Silver Shiner (Notropis photogenis) Population Abundance in Sixteen Mile Creek, Ontario (2022-2023)

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2026

Canadian Manuscript Report of Fisheries and Aquatic Sciences 3319

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Cat. No. Fs 97-4/3319E-PDF ISBN 978-0-660-98427-8 ISSN 1488-5387

https://doi.org/10.60825/wgbn-vb74

Correct citation for this publication:

Lamothe, K.A., and Drake, D.A.R. 2026. Silver Shiner (*Notropis photogenis*) Population Abundance in Sixteen Mile Creek, Ontario (2022–2023). Can. Manuscr. Rep. Fish. Aquat. Sci. 3319: vi + 16 p. https://doi.org/10.60825/wgbn-vb74

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ABSTRACT

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Reliable estimates of population abundance for freshwater fish species listed under Canada's *Species at Risk Act* (SARA) are essential for guiding conservation efforts, yet such estimates are often outdated or lacking altogether. We developed N-mixture models using 2022–2023 field data to estimate detection probability (p) and site abundance (λ_i) of adult Silver Shiner (*Notropis photogenis*; SARA Threatened) in Sixteen Mile Creek, Ontario, and extrapolated results to the population. Year was identified as a significant covariate for both p and λ_i in the most well-supported N-mixture model. Detection probability in the first seine haul (p_1) was significantly greater in 2022 (0.210; 95% CI: 0.167–0.257) than in 2023 (0.108; 95% CI: 0.074–0.147). In contrast, the mean and variance of λ_i were significantly greater in 2023 (74.01; 95% CI: 0–866) compared to 2022 (17.97; 95% CI: 0–147). Greater and more variable estimates of λ_i in 2023 led to a significantly greater, but more uncertain, population abundance estimate (14,481 adults; 95% CI: 10,481–20,570) than in 2022 (3,631; 95% CI: 3,032–4,413). K-fold cross-validation revealed increased uncertainty in population abundance estimates and the observed year effects on p and λ_i when sampling effort was reduced by 10, 20, or 30 sites from 100, underscoring the importance of sustained effort allocation. These updated population abundance estimates will support future conservation status assessments, recovery planning, and regulatory decision-making under SARA. Further monitoring is needed to better understand annual and seasonal patterns in Silver Shiner distribution and abundance within Sixteen Mile Creek.

RÉSUMÉ

Lamothe, K.A., and Drake, D.A.R. 2026. Silver Shiner (*Notropis photogenis*) Population Abundance in Sixteen Mile Creek, Ontario (2022–2023). Can. Manuscr. Rep. Fish. Aquat. Sci. 3319: vi + 16 p. https://doi.org/10.60825/wgbn-vb74

Des estimations fiables de l'abondance des populations d'espèces de poissons d'eau douce inscrites sur la liste de la *Loi sur les espèces en péril* (LEP) du Canada sont essentielles pour orienter les efforts de conservation, mais les estimations disponibles sont souvent désuètes ou inexistantes. Nous avons élaboré des modèles de mélange de N en utilisant les données de terrain recueillies en 2022 et 2023 pour estimer la probabilité de détection (p) et l'abondance au site (λ_i) du méné miroir adulte (*Notropis photogenis*; espèce menacée selon la LEP) dans le ruisseau Sixteen Mile, en Ontario, et extrapoler les résultats à la population. L'année a été déterminée comme une covariable importante pour p et λ_i dans le modèle de mélange de N le plus appuyé. La probabilité de détection de l'espèce dans le premier trait de senne (p_1) était considérablement plus élevée en 2022 (0,210; IC à 95 % : 0,167 à 0,257) qu'en 2023 (0,108; IC à 95 % : 0,074 à 0,147). Par contre, la moyenne et la variance de λ_i étaient considérablement plus élevées en 2023 (74,01; IC à 95 % : 0 à 866) qu'en 2022 (17,97; IC à 95 % : 0 à 147). Des estimations de λ_i plus élevées et plus variables en 2023 ont généré une estimation de l'abondance de la population (14 481 adultes; IC à 95 % : 10 481 à 20 570) beaucoup plus élevée, mais aussi plus incertaine qu'en 2022 (3 631 adultes; IC à 95 : 3 032 à 4 413). La validation croisée à k blocs a révélé une incertitude accrue dans les estimations de l'abondance de la population et les effets de l'année observés sur p et λ_i lorsque l'effort d'échantillonnage a été réduit de 10, 20 ou 30 sites à partir de 100, ce qui souligne l'importance d'une répartition stable de l'effort. Les estimations actualisées de l'abondance de la population appuieront la planification du rétablissement, la prise de décisions réglementaires en vertu de la LEP et les évaluations du statut de conservation réalisées à l'avenir. D'autres efforts de suivi sont nécessaires pour mieux comprendre les tendances annuelles et saisonnières de la répartition et de l'abondance du méné miroir dans le ruisseau Sixteen Mile.

INTRODUCTION

Species abundance is a fundamental metric in conservation biology that directly informs assessments of extinction risk and recovery potential. Accurate estimates of abundance allow scientists to assess trends, inform recovery strategies and assess their effectiveness, and evaluate the potential impacts of development, habitat alteration, or other anthropogenic stressors (Reynolds et al. 2016; Lindenmayer et al. 2020; Callaghan et al. 2024). However, up-to-date and robust abundance estimates remain unavailable or highly uncertain for many imperilled species, including most of the freshwater fishes listed under Canada's *Species at Risk Act* (SARA 2002; Cooke et al. 2012; Drake et al. 2021). This data deficiency introduces significant ambiguity into conservation assessments and decision-making processes. Without reliable abundance data, it becomes difficult to determine whether a species is recovering, stable, or in decline, and to quantify the potential harm imposed by land-use practices. Addressing gaps in SARA-listed species abundance is therefore essential for evidence-based conservation planning.

One example of a SARA-listed species that lacks a current population abundance estimate is Silver Shiner (*Notropis photogenis*; Threatened). Silver Shiner is a freshwater fish species that occupies pools and runs of riverine systems and feeds on aquatic and terrestrial invertebrates (Burbank et al. 2022). There are five known populations in Canada, all located in southern Ontario. Contaminants, toxic substances, and nutrient loading resulting from urbanization and agriculture have been identified as the greatest threats to Silver Shiner in Canada based on the likelihood and potential magnitude of effects (COSEWIC 2011; Bouvier et al. 2013; Fisheries and Oceans Canada 2022). Population abundance has been estimated for four of the five known populations of Silver Shiner, but the estimates are over 10 years old (Young and Koops 2013). Here, we sought to generate new population abundance estimates for the Sixteen Mile Creek population based on data collected in 2022 and 2023 (Lopez et al. 2024; Lamothe et al. 2025). The results of this work will inform future conservation status assessments, listing considerations, permitting decisions, and recovery actions.

METHODS

STUDY LOCATION

Sixteen Mile Creek is a Lake Ontario tributary located on the western side of the Greater Toronto Area in southern Ontario, Canada (Figure 1). The river runs through both agricultural and urban landscapes, with the land-use becoming increasingly urbanized in downstream reaches. Critical habitat for Silver Shiner is located throughout Sixteen Mile Creek from the upstream reaches south of Derry Line to the outflow at Lake Ontario (Figure 1). Sampling for this study was performed downstream of Britannia Road to the Queen Elizabeth Way (QEW), with sites located within the east branch, west branch, and mainstem of the river (Figure 1).

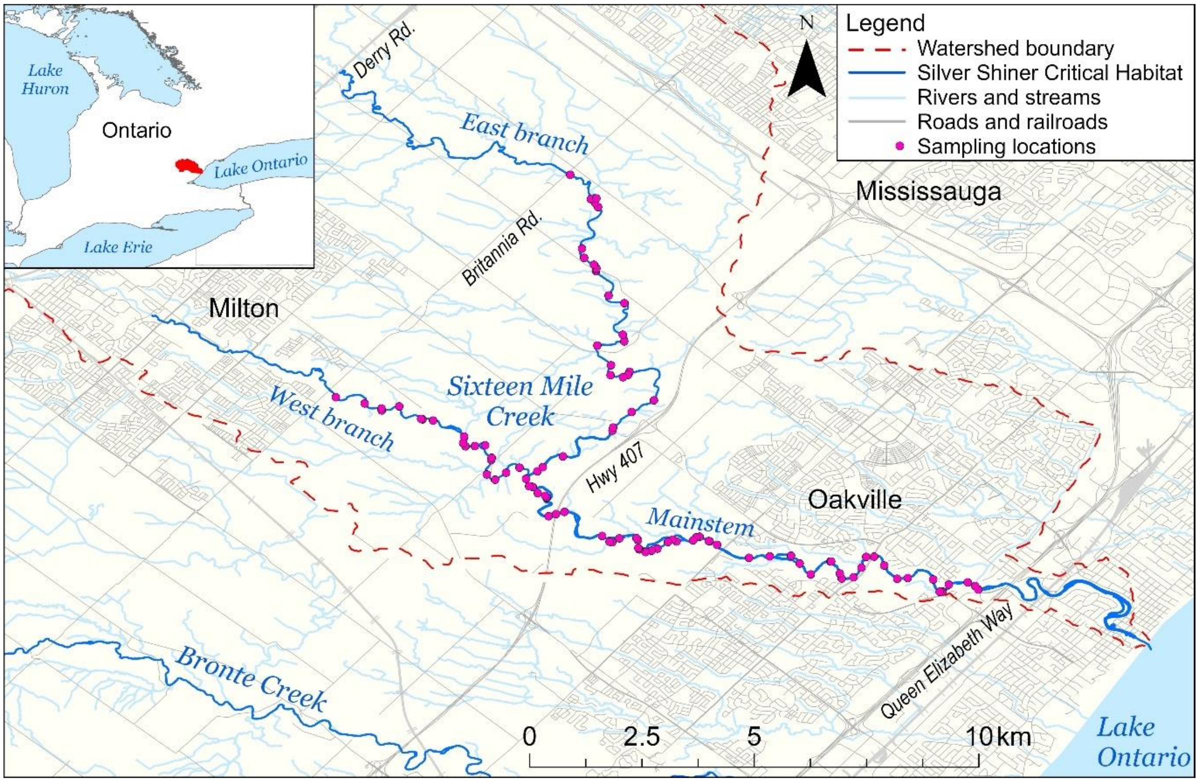

Figure 1. *Locations of Silver Shiner sampling (pink points) conducted in the Sixteen Mile Creek watershed (red dashed line) in southern Ontario, Canada (2022–2023). Dark blue segments indicate areas within which Silver Shiner critical habitat is found (Fisheries and Oceans Canada 2022).*

FISH AND HABITAT SAMPLING

Methods for fish and habitat sampling in Sixteen Mile Creek were previously described in Lopez et al. (2024) and Lamothe et al. (2025). In 2022, the location and depth of each riffle, run, and pool habitat were measured within the study area. Site selection for sampling fish and habitat parameters was subsequently performed using a depth-stratified approach that was based on the probabilistic relationship between adult Silver Shiner occupancy probability and mean site depth in runs or pools (Lamothe and Drake 2022). Riffle habitats were excluded as the adult life stage spends minimal time in these habitats. A total of 101 runs or pools (i.e., sites) were surveyed in 2022 between September 21 and October 21. Due to time constraints, 98 of 101 sites were resampled in 2023 (October 10 to November 8) and one new run site was added (Lamothe et al. 2025). At each site, three replicate seine hauls were performed. After each haul, individual Silver Shiner were removed from the site and placed in temporary bins until all hauls at a site were complete (i.e., depletion). Only adult individuals (> 80 mm; Burbank et al. 2021) were counted to align with criteria used for listing recommendations (COSEWIC 2021). Fish were released back to the creek only once enumeration of individuals from all three hauls was complete.

Sampling was conducted under SARA Permit Number 22-PCAA-00061 in 2022 and 23-PCAA-00048 in 2023. Live capture seine netting was conducted under Standard Operating Protocol GWACC-116, approved by Fisheries and Oceans Canada and Environment and Climate Change Canada Animal Care Committee (operated under the approval of the Canadian Council on Animal Care). In addition to fish sampling, stream width (m) was recorded at the center of each site and stream depth (m) was recorded at the upstream, downstream, and middle point of the site and averaged in both years.

SPATIAL PATTERNS

Moran's I statistic was used to quantify spatial autocorrelation in observed site-level abundance and a permutation test was applied to assess whether spatial clustering differed significantly between 2022 and 2023. First, the true observed difference in Moran's I between the two years was calculated. Then, the 2022 and 2023 site abundance measurements were randomly permuted across the geographic locations of surveyed sites, Moran's I was recalculated, and the difference between the permuted 2022 and 2023 Moran's I estimates for each permutation was calculated. This permutation process was repeated 999 times, which yielded a null distribution of differences expected under the hypothesis of no true difference in spatial clustering between years. The true difference was then compared to this null distribution. A p-value was calculated as the proportion of permutations where the absolute difference in Moran's I was greater than or equal to the absolute observed difference, divided by 999. A p-value < 0.05 was interpreted as evidence of a significant difference in spatial clustering between years. All spatial analyses were performed using the 'sp' (Pebesma and Bivand 2005; Bivand et al. 2013) and 'spdep' packages (Bivand et al. 2013) in R (R Core Team 2024).

ABUNDANCE MODELS

Counts of animals in the wild almost always represent an imperfect sample, as individuals of the species can be missed despite being susceptible to the capture technique (Williams et al. 2002; Chambert et al. 2016). This reality creates a source of uncertainty when generating population abundance estimates; true abundance is likely to be greater than what was observed, assuming no false positive detections. Spatial and/or temporal replication of sampling is recommended to reduce uncertainty around the true number of individuals in the sampled region when estimating population abundance. N-mixture models were designed to use replicated count data to estimate abundance while explicitly accounting for the imperfect observation process (Royle 2004). Although the number of extensions to the original formulation continues to grow (Madsen and Royle 2023), all N-mixture models share a common hierarchical structure that jointly models the ecological process (true site-level abundance; λ_i) and the observation process (detection probability; p).

Stewart et al. (2019) developed a multispecies N-mixture model that accounts for variation in p across sites (i) and surveys (j)—a situation likely common when sampling small-bodied freshwater fishes. Following Lamothe and Drake (2025), we applied this framework for a single species, specifying p as:

$$p_1 + (p_2 - p_1)(1 - c^{j-1})$$

where p_1 is the initial detection probability during the first seine haul, p_2 is the asymptotic detection probability, and c is the rate of change in detection probability across successive seine hauls (Schnute 1983; Stewart et al. 2019). This formulation assumes a monotonic change in detection probability over repeated sampling events at an individual site. To model site-level variation in detection, we used a logit-link function:

$$logit(p_i) = \gamma_0 + \sum_{v=1}^{w} \gamma_v x_{v,i}$$

where $x_{v,i}$ are predictors $v = 1,2, \dots w$ measured at a site i, and γ represents the intercept and slope parameters. We considered mean site depth (m) and year (2022 = 0; 2023 = 1) as potential covariates given that greater depth may reduce seine efficiency and previous research showed interannual differences in species detection probability between 2022 and 2023 (Lamothe et al. 2025). Note that in this study, p represents the probability of detecting an individual of the species given the individual is present and available to be captured at a site, whereas for the occupancy models presented in Lamothe et al. (2025), detection probability (also denoted as p) represents the probability of detecting the *species* given that the *species* is present and available to be captured at a site.

The latent abundance state in the N-mixture models was defined as a Poisson distribution with an added hierarchical component to account for extra site-level variation (Stewart et al. 2017, 2019). Although a negative binomial distribution was also considered to address potential overdispersion, it produced nearly identical results to the Poisson model based on the deviance information criterion (DIC) and was therefore excluded from further analysis. Site abundance, λ_i, was specified using a log-link function:

$$log(\lambda_i) = \alpha_0 + \sum_{v=1}^{w} \alpha_v x_{v,i}$$

where α represents the intercept and slope parameters, and $x_{v,i}$ are site-level covariates. Mean site depth (m), stream width (m), and year (2022 = 0; 2023 = 1) were considered as potential covariates. Previous research has demonstrated a positive relationship between Silver Shiner occupancy probability and mean site depth (Lamothe and Drake 2022), and stream width was included under the hypothesis that sites sampled in wider areas of the river may support more fish. Site depth and width were scaled and centered (i.e., mean = 0, standard deviation (SD) = 1) prior to modelling to allow comparisons of covariate effects.

The strength of evidence for covariate effects on p_i and λ_i was evaluated by fitting a fully parameterized model, which included all covariates alongside corresponding inclusion parameters. These inclusion parameters were defined for each covariate x_v as latent binary variables such that the prior probability that variable v was included in the model was 0.5 (Stewart et al. 2019). Therefore, when the posterior probability for the inclusion variable on x_v = 0, variable v has zero effect. Alternatively, if the posterior probability for the inclusion variable on x_v = 1, then v has a linear effect with a high degree of support. Only covariates with a posterior inclusion probability > 0.70 were retained in the final model (Stewart et al. 2019).

Models were implemented using the ‘rjags’ package with the ‘jagsUI’ interface in R (Kellner 2024; Plummer 2024; R Core Team 2024). Non-informative normal priors (mean = 0; SD = 0.01) were used for regression coefficients and a uniform prior (range: 0.01–100) was applied to the overdispersion parameter. Each model was run using 500,000 iterations across three chains, with the first half discarded as burn-ins and a thinning interval of 20. Traceplots were used to diagnose sufficient mixing and the Gelman-Rubin diagnostic statistic ($\hat{R}$) was used to assess chain convergence (Brooks and Gelman 1998).

POPULATION ABUNDANCE ESTIMATES

Population abundance (N) estimates for the surveyed area of Sixteen Mile Creek were derived by combining posterior estimates of site-level abundance from the final N-mixture model with estimates of total available habitat for 2022 and 2023 (*sensu* Lamothe and Drake 2025). In 2022, field crews walked the entirety of Sixteen Mile Creek from Britannia Road to the QEW (Figure 1) and recorded the depth (m) and coordinates of all riffles, runs, and pools. A total of 112 runs and 87 pools were identified, distributed across the east branch (49 runs or pools), west branch (36), and main channel (114). Approximately half of the identified pools and runs were sampled each year. More information about locations of individual pools and runs can be found in Lopez et al. (2024).

To estimate total population size, the posterior samples of site-level abundance (λ_i) were summed across all surveyed sites for each MCMC iteration, generating a posterior distribution of total estimated abundance within the sampled area per year. Then, each draw was scaled by a factor of two to account for the fact that approximately half of the available pools and runs were surveyed in each year. This simple approach allowed uncertainty to be propagated from the model through to the final population abundance (N) estimates, resulting in a posterior distribution of N that reflects both detection uncertainty and sampling coverage. From this distribution, the posterior mean and 95% credible intervals (CI) are reported as the annual population abundance estimates. This approach to extrapolation assumes that unsampled pools and runs are equally suitable for Silver Shiner relative to the sampled sites within Sixteen Mile Creek.

K-FOLD CROSS VALIDATION

K-fold cross validation (CV) was used to evaluate how sampling fewer sites could influence the observed year effect on detection probability and site abundance (λ_i), and population abundance (N) estimates. To do this, k = 10, 20, or 30 observations from the observed 2022 and 2023 data were randomly removed, the most well-supported N-mixture model was re-run, annual site abundance estimates across 199 runs or pools were extrapolated, and the results were compared to the models built using the full data set. This process was only repeated 100 times for each level of sample size reduction (k) due to computational limitations. Cross-validated estimates of N are denoted with k and CV as a superscript and subscript, respectively. For example, $N_{CV}^{k=10}$ is a cross-validated estimate of abundance when k = 10 and $\bar{N}_{CV}^{k=30}$ is the mean N across 100 cross-validation iterations when k = 30.

RESULTS

PATTERNS IN OBSERVED COUNTS

In 2022, 787 adult Silver Shiner were captured, with individuals detected at 75 of the 101 surveyed sites (Figure 2). Mean observed counts of adult Silver Shiner at sites in 2022 was 7.79 ± 12.37 SD, ranging from 0 to 63 individuals. A total of 1,185 adult Silver Shiner were captured in 2023 across 47 of 99 surveyed sites (Figure 2), with a higher mean observed count of 11.94 ± 32.88 SD (range = 0–215). In both 2022 and 2023, approximately 21% of surveyed sites had 10 or more adults detected. Across years, mean observed adult counts were significantly greater in pools (14.48 ± 34.00 SD) than in runs (6.63 ± 14.75 SD; $F_{1,198}$ = 4.95, p = 0.027). There was no significant difference in mean site depth between 2022 (0.62 m ± 0.18 SD) and 2023 (0.59 m ± 0.19 SD; $F_{1,198}$ = 0.939, p = 0.334) across pools and runs.

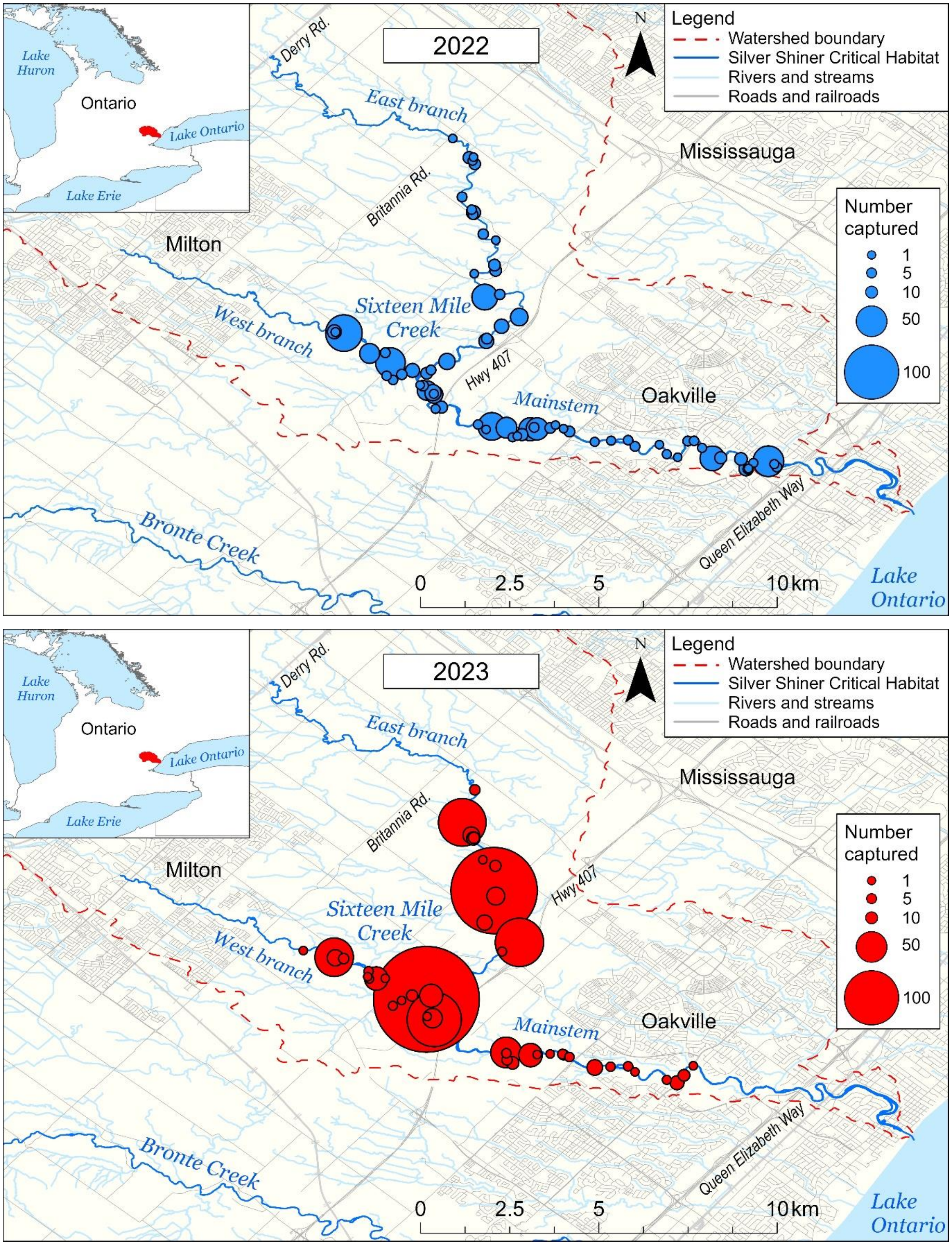

Figure 2. *Number of adult Silver Shiner detected at sites surveyed in 2022 (blue; top) and 2023 (red; bottom) in Sixteen Mile Creek. Point size is scaled by the number of individuals captured and scales are equal between the two maps.*

Significant spatial autocorrelation in the number of observed adult Silver Shiner per site was detected in 2023 (Moran's I = 0.18, p = 0.046) but not in 2022 (Moran's I = -0.08, p = 0.705), and a permutation test indicated that there was a marginally significant difference in Moran's I between years (p = 0.090). Overall, 47.9% (n = 377) of adult Silver Shiner were captured in the east or west branches in 2022, compared to 64.5% (n = 762) in 2023 (Figure 2). The most adult Silver Shiner captured at a site in the main channel was 50 in 2022 and 172 in 2023.

ABUNDANCE ESTIMATES

Traceplots and $\hat{R}$ (range: 1.00–1.06) suggested sufficient mixing and convergence of the final N-mixture model, respectively. The final model included year ($Year$ = 0 reflects 2022 and $Year$ = 1 reflects 2023) as a significant covariate for both the detection (p) and site abundance (λ_i) components:

$$logit(p) = -0.800 * Year - 1.328$$

$$log(\lambda_i) = 1.378 * Year + 2.932$$

where the 95% credible intervals (CIs) for the year effect ranged between -1.278 and -0.337 for detection and between 0.618 and 2.188 for abundance. Detection probability was significantly greater in 2022 than 2023; mean detection probability in the first seine haul equaled 0.210 (95% CI: 0.167–0.257) in 2022 and 0.108 (95% CI: 0.074–0.147) in 2023. Detection probability increased across subsequent seine hauls at 79% and 84% of the sites surveyed in 2022 and 2023, respectively.

The mean and variance of the posterior λ_i estimates were significantly greater in 2023 ($\overline{\lambda_i}$ = 74.01 adults; 95% CI: 0–866) than 2022 ($\overline{\lambda_i}$ = 17.97 adults; 95% CI: 0–147), but 46.4% of λ_i estimates equalled zero in 2023 compared to 23.7% in 2022. Mean site abundance estimates were greater than 500 adults at five sites in 2023, with the highest $\bar{\lambda}_i$ estimate in 2022 being 194 (main branch) and 1,557 in 2023 (main branch; Figure 3). Across years, λ_i was less than 198 adults across 95% of the posterior λ_i estimates. There was particularly high uncertainty in λ_i estimates at sites with high catch numbers that were surveyed in 2023 (Figure 3). Extrapolating λ_i estimates from the top model to the entirety of the sampling frame using the existing habitat survey led to an estimated mean adult population abundance ($\overline{N}$) of 3,631 (95% CI: 3,033–4,413) in 2022 and 14,481 (95% CI: 10,481–20,570) in 2023 (Figure 4).

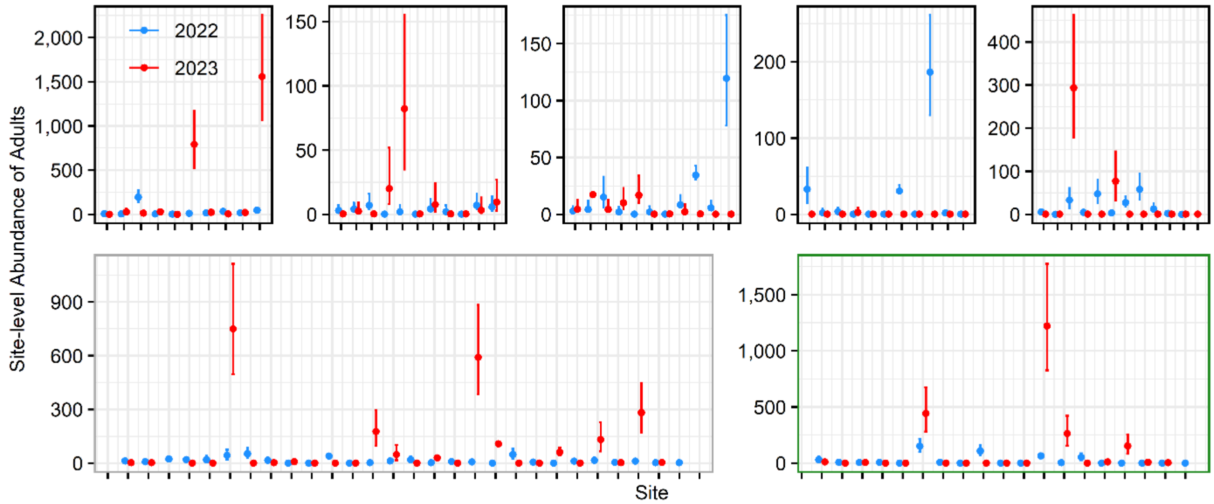

Figure 3. *Estimated site-level abundance of adult Silver Shiner in 2022 (blue) and 2023 (red). Border colour denotes river section: main channel (black), east branch (grey), and west branch (green). Sites in the main branch are split into panels with differing y-axis scales to improve visualization of within- and between-site variation. Circles indicate model-derived mean estimates and error bars indicate 95% credible intervals.*

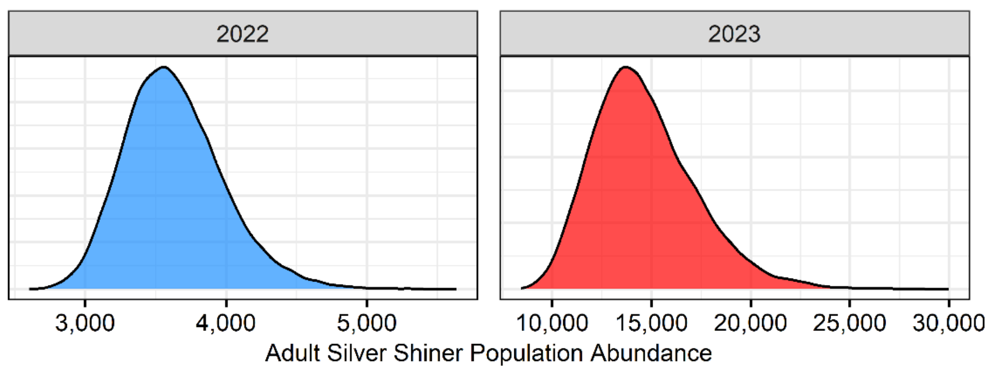

Figure 4. *Estimated adult Silver Shiner population abundance based on site-level abundance estimates from the top N-mixture model and estimates of total habitat availability. Note the different in x-axis scale.*

K-FOLD CROSS VALIDATION

K-fold cross-validation indicated increased uncertainty in N if fewer sites were sampled irrespective of year (Figure 5). Generally, 95% CIs of N_{CV} were wider than N (i.e., estimates from the full model). In 2022, the 95% CI width of N = 1,380 was smaller when compared to $\bar{N}_{CV}^{k}$ = 1,406, 1,500, and 1,577 when k = 10, 20, and 30, respectively. Similarly, the 95% CI width of N in 2023 = 10,089 was smaller compared to $\bar{N}_{CV}^{k}$ = 11,104, 12,074, and 11,586 when k = 10, 20, and 30, respectively. In addition, the mean difference between N_{CV} estimates and N estimates increased with k. At k = 10, the $\bar{N}_{CV}$ estimates in 2022 were outside the 95% CI of N nine times, with each of those $\bar{N}_{CV}$ estimates being less than the lower 95% CI threshold of N (3,033; Table 1). In comparison, $\bar{N}_{CV}^{k=30}$ in 2023 was less than the lower 95% CI of N (10,481) in 27 iterations and greater than the upper 95% CI (20,570) in 18 iterations (Table 1).

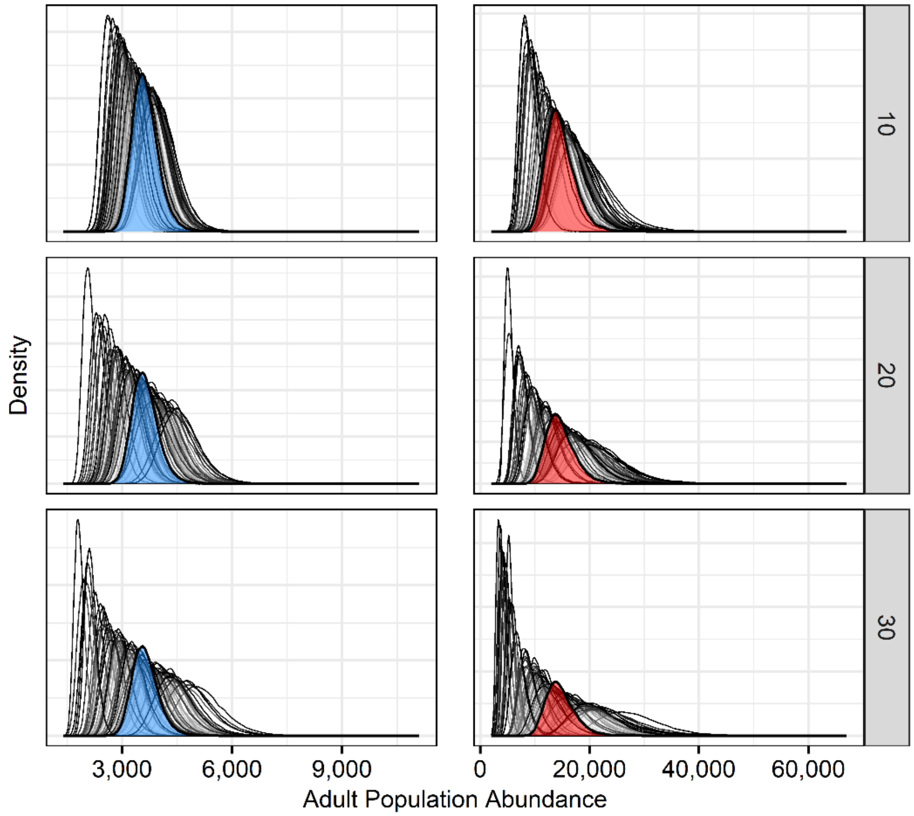

Figure 5. *Results of k-fold cross validation for adult Silver Shiner population abundance estimates for 2022 (left) and 2023 (right). White transparent distributions represent population abundance estimates when k = 10, 20, or 30 observations are removed (rows). Blue and red distributions are the true abundance estimates for 2022 and 2023, respectively, based on the full model. Note the difference in x-axis scale between 2022 and 2023.*

Table 1. *Percent of iterations from k-fold cross validation where the mean annual population abundance estimates ($\bar{N}_{CV}$) were less than or greater than the mean population abundance estimates from the full model ($\bar{N}$) and outside the 95% credible intervals (CI) of the full model. 2022: $\bar{N}$ = 3,631 (95% CI: 3,032–4,413). 2023: $\bar{N}$ = 14,481 (95% CI: 10,481–20,570).*

Metric	Year	k = 10	k = 20	k = 30
$\bar{N}_{CV} < \bar{N}$	2022	52%	51%	56%
$\bar{N}_{CV} > \bar{N}$	2022	48%	49%	44%
$\bar{N}_{CV} < N$ 95% CI	2022	9%	17%	26%
$\bar{N}_{CV} > N$ 95% CI	2022	0%	7%	14%
$\bar{N}_{CV} < \bar{N}$	2023	34%	40%	56%
$\bar{N}_{CV} > \bar{N}$	2023	66%	60%	44%
$\bar{N}_{CV} < N$ 95% CI	2023	9%	15%	27%
$\bar{N}_{CV} > N$ 95% CI	2023	1%	8%	18%

As k increased, there was a greater probability that the 95% CI for the year effect on both site abundance (λ_i) and detection (p) would overlap zero (Figure 6; Table 2), indicating no annual differences. The upper 95% CI of the year effect on p overlapped zero for 11, 17, and 33 iterations when k = 10, 20, and 30, respectively (Table 2). A significant *positive* effect of year on p was observed in three iterations at k = 30, opposing the direction observed in the final model (Figure 5; Table 2). In contrast, the lower 95% CI of the year effect never overlapped zero for λ_i across all k scenarios (Table 2). However, the significance of the year effect on λ_i was lost in seven and 19 iterations when k = 20 and 30, respectively (Table 2).

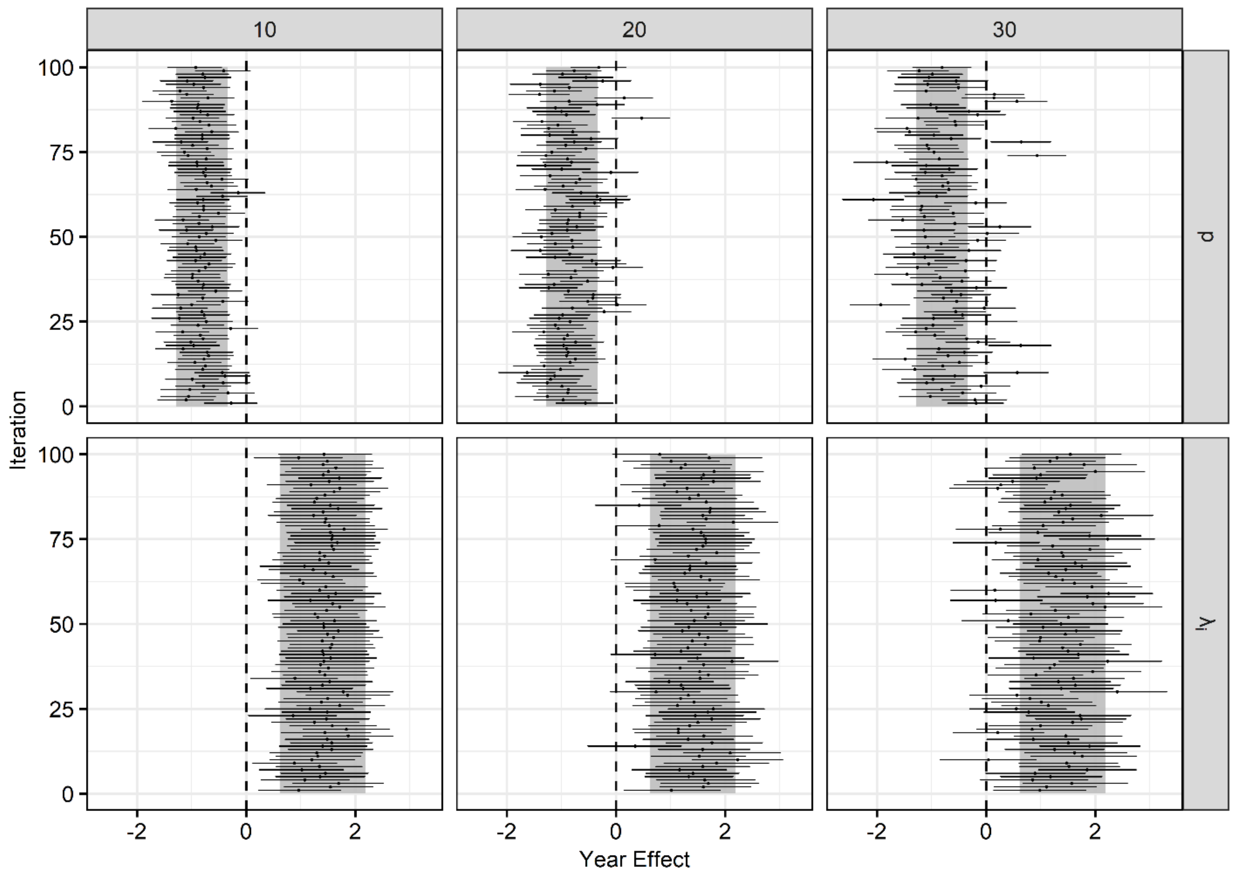

Figure 6. *K-fold cross validation results of the year effect (x-axis) on detection probability (p; top row) and site abundance (λ_i; bottom row) across 100 iterations (y-axis) when k = 10, 20, or 30 (columns). Points and error bars indicate mean and 95% credible intervals for each iteration. The grey shaded region represents the 95% CI of year effects from the full model.*

Table 2. *Percent of iterations from k-fold cross validation when the mean and 95% credible intervals (CI) of the year effect were either greater than or less than zero for detection probability (p) and site abundance (λ_i), respectively.*

Level	Measure	k = 10	k = 20	k = 30
p	Lower CI > 0	0%	0%	3%
p	Mean > 0	0%	3%	10%
p	Upper CI > 0	11%	17%	33%
λ_i	Lower CI < 0	0%	7%	19%
λ_i	Mean < 0	0%	0%	0%
λ_i	Upper CI < 0	0%	0%	0%

DISCUSSION

Abundance estimates are lacking or out of date for many populations of SARA-listed freshwater fishes. Here, new population abundance estimates are presented for adult Silver Shiner in Sixteen Mile Creek, Ontario. Results indicated an inverse relationship between species abundance and occurrence over the two years; adult Silver Shiner were more widely distributed across sampled sites in 2022 than in 2023, but mean site abundance was significantly greater at sampled sites in 2023 than in 2022. The mean adult population abundance estimate for 2023 was more than four times greater than that for 2022; however, the uncertainty around the 2023 estimate, as reflected by its 95% credible interval, was approximately 10 times greater than the 2022 estimate. Data limitations restrict causal explanations for the annual differences in the abundance estimates, but were likely in part the result of differences in timing of annual surveys and differences in abiotic and biotic conditions. Continued monitoring across seasons and over time is needed to improve knowledge of natural variation in Silver Shiner occurrence and abundance.

Imperfect detection led to significant uncertainty in the site-level abundance estimates, particularly at sites where many individuals were captured and in 2023 when detection probability was reduced. Imperfect detection is a common challenge when sampling small-bodied imperilled riverine fishes (Dextrase et al. 2014a, 2014b). Previous research has demonstrated that detection probability of freshwater fishes can vary by gear type (Haynes et al. 2013; Smith et al. 2015) and as a result of seasonal behaviours (Larocque et al. 2020) and habitat characteristics (Dextrase et al. 2014b). In this study, there were no covariates determined to be significant predictors of Silver Shiner detection probability besides the year effect. The year effect suggests that there were some unmeasured factors influencing the detection and capture rate of adult Silver Shiner between September–October 2022 and October–November 2023. Species behaviours such as feeding and migration are often driven by seasonal factors (e.g., temperature), and the later sampling in 2023 may have coincided with distinct behaviours not expressed during the time of sampling in 2022. For instance, video footage from the Grand River, Ontario, suggests that Silver Shiner undergoes within-stream migration to over-wintering habitat in early fall (late September–October; Bunt 2016). During this period, increased movement may reduce detection probability as fish become more mobile and potentially less prone to capture.

Spatial autocorrelation was detected in the number of adult Silver Shiner captured at a site in 2023 but not in 2022. This pattern suggests that nearer sites exhibited similar capture rates and thus there was a lack of statistical independence among observations. This spatial dependence can lead to biased model estimates and inflate confidence in predictions if not accounted for. To address this, a model incorporating a spatial random effect was developed, but due to poor convergence and mean predictions that closely resembled those of the non-spatial model, it was not included in this study. Generally, the inclusion of spatial random effects tends to increase uncertainty in site-level predictions (Dormann et al. 2007; Guélat and Kéry 2018). Hence, the uncertainty associated with the predictions presented in this study may be underestimated, but due to the way the results were extrapolated, the increased uncertainly likely had minimal influence on population abundance estimates.

There are several considerations about the sampling design and assumptions of the estimation process that must be considered when interpreting the population abundance estimates. First, the adult population abundance estimates reported in this study only include pools and runs between the QEW and Britannia Road. These estimates therefore exclude riffle habitats and the unsampled upstream and downstream areas containing critical habitat (Figure 1). Capture records of Silver Shiner upstream and downstream of the sampled area exist (Fisheries and Oceans Canada 2022), but there is uncertainty about the quality and quantity of habitat in those sections of the river, and therefore we did not extrapolate model results to these areas. Moreover, while Silver Shiner must inevitably swim through riffles to locate available pool and run habitats, the adult life stage spends minimal time in riffles and therefore riffles were not considered when extrapolating site abundance estimates to the population.

Minimum viable population size (MVP) and population abundance were previously estimated for Silver Shiner in Sixteen Mile Creek (Young and Koops 2013). MVP is defined as the minimum population size that has a high probability of persistence (e.g., 99%) over a period of time (e.g., 100 years) despite the ongoing effects of demographic and environmental stochasticity. MVP was estimated as 25,984 and 779,754 age 1+ individuals at 1% probability of extinction over 100 years assuming an extinction threshold of two adults and generational catastrophe rate of 5% and 10%, respectively (Young and Koops 2013). The 95% CI of the 2022 population abundance posterior distribution was entirely below the estimated $MVP_{99\%}$ regardless of the rate of catastrophe, and < 1% of abundance estimates from 2023 were > 25,984 adults. In addition to MVP, Young and Koops (2013) estimated adult population abundance in Sixteen Mile Creek, which was based on data collected at eight locations in 2011 where each location consisted of a single riffle, run, and pool. Population abundance was estimated by first summing the abundance of adult Silver Shiner captured in the riffle, run, and pool habitats at each location, dividing the sum by location area (300 m^2) to generate a location-specific density, and then averaging those densities across locations. The mean density estimate (0.1775 fish/m^2) was multiplied by the mean stream width measured at the surveyed locations (17.8 m) and the length of river known to support the species (10 km), resulting in a population abundance estimate of 31,595 adults (95% confidence interval: 18,328–54,462). They further extrapolated the results to include areas of known *and suspected* available habitat (32 km), resulting in an estimated population abundance of 101,104 adults (95% confidence interval: 58,650–174,279).

Differences in the data used and analytical approach implemented make comparisons of population abundance estimates between studies challenging. Nevertheless, the historical mean density estimate made across riffle, run, and pool habitats in Young and Koops (2013) was greater than what was observed in 2022 (~0.08 fish/m^2) and 2023 (~0.12 fish/m^2) in pools and runs, but less than model-derived mean estimates (2022: 0.18 fish/m^2; 2023: 0.74 fish/m^2). Furthermore, the mean adult population abundance estimate in known available habitat from Young and Koops (2013) was nearly 10 times the 2022 estimate presented in this study and more than double the 2023 estimate. As a comparison to Young and Koops (2013), if it is assumed that the mean width of Sixteen Mile Creek is 17.8 m, that there is 10 km of occupied habitat, and that adult Silver Shiner are present at a density of 0.18 fish/m^2 in 2022 and 0.74 fish/m^2 in 2023, then comparative estimates of adult population abundance would be 32,040 in 2022 (17.8 m x 10,000 m x 0.18 fish/m^2) and 131,720 in 2023 (17.8 x 10,000 x 0.74).

There were few surveyed sites that consistently had high abundance of adult Silver Shiner between years, indicating that adults actively move within the system. Such movement is likely driven by a combination of ecological factors such as the search for optimal foraging conditions and resources, avoiding unsuitable habitats, or avoiding predation or competition with co-occurring species (Huntingford 1993). Due to the lack of dedicated mark-recapture or telemetry studies, knowledge of the magnitude, direction, and frequency of movements by Silver Shiner remains unknown. However, group foraging behaviour has been observed for Silver Shiner, with aggregations ranging from small groups of five individuals to large schools exceeding 200, and the species has been observed actively avoiding areas dominated by predatory species (e.g., Smallmouth Bass *Micropterus dolomieu*; Bunt 2016). Ultimately, quantifying within-system movement for Silver Shiner could help better understand the differences in distribution and abundance observed across sampling efforts in 2022 and 2023.

Overall, this study provides updated Silver Shiner population abundance estimates in Sixteen Mile Creek for 2022 and 2023. Sampling conducted during both years indicated that the species is broadly distributed throughout the sampled area of Sixteen Mile Creek, and that its abundance and distribution can vary in space and time. N-mixture models identified imperfect detection of Silver Shiner with differing effects per year. The uncertainty in site-level abundance estimates resulting from imperfect detection was carried through to the overall population abundance estimates, most notably in 2023, which were reported as means and 95% credible intervals. Results of the k-fold cross-validation demonstrated the importance of maintaining the current effort levels to evaluate trends in Silver Shiner abundance over time. Ongoing monitoring of this population will be essential for understanding temporal fluctuations in abundance. To improve future analyses of abundance trends, timing of sampling should be standardized by date or thermal conditions of the river.

ACKNOWLEDGEMENTS

The authors thank Jason Barnucz, Amy Boyko, Kayla Bruce, Julia Colm, Andrea Dunn, Robin Gáspárdy, Mandy Gibson, Lianna Lopez, Lenka Trivett, and Carly White for collecting and processing the data used in this study, and Matt Guzzo for reviewing a prior version of the report. Funding was provided by Fisheries and Oceans Canada's Species at Risk Program.

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