areas in Lake Michigan.

Parameter	Estimate	SE	Lower 95% C.I.	Upper 95% C.I.
$S_\cdot$	0.12	0.00	0.11	0.12
$\psi_\cdot^{IL\text{-}1,\,IL\text{-}2}$	1.00	0.00	1.00	1.00
$\psi_\cdot^{IL\text{-}1,\,IL\text{-}3}$	0.71	0.02	0.67	0.75
$\psi_\cdot^{IL\text{-}1,\,IL\text{-}4}$	0.03	0.01	0.02	0.05
$\psi_\cdot^{IL\text{-}1,\,IL\text{-}5}$	0.00	0.00	0.00	0.02
$\psi_\cdot^{IL\text{-}1,\,LM}$	0.01	0.00	0.00	0.02
$\psi_\cdot^{IL\text{-}2,\,IL\text{-}1}$	0.02	0.01	0.01	0.04
$\psi_\cdot^{IL\text{-}2,\,IL\text{-}3}$	0.11	0.01	0.09	0.13
$\psi_\cdot^{IL\text{-}2,\,IL\text{-}4}$	0.19	0.01	0.17	0.22
$\psi_\cdot^{IL\text{-}2,\,IL\text{-}5}$	0.02	0.01	0.02	0.04
$\psi_\cdot^{IL\text{-}2,\,LM}$	0.11	0.01	0.09	0.13
$\psi_\cdot^{IL\text{-}3,\,IL\text{-}1}$	0.02	0.00	0.01	0.03
$\psi_\cdot^{IL\text{-}3,\,IL\text{-}2}$	0.02	0.00	0.01	0.03
$\psi_\cdot^{IL\text{-}3,\,IL\text{-}4}$	0.81	0.01	0.79	0.84
$\psi_\cdot^{IL\text{-}3,\,IL\text{-}5}$	0.01	0.00	0.01	0.02
$\psi_\cdot^{IL\text{-}3,\,LM}$	0.04	0.01	0.03	0.05
$\psi_\cdot^{IL\text{-}4,\,IL\text{-}1}$	0.01	0.00	0.00	0.01
$\psi_\cdot^{IL\text{-}4,\,IL\text{-}2}$	0.04	0.02	0.01	0.09
$\psi_\cdot^{IL\text{-}4,\,IL\text{-}3}$	0.69	0.04	0.60	0.77
$\psi_\cdot^{IL\text{-}4,\,IL\text{-}5}$	0.14	0.03	0.09	0.22
$\psi_\cdot^{IL\text{-}4,\,LM}$	0.09	0.03	0.05	0.16
$\psi_\cdot^{IL\text{-}5,\,IL\text{-}1}$	0.01	0.01	0.00	0.06
$\psi_\cdot^{IL\text{-}5,\,IL\text{-}2}$	0.01	0.00	0.00	0.02
$\psi_\cdot^{IL\text{-}5,\,IL\text{-}3}$	0.79	0.01	0.76	0.82
$\psi_\cdot^{IL\text{-}5,\,IL\text{-}4}$	0.11	0.01	0.09	0.13
$\psi_\cdot^{IL\text{-}5,\,LM}$	0.01	0.00	0.01	0.02
$\psi_\cdot^{LM,\,IL\text{-}1}$	0.00	0.00	0.00	0.01
$\psi_\cdot^{LM,\,IL\text{-}2}$	0.00	0.00	0.00	0.01
$\psi_\cdot^{LM,\,IL\text{-}3}$	1.00	0.00	1.00	1.00
$\psi_\cdot^{LM,\,IL\text{-}4}$	0.00	0.00	0.00	0.00
$\psi_\cdot^{LM,\,IL\text{-}5}$	0.00	0.00	0.00	0.00
$p_\cdot$	0.00	0.00	0.00	0.00
$r_{1996}^{IL\text{-}1}$	1.00	0.00	1.00	1.00

Table 6, cont.

$r_{1997}^{IL\text{-}1}$	1.00	0.00	1.00	1.00
$r_{1998}^{IL\text{-}1}$	1.00	0.00	1.00	1.00
$r_{1999}^{IL\text{-}1}$	0.98	0.01	0.95	0.99
$r_{2000}^{IL\text{-}1}$	1.00	0.00	1.00	1.00
$r_{1996}^{IL\text{-}2}$	0.03	0.00	0.02	0.03
$r_{1997}^{IL\text{-}2}$	0.04	0.01	0.02	0.05
$r_{1998}^{IL\text{-}2}$	0.04	0.01	0.03	0.06
$r_{1999}^{IL\text{-}2}$	0.00	0.00	0.00	0.00
$r_{2000}^{IL\text{-}2}$	0.00	0.00	0.00	0.00
$r_{1996}^{IL\text{-}3}$	0.01	0.00	0.01	0.02
$r_{1997}^{IL\text{-}3}$	1.00	0.00	1.00	1.00
$r_{1998}^{IL\text{-}3}$	1.00	0.00	1.00	1.00
$r_{1999}^{IL\text{-}3}$	1.00	0.00	1.00	1.00
$r_{2000}^{IL\text{-}3}$	1.00	0.00	1.00	1.00
$r_{1996}^{IL\text{-}4}$	0.02	0.01	0.01	0.04
$r_{1997}^{IL\text{-}4}$	1.00	0.00	0.00	1.00
$r_{1998}^{IL\text{-}4}$	1.00	0.00	1.00	1.00
$r_{1999}^{IL\text{-}4}$	0.00	0.00	0.00	0.00
$r_{2000}^{IL\text{-}4}$	1.00	0.00	1.00	1.00
$r_{1996}^{IL\text{-}5}$	0.02	0.00	0.02	0.03
$r_{1997}^{IL\text{-}5}$	1.00	0.00	0.00	1.00
$r_{1998}^{IL\text{-}5}$	1.00	0.00	0.00	1.00
$r_{1999}^{IL\text{-}5}$	1.00	0.00	1.00	1.00
$r_{2000}^{IL\text{-}5}$	1.00	0.00	1.00	1.00
r_{1996}^{LM}	1.00	0.00	1.00	1.00
r_{1997}^{LM}	1.00	0.00	1.00	1.00
r_{1998}^{LM}	0.00	0.00	0.00	1.00
r_{1999}^{LM}	0.00	0.02	0.00	1.00
r_{2000}^{LM}	1.00	0.00	1.00	1.00

Table 7. Estimated angler effort (hr) directed at yellow perch for summer (June – August) and non-summer recovery periods for each creel location (county or port) from 1996 to 2001. A dash (-) indicates that effort was not estimated.

Creel location	Year 1996	1997	1998	1999	2000	2001
			Summer			
Door County (GB)	108,264	63,506	67,161	68,708	55,042	42,327
Marinette County	15,572	3,090	2,026	4,792	1,801	9,183
Brown County	69,279	35,589	37,043	42,808	32,740	34,448
Oconto County	40,196	20,510	15,582	31,094	14,659	14,337
Ozaukee County	3,174	1,944	3,789	2,966	5,361	11,004
Milwaukee County	9,198	8,244	10,313	18,384	17,277	64,777
Racine County	10,974	9,033	7,182	11,632	7,851	28,512
Kenosha County	17,987	9,570	8,190	12,038	11,281	31,112
North Point	8,945	2,685	1,185	592	603	3,210
Waukegan	18,821	7,680	8,136	9,456	8,355	17,096
Montrose	99,001	54,841	38,688	44,251	37,599	45,236
Diversey	17,951	9,397	2,468	4,479	2,590	2,806
Burnham	36,875	8,970	9,214	8,718	8,652	14,828
Jackson	11,311	6,521	3,401	6,413	3,364	5,614
Calumet	11,914	7,016	3,075	3,226	4,776	3,530
East Chicago	31,777	7,797	12,044	16,808	21,800	30,908
Hammond	10,792	3,992	2,954	12,360	10,629	12,520
Burns	13,746	9,953	10,327	29,155	27,602	29,765
Michigan City	15,105	10,435	10,795	24,433	22,031	39,186
New Buffalo	4,452	13,612	6,763	20,838	7,572	14,231
St. Joseph	15,030	13,972	20,539	29,438	17,020	23,914
South Haven	10,419	40,497	23,305	28,266	16,141	21,931
Saugatuck	-	-	2,918	-	-	-
Holland	8,813	14,444	-	5,169	5,262	10,804
Port Sheldon	-	-	-	4,530	5,628	12,780
Grand Haven	16,199	26,676	19,270	18,864	22,004	39,266
Muskegon	14,226	14,813	8,738	3,034	27,599	13,280

Table 7, cont.

			Non-summer			
Door County (GB)	26,243	20,647	19,893	20,711	15,351	10,735
Marinette County	5,123	4,689	3,901	5,000	1,646	1,272
Brown County	22,370	20,992	29,093	22,701	13,657	16,347
Oconto County	19,703	14,676	9,538	16,679	10,489	5,420
Ozaukee County	301	44	52	548	108	690
Milwaukee County	3,737	776	4,335	10,834	10,228	30,731
Racine County	1,169	0	1,339	981	764	1,144
Kenosha County	1,475	1,446	5,119	2,745	2,045	5,417
North Point	102	0	207	609	0	0
Waukegan	2,162	60	462	294	396	5,256
Montrose	11,334	16,381	24,995	11,316	16,142	18,273
Diversey	612	2,170	3,242	2,794	1,988	2,408
Burnham	2,378	752	1,499	1,988	2,409	1,424
Jackson	300	36	114	220	1,271	29
Calumet	681	304	2,502	1,326	1,072	1,414
East Chicago	1,512	930	746	1,936	6,651	3,999
Hammond	953	89	665	625	1,632	1,238
Burns	1,563	153	1,003	2,747	4,130	3,691
Michigan City	912	589	1,591	2,559	2,061	1,463
New Buffalo	1,273	84	21	915	1,430	31
St. Joseph	10,208	2,479	3,143	10,803	1,587	1,175
South Haven	6,358	32,422	25,418	75,451	40,918	20,344
Saugatuck	-	-	4,925	-	-	-
Holland	3,878	2,247	-	11,032	2,127	79
Port Sheldon	-	-	-	4,292	815	1,887
Grand Haven	7,577	22,163	5,485	7,392	2,487	4,379
Muskegon	5,572	17,572	5,485	22,339	5,325	38,195

Table 8. Location identifier, parameter estimates from the exponential sigmoid function with SE in parentheses, number of unique locations at which anglers recovered tagged fish (N), the proportion of total variation explained (R^2) by the exponential sigmoid model, and 90 % dispersal distance (km) estimates for summer and total recovery periods. A dash (-) indicates that the 90% dispersal distance was not determined because the cumulative proportion of RPE (α) did not reach 0.90 during model fitting.

Location identifier	α	β	K	N	R^2	Dispersal distance (km)
			Summer			
GB-1	0.911 (0.059)	3.838 (1.587)	-0.200 (0.059)	14	0.98	28.7
WI-1	0.984 (0.079)	14.052 (5.301)	-0.114 (0.021)	10	0.99	44.0
IL-1	0.897 (0.025)	8.808 (4.178)	-0.493 (0.119)	18	0.99	12.8
IL-2	0.985 (0.048)	2.266 (0.308)	-0.065 (0.011)	23	0.99	49.2
IL-3	0.946 (0.044)	4.169 (0.578)	-0.053 (0.006)	22	0.99	82.9
IL-4	1.025 (0.061)	9.236 (2.087)	-0.079 (0.011)	14	0.99	53.4
IL-5	0.971 (0.016)	14.791 (2.312)	-0.096 (0.006)	29	0.99	54.6
IN-1	0.954 (0.044)	2.342 (0.399)	-0.083 (0.015)	22	0.99	44.3
MI-1	1.104 (0.201)	5.222 (1.316)	-0.039 (0.010)	20	0.94	101.4
MI-2	1.015 (0.095)	3.183 (0.740)	-0.038 (0.011)	16	0.97	84.9
			Total			
GB-1	0.933 (0.046)	3.878 (0.953)	-0.147 (0.026)	21	0.99	31.6
WI-1	0.970 (0.031)	11.168 (3.023)	-0.096 (0.011)	13	0.99	51.7
IL-1	0.897 (0.025)	8.808 (4.178)	-0.493 (0.119)	22	0.99	-
IL-2	0.978 (0.043)	2.442 (0.320)	-0.064 (0.009)	24	0.99	52.2
IL-3	1.012 (0.023)	6.382 (0.640)	-0.048 (0.003)	26	0.99	82.0
IL-4	1.017 (0.060)	8.242 (1.792)	-0.071 (0.010)	16	0.99	58.2
IL-5	1.004 (0.022)	13.766 (2.128)	-0.076 (0.005)	32	0.99	62.6
IN-1	1.013 (0.045)	3.072 (0.431)	-0.054 (0.006)	27	0.99	58.8
MI-1	0.954 (0.044)	3.994 (0.533)	-0.042 (0.006)	33	0.98	80.5
MI-2	0.790 (0.044)	30.772 (49.575)	-2.166 (0.972)	20	0.97	-

Table 9. Mean distance traveled (km) determined from sport recoveries, adjusted by directed angler effort for yellow perch (hr) for each direction with respect to the tagging location (SE in parentheses) for summer and non-summer recovery periods. An asterick (*) represents the statistically largest mean of a site ($\alpha = 0.025$). A dash (-) indicates that few or no yellow perch were recovered in that respective direction and was removed from the analysis.

Location identifier	Distance moved from tagging location			
	N	S	E	W
		Summer		
GB-1	12.6 (2.7)	5.0 (4.5)	11.6 (3.4)	-
WI-1	20.5 (4.2)	31.3 (2.1)*	-	-
IL-1	10.7 (3.0)	11.4 (3.2)	-	-
IL-2	10.9 (2.1)	28.7 (2.7)*	-	-
IL-3	19.1 (3.4)	39.6 (4.3)*	-	-
IL-4	22.7 (4.4)	44.2 (5.9)*	-	-
IL-5	24.4 (2.4)	35.1 (2.4)*	-	-
IN-1	-	1.1 (4.5)	9.1 (2.2)	40.5 (3.5)*
MI-1	38.8 (4.5)*	1.0 (5.0)	-	-
MI-2	18.3 (5.1)	19.1 (10.0)	-	-
		Non-summer		
GB-1	-	3.9 (2.3)	20.2 (1.9)*	-
WI-1	6.0 (4.2)	-	-	-
IL-1	23.2 (8.6)	64.6 (4.1)*	-	-
IL-2	11.6 (3.0)	55.2 (3.9)*	-	-
IL-3	33.9 (8.6)	50.6 (2.5)*	-	-
IL-4	12.5 (4.1)	51.2 (4.8)*	-	-
IL-5	27.1 (4.2)	50.0 (1.9)*	-	-
IN-1	-	-	-	49.0 (1.8)
MI-1	62.1 (6.6)	-	-	-
MI-2	43.3 (6.6)	-	-	-

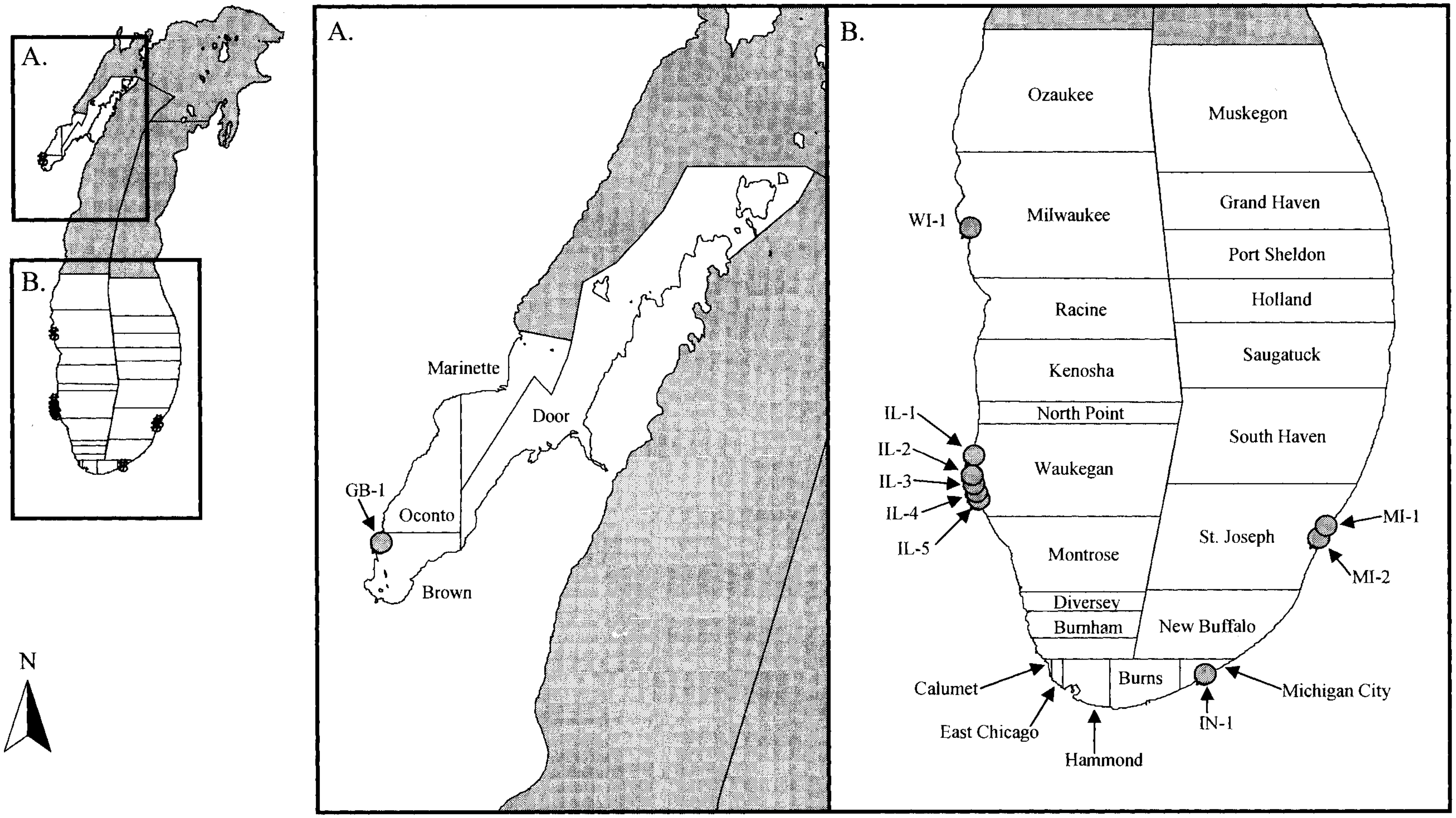

Figure 1. Map of Lake Michigan with inset sections of Green Bay (A.) and southern Lake Michigan (B.), tagging sites (◉) and creel survey sampling locations. Creel units for each port were derived by drawing straight lines from the midpoint between ports to their respective state line. Existing county lines were used to delineate creel units in Wisconsin waters of Lake Michigan and Green Bay. No recoveries occurred in the gray shaded area. GB-1= Little Tail Point; WI-1 = Green Can Reef; IL-1 = Waukegan Wiremill; IL-2 = Lake Bluff; IL-3 = North Lake Forest; IL-4 = South Lake Forest; IL-5 = Fort Sheridan; IN-1 = Mt. Baldy; MI-1 = St. Joseph North; MI-2 = St. Joseph South.

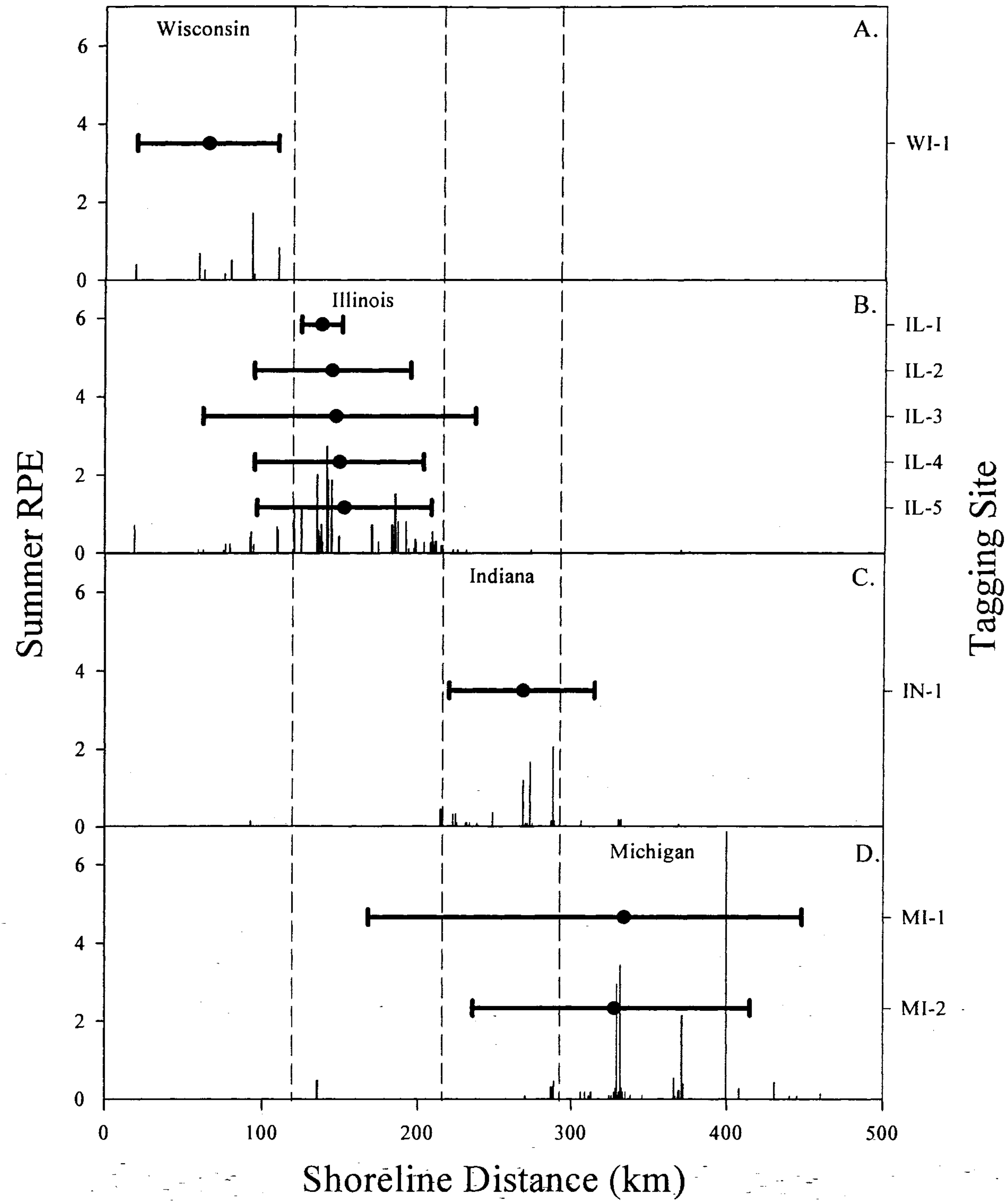

Figure 2. Number of recoveries per effort (RPE) during summer in terms of shoreline distance (km) for Wisconsin (A.), Illinois (B.), Indiana (C.), and Michigan (D.) released fish. Also shown are the tagging sites (solid circles) within the southern basin and the 90% dispersal distance (horizontal bars). Vertical dashed lines represent the point on the scale at which state boundaries occur. The shoreline distance scale begins at the northern border of Ozaukee County, WI.

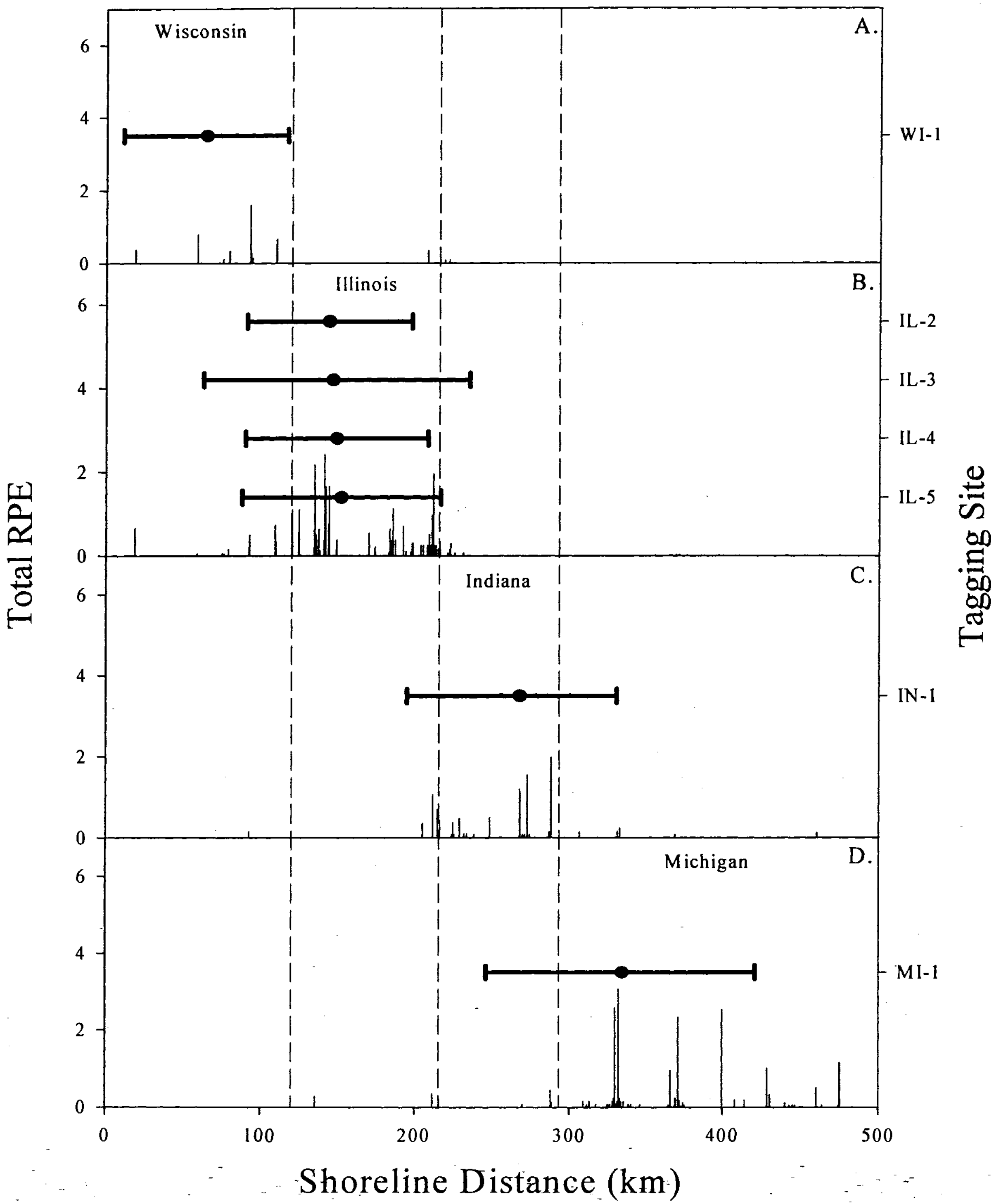

Figure 3. Number of recoveries per effort (RPE) during the total time period in terms of shoreline distance (km) for Wisconsin (A.), Illinois (B.), Indiana (C.), and Michigan (D.) released fish. Also shown are the tagging sites (solid circles) within the southern basin and the 90% dispersal distance (horizontal bars). Vertical dashed lines represent the point on the scale at which state boundaries occur. The shoreline distance scale begins at the northern border of Ozaukee County, WI.

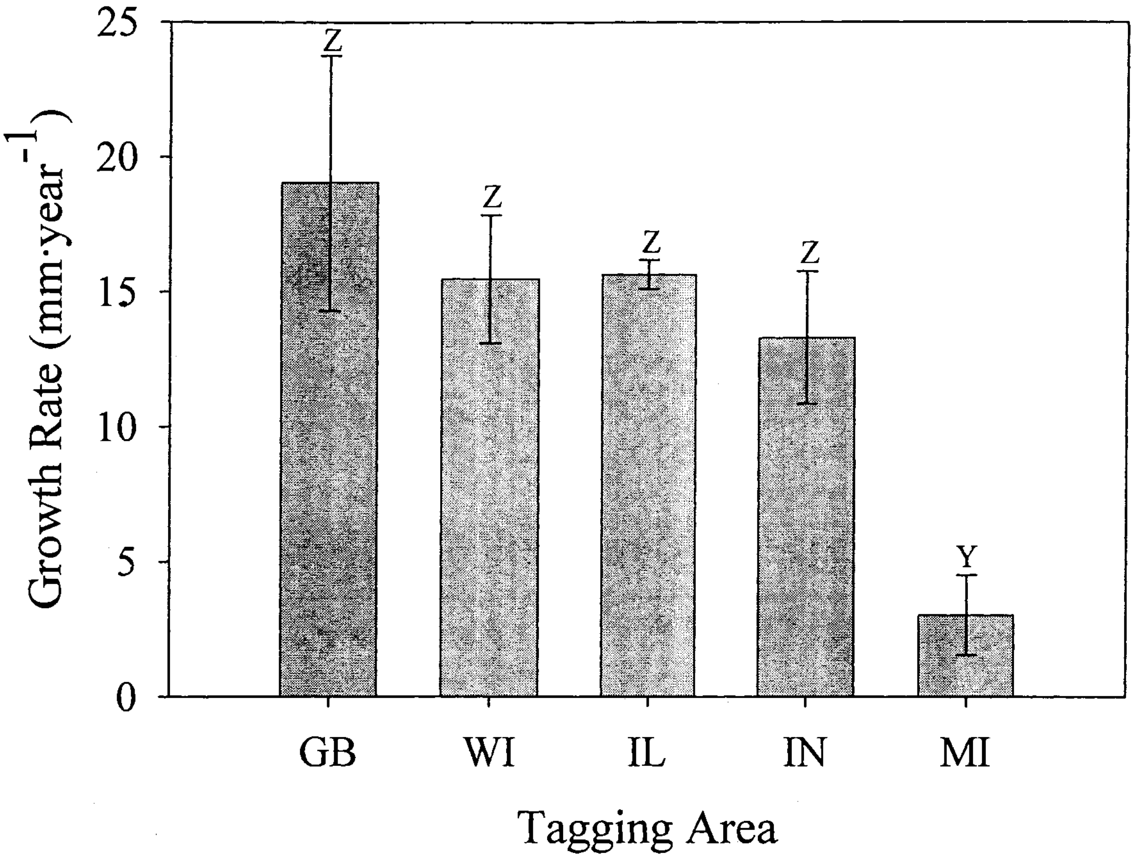

Figure 4. Mean growth rate (mm·year^{-1}) adjusted for time-at-liberty for each tagging area. Error bars represent the 95% confidence interval. Significantly different means ($\alpha = 0.05$) are indicated by a different letter above the error bars.

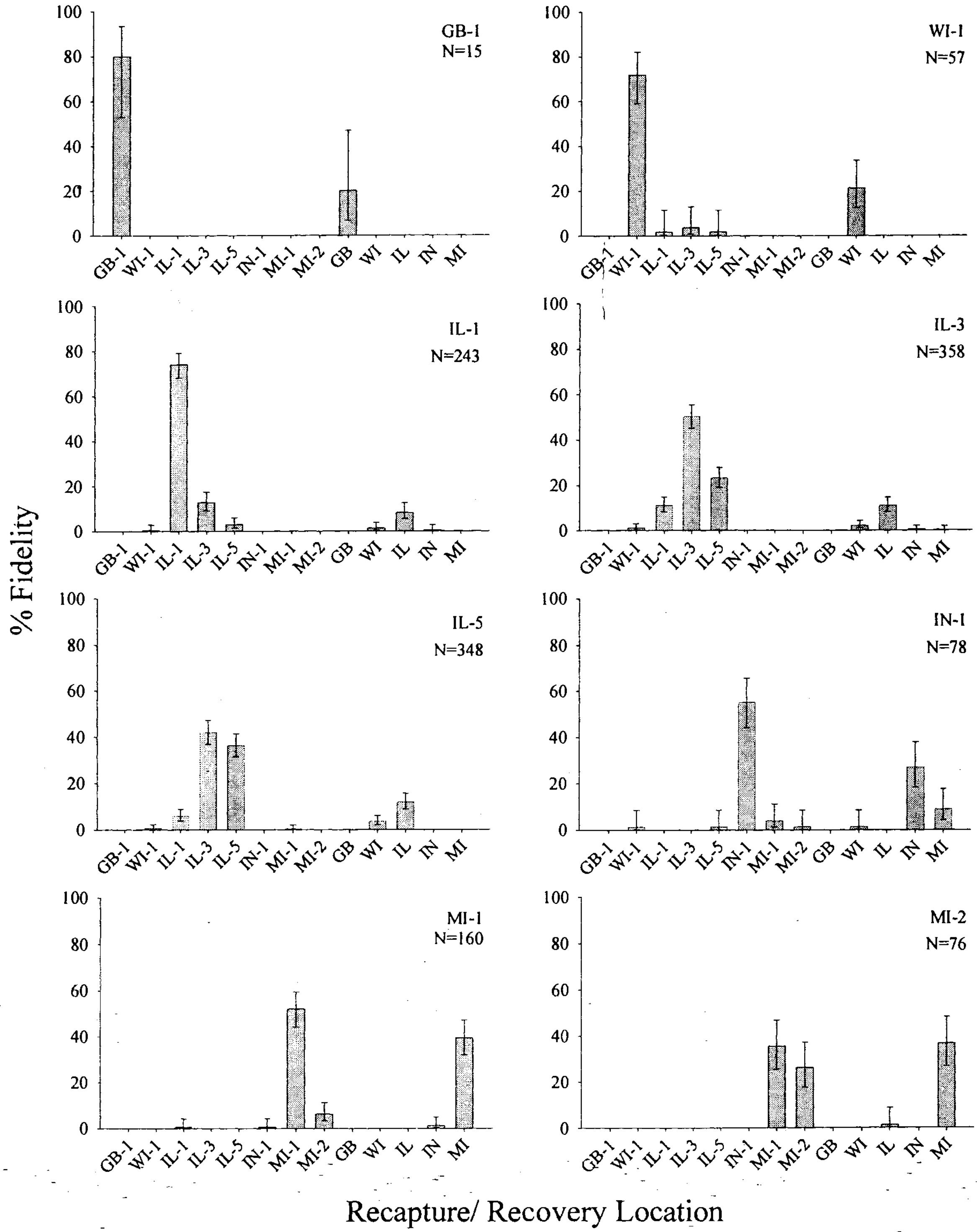

Figure 5. Percent of yellow perch recaptured or recovered at various release locations. Error bars represent the 95% confidence interval.

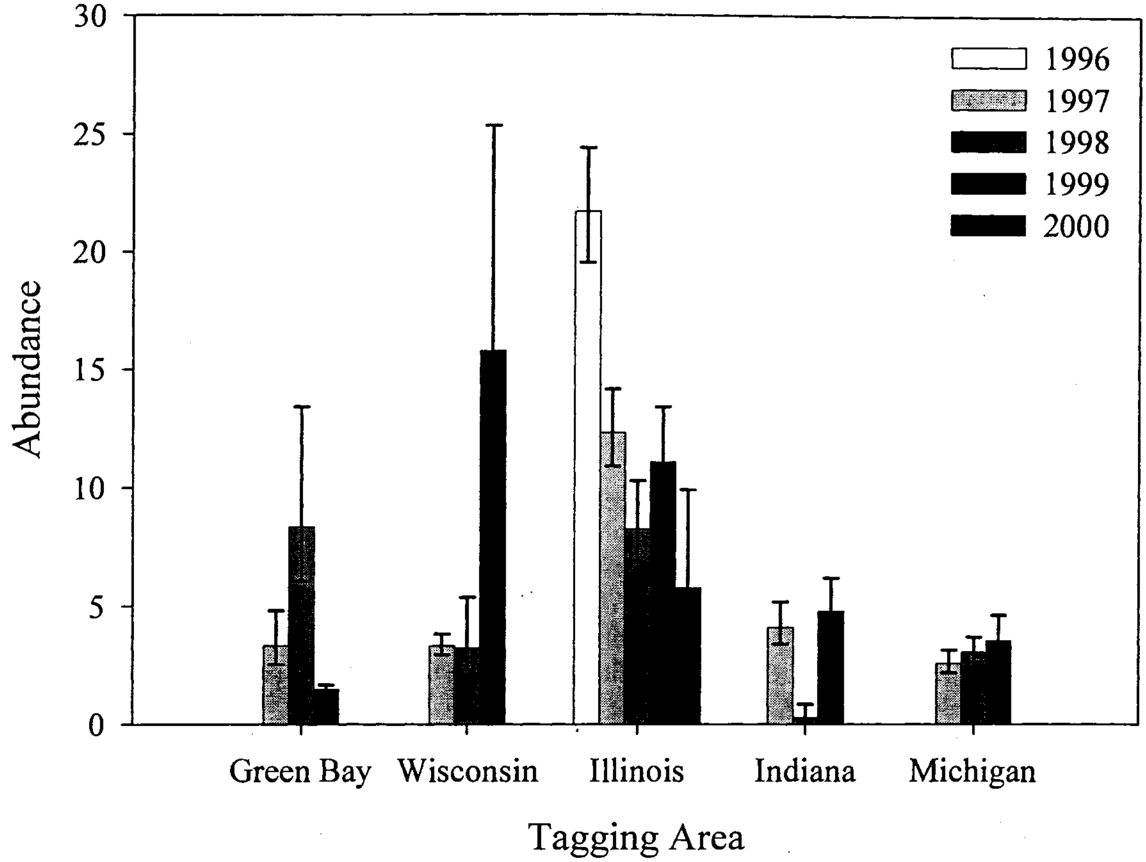

Figure 6. Estimated yellow perch spawning abundance (10,000s) at each tagging area for each year tagging was conducted. Error bars represent the 95% confidence interval.

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DELIVERABLES:

Database Systems:

GLFC Yellow Perch Tagging Database (see Objective 1).

Annual Progress Reports:

Glover, D.C., J. M. Dettmers, and D.F. Clapp. 2004. Evaluation of yellow perch (Perca flavescens) stock discreteness in Lake Michigan: an analysis using lake-wide mark and recapture data. Annual progress report to the Great Lakes Fishery Commission. 16 pp.

Glover, D.C., J. M. Dettmers, and D.F. Clapp. 2003. Evaluation of yellow perch (Perca flavescens) movements in Lake Michigan: an analysis using lake-wide mark and recapture data. Annual progress report to the Great Lakes Fishery Commission. 8 pp.

Oral Presentations:

Glover, D.C., J.M. Dettmers, D.H. Wahl, and D.F. Clapp. 2005. Delineation of yellow perch stocks in Lake Michigan: an analysis from lake-wide mark-recapture. May 26, 2005, 48th Annual International Association of Great Lakes Research Conference, University of Michigan, Ann Arbor, MI.

Glover, D.C., J.M. Dettmers, D.H. Wahl, and D.F. Clapp. 2004. Delineation of yellow perch subpopulations in Lake Michigan: an analysis from lake-wide mark-recapture. December 15, 2004, 64th Annual Midwest Fish and Wildlife Conference, Indianapolis, IN.

Glover, D.C., J.M. Dettmers, and D.F. Clapp. 2004. Lake-wide mark-recapture investigation of adult yellow perch in Lake Michigan. November 30, 2004, Lake Michigan Yellow Perch Task Group Meeting, Des Plaines, IL.

Glover, D.C., J.M. Dettmers, and D.F. Clapp. 2004. Lake-wide mark-recapture investigation of adult yellow perch in Lake Michigan. March 23, 2004, Lake Michigan Yellow Perch Task Group Meeting, Ypsilanti, MI.

Glover, D.C., J.M. Dettmers, and D.F. Clapp. 2002. Lake-wide mark-recapture investigation of adult yellow perch in Lake Michigan. December 2, 2002, Lake Michigan Yellow Perch Task Group Meeting, Des Moines, IA.

Thesis:

Glover, D.C. 2005. Evaluations of yellow perch stock structure in Lake Michigan: an analysis using mark-recapture data. M.Sc. thesis, University of Illinois at Urbana-Champaign, Urbana, Illinois.

Manuscript in Preparation:

Glover, D.C., J.M. Dettmers, D.H. Wahl, D.F. Clapp. Evaluation of adult yellow perch stock structure in Lake Michigan inferred from movements and growth rates. To be submitted to North Am. J. Fish. Manag. Estimated date for submission - September 2005.

PRESS RELEASE:

In Lake Michigan, yellow perch *Perca flavescens* have experienced poor recruitment since 1989. Continued poor recruitment led to strict harvest regulations on the recreational fishery and a moratorium on commercial fishing in the main lake. To enhance future management of the yellow perch fishery, a large-scale collaborative effort was developed to better understand the population makeup of adult yellow perch within Lake Michigan. Specifically, a lake-wide tagging study occurred from 1996-2001 to document movement patterns of yellow perch. This study allowed researchers to determine the tendency of yellow perch to return to the same spawning areas in later years, the degree of population movement, and evaluate current management boundaries.

Yellow perch tended to return to the same spawning site, with 35 to 80% of spawners returning to their original tagging location in later years. Results from Illinois waters suggest this fidelity was directed toward larger areas rather than specific sites, indicating that large spawning complexes exist. Although most spawning yellow perch returned to the same spawning site, enough individuals strayed to other spawning grounds to promote gene flow and maintain a well mixed genetic population of yellow perch throughout the southern basin of Lake Michigan. Dispersal of yellow perch from their original tagging location ranged from about 13 to over 100 km, which resulted in overlap among yellow perch within the southern basin, particularly between adjacent states. Yellow perch preferred to move toward or within rocky substrate when it was nearby. Movement was generally random without rocky substrate nearby, but local movements may have been directed by small streams, rivers, harbors, and other backwater areas along the Eastern shoreline.

These results generally confirm previous genetic studies that indicate a homogenous genetic population of yellow perch within the southern basin. Movement of adults, although relatively modest compared to more pelagic fishes, was sufficient for thorough mixing to occur. Nevertheless, growth of fish tagged in Michigan waters was less than from other locations. This suggests that the southern basin population is not completely uniform and may display regional differences in characteristics that justify managing portions of the population separately. Most tagged yellow perch remained within their respective management unit, indicating the current management unit delineation is reasonable. Because movement occurred across management boundaries between adjacent states, we recommend that adjacent states not enact drastically different yellow perch harvest regulations.

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