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# Recovery Potential Modelling of Lake Sturgeon (Acipenser fulvescens) in Canada 

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## Foreword

This series documents the scientific basis for the evaluation of aquatic resources and ecosystems in Canada. As such, it addresses the issues of the day in the time frames required and the documents it contains are not intended as definitive statements on the subjects addressed but rather as progress reports on ongoing investigations.

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#### Abstract

The Committee on the Status of Endangered Wildlife in Canada (COSEWIC) has assessed the Lake Sturgeon (Acipenser fulvescens) across 4 designatable units (DU) as Endangered in DUs 1 and 2, Threatened in DU 4 and Special Concern in DU 3. Here population modelling, using a female only matrix model, is presented to determine population-based recovery targets, assess the impact of anthropogenic harm, and conduct long-term projections of population recovery in support of a recovery potential assessment (RPA) for Lake Sturgeon populations in Canada. Simulations were conducted for populations with 5 distinct growth patterns characterized by their rate of growth and maximum size to incorporate the diversity in life-history characteristics expected across Canada. Populations with slower growth were estimated to have greater adult survival rates ( $\sim 95 \%$ ) and resultantly greater generation times (> 50 years) relative to faster growth sturgeon populations ( $\sim 92 \%$ and $\sim 30$ years). Patterns in elasticity estimates were similar across growth types. Under most scenarios population growth rate ( $\lambda$ ) was most sensitive to changes to survival of young adults (age 26-62). The exception to this was populations experiencing continued reduced adult survival (leading to population decline); which were most sensitive to changes to the survival rate of older juveniles (age 13-25). This indicates the importance of understanding the current stresses and the age distribution of a population when considering harm or determining what recovery actions to take. To achieve demographic sustainability, (i.e., a self-sustaining population over the long term) under conditions with a catastrophe probability of $0.15 /$ generation and a quasi-extinction threshold of 25 adult females at a $1 \%$ probability of extinction over 250 years, population sizes ranging from 1,255-4,840 adult females were required. Populations with faster growth required larger population sizes $(\sim 5,000)$ to achieve demographic sustainability as their lower generation times resulted in increased frequency of catastrophic population decline (a $50 \%$ decline every 200-250 years compared to 340-360 years for slow growth Lake Sturgeon). MVP population size required between $550->13,000$ ha of lake habitat and 162-3,800 ha of river habitat (inclusive of both sexes and all age classes). Recovery times are variable, dependent on initial population size and the rate of population growth.


## INTRODUCTION

The status of Lake Sturgeon (Acipenser fulvescens) in Canada was previously assessed by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) in 2006. At the time, Lake Sturgeon populations were divided into 8 designatable units (DUs). Five DUs were assessed as Endangered, one as Threatened and two as Special Concern (COSEWIC 2006). Subsequent research has resulted in DUs 2-5 being reclassified as a single DU with Lake Sturgeon in Canada now considered to consist of 4 DUs delineated by the national freshwater biogeographic zones: DU 1 - Western Hudson Bay; DU 2 - Saskatchewan-Nelson River; DU 3 Southern Hudson Bay-James Bay; DU 4 - Great Lakes-Upper St. Lawrence. After reassessment of Lake Sturgeon population status in 2017, COSEWIC assessed populations in DUs 1 and 2 as Endangered, DU 3 as Special Concern, and DU 4 as Threatened (COSEWIC 2017).

In accordance with the Species at Risk Act (SARA), which mandates the development of strategies for the protection and recovery of species that are at risk of extinction or extirpation from Canada, Fisheries and Oceans Canada (DFO) has developed the recovery potential assessment (RPA; DFO 2007a, b) as a means of providing information and scientific advice. There are three components to each RPA - an assessment of species status, the scope for recovery, and scenarios for mitigation and alternatives to activities - that are further broken down into 22 elements. This report contributes to components two and three and elements 3 , $12,13,14,15,19,20,21$, and 22 by identifying recovery targets, assessing the impact of anthropogenic harm, projecting recovery timeframes and identifying mitigation strategies with associated uncertainty for Canadian populations of Lake Sturgeon.
A previous RPA model was conducted for Lake Sturgeon in 2008 following its assessment by COSEWIC (Vélez-Espino and Koops 2008). The previous analysis estimated abundance recovery targets with use of a predictive relationship (Reed et al. 2003b), based on maximum population growth rate, which gave an estimate of the number of spawning females required to provide a $99 \%$ probability of persistence over 40 generations ( $\sim 1,164-1,616$ years). This produced an average minimum viable population (MVP) size of 1,188 annually spawning females. Harm analysis revealed that the young adult stage ( $\sim$ age 14-30) was the most vulnerable to harm with the population becoming vulnerable to decline with an increase in mortality rate of only $3 \%$.
This document provides an update to the previous RPA utilizing new methodological approaches to estimating recovery targets and allowable harm, as well, this analysis incorporates new data collected since the previous assessment. This work is based on a demographic approach developed by Vélez-Espino and Koops (2009, 2012) and Vélez-Espino et al. (2010) which determines a population-based recovery target based on long-term population projections.

## METHODS

The analysis consisted of five parts:

1. information on vital rates was compiled to build projection matrices incorporating variability within stochastic simulations.

With these projection matrices:
2. stochastic sensitivity of the population growth rate to perturbations of vital rates was determined and used to estimate recovery efforts and the impact chronic anthropogenic harm following Vélez-Espino and Koops (2009);
3. simulation analyses were used to estimate the impact of chronic (permanent) and transient harm (a one-time removal of fish of various life stages) on population growth;
4. population viability analysis (PVA) was conducted to estimate the minimum viable population (MVP) and the minimum area for population viability (MAPV; i.e., the amount of suitable habitat required to support the MVP); and,
5. using MVP as a recovery target, simulations were conducted to estimate the probable time to recovery under various recovery scenarios.

## SOURCES

Lake Sturgeon is a well-studied species in Canada. Data and relationships from the primary literature were used to inform estimates of growth, survival, reproduction and abundance. All analyses and simulations were conducted using the statistical program R 3.3.2 (R Core Team 2016).

## THE MODEL

Lake Sturgeon life cycle was modeled using a female only, birth-pulse, post-breeding, stage-structured matrix model with annual projection intervals and five life stages (Caswell 2001). The five life stages represented: young-of-the-year (YOY) fish (age 0); young juveniles (age 1-12); old juveniles (age 13-25); young adults (age 26-62); and old adults (age 63-100; Figure 1). Isolating the YOY stage allows for incorporation of a type III survivorship curve with high mortality rates early in life. Juvenile and adult life stages were divided into "young" and "old" sub-stages because of their protracted length and to allow for inclusion of greater variability in mortality and fertility rates. The juvenile stage was divided at the median between age 1 and age-at- $50 \%$-maturity (age $26 ; t_{\text {mat }}$ ) and the adult stage was divided at the median between age-at-maturity and maximum longevity (age 100; $t_{\max }$ ). The duration of stages impacts model output such as stage-specific elasticity estimates and the stable-stage distribution. Therefore it is necessary that stages have some biological or management relevance and their definitions must be kept in mind when interpreting results. Use of a stage-structured matrix model was necessary for Lake Sturgeon because of the increased computer time required to execute stochastic simulations on large matrices. Use of an agestructured matrix would have likely provided limited improvement to simulation results (VélezEspino et al. 2006, Vélez-Espino and Koops 2009). As a female only model all model outputs and population estimates, unless otherwise stated, represent only the female portion of the population. A female only model assumes that breeding success is independent of the number of males present in the population (Caswell 2001).
Matrix population models use estimates of vital rates (growth, survival, and fecundity) to project age- or stage-specific population sizes. The dominate eigenvalue of the matrix represents the population growth rate $(\lambda)$ and indicates the long term status of the population based on current conditions (Caswell 2001). A $\lambda>1$ indicates that the population is growing exponentially, a $\lambda=1$ indicates a population that is stable, and a $\lambda<1$ indicates a population that is declining towards 0 . The dominant right eigenvector of the matrix represents the stable stage structure of the population and indicates the proportional distribution of individuals among stages/ages. This can be used to estimate the number of individuals in all other stages/ages if one is known.

Elements incorporated into a stage-structured matrix include $P_{i}$, the probability of surviving and remaining in stage $i, G_{i}$, the probability of surviving and moving to stage $i+1$, and $F_{i}$, the fertility rate or the mean number of female offspring produced annually per individual in stage i. $P_{i}$ and $\mathcal{G}_{i}$ are calculated from the annual survival probability, $\sigma_{i}$, and the transition probability, $\gamma_{i}$, which quantifies the probability of moving from one stage to the next, with $P_{i}=\sigma_{i}\left(1-\gamma_{i}\right)$ and $G_{i}=$ $\sigma_{i} \gamma_{i}$. The transition probability, $\gamma_{i}$, was estimated assuming a fixed stage duration, $D_{i}$, and the probability of graduating depending on the mean age-distribution within each stage, such that (Caswell 2001):

$$
\begin{equation*}
\gamma_{i}=\frac{\left(\sigma_{i} / \lambda\right)^{D_{i}}-\left(\sigma_{i} / \lambda\right)^{D_{i}-1}}{\left(\sigma_{i} / \lambda\right)^{D_{i}}-1} \tag{1}
\end{equation*}
$$

Use of this formula implies that stages, as included in the matrix, are simply groups of age classes rather than distinct and significant life stages (Caswell 2001). This definition is appropriate for how stages were defined within the Lake Sturgeon matrix.
The fertility coefficient incorporates: mean stage-specific fecundity $\left(f_{i}\right)$; the proportion of the offspring that are female ( $\varphi$ ); proportion of individuals in stage $i$ that are mature ( $\rho_{i}$ ); and, the spawning periodicity $(T)$ or the number of years between spawning events. As a post breeding matrix structure is being used the fertility coefficient of stage $i$ must also incorporate the probability of surviving and remaining in stage $i\left(P_{i}\right)$, as well as, the eggs produced by individuals that survive and move on to stage $i+1$. Stage specific fertility is calculated as:

$$
\begin{equation*}
F_{i}=\frac{\varphi \rho_{i} f_{i}}{T} \sigma_{i}\left(1-\gamma_{i}\right)+\frac{\varphi \rho_{i+1} f_{i+1}}{T} \sigma_{i} \gamma_{i} \tag{2}
\end{equation*}
$$



$$
\begin{gathered}
\mathbf{A}=\left[\begin{array}{ccccc}
0 & 0 & F_{j} & F_{a 1} & F_{a 2} \\
G_{Y O Y} & P_{j 1} & 0 & 0 & 0 \\
0 & G_{j 1} & P_{j 2} & 0 & 0 \\
0 & 0 & G_{j 2} & P_{a 1} & 0 \\
0 & 0 & 0 & G_{a 1} & P_{a 2}
\end{array}\right] \\
\mathbf{A}=\left[\begin{array}{ccccc}
0 & 0 & \frac{\varphi \rho_{j 2} f_{j 2}}{T} \sigma_{j 2}\left(1-\gamma_{j 2}\right)+\frac{\varphi \rho_{a 1} f_{a 1}}{T} \sigma_{j 2} \gamma_{j 2} & \frac{\varphi \rho_{a 1} f_{a 1}}{T} \sigma_{a 1}\left(1-\gamma_{a 1}\right)+\frac{\varphi \rho_{a 2} f_{a 2}}{T} \sigma_{a 1} \gamma_{a 1} & \frac{\varphi \rho_{a 2} f_{a 2}}{T} \sigma_{a 2}\left(1-\gamma_{a 2}\right) \\
\sigma_{Y O Y} & \sigma_{j 1}\left(1-\gamma_{j 1}\right) & 0 & 0 & 0 \\
0 & \sigma_{j 1} \gamma_{j 1} & \sigma_{j 2}\left(1-\gamma_{j 2}\right) & 0 & 0 \\
0 & 0 & \sigma_{j 2} \gamma_{j 2} & \sigma_{a 1}\left(1-\gamma_{a 1}\right) & 0 \\
0 & 0 & 0 & \sigma_{a 1} \gamma_{a 1} & \sigma_{a 2}\left(1-\gamma_{a 2}\right)
\end{array}\right]
\end{gathered}
$$

Figure 1. Generalized life cycle and matrix structure used to model the population dynamics of Lake Sturgeon. $F_{i}$ represents stage-specific annual fertility, $P_{i}$ represents the probability of surviving and remaining in stage $i$, and $G_{i}$ represents the probability of surviving and moving to stage $i+1$.

## PARAMETER ESTIMATES

## Growth

Lake Sturgeon growth is well described with use of the von Bertalanffy growth function (VBGF):

$$
\begin{equation*}
L_{t}=L_{\infty}\left(1-e^{-k\left(t-t_{0}\right)}\right) \tag{3}
\end{equation*}
$$

Where $L_{t}$ is total length (TL) in mm at age $t, t_{0}$ is the hypothetical age at which the fish would have had a length of $0, L_{\infty}$ is the asymptotic size, and $k$ is a growth parameter.

Growth curves for Lake Sturgeon were not available at the DU level; however size-at-age of Lake Sturgeon has previously been related to geographic location (Fortin et al. 1996, Power and McKinley 1997, Noakes et al. 1999). Fortin et al. (1996) found that length of Lake Sturgeon between ages 23 and 27 was best described as a linear function of latitude and longitude as: $T L_{23-27}=2569.6-49.1 L A T+11.4 L O N$. Power and McKinley (1997) also found a decrease in length-at-age $25\left(L_{25}\right)$ with latitude. Building on these previous relationships the data from each study was compiled, as well as, additional length-at-age 25 data not used in fitting the initial relationships and fitting a new relationship to predict $L_{25}$ (Appendix Table A1). In total the dataset had 56 observations of $L_{25} ; 18$ of which were not included in either Fortin et al. (1996) or Power and McKinley (1997). Female and male Lake Sturgeon likely grow at different rates and reach different maximum sizes (Bruch 2008); however, sex-specific data on length-at-age were not available in most instances and, as a result, data from both sexes were used to estimate growth and incorporated into our female-only population model. As well, age estimates were not adjusted for any potential biases from aging methodologies (Bruch et al. 2009). The relationship was fit using a linear model and centring latitude and longitude to reduce multicollinearity (i.e., the means were subtracted from each predictive variable; latitude: $47.91^{\circ} \mathrm{N}$, longitude: $83.59^{\circ} \mathrm{W}$ ). The best fit model, based on AIC, included latitude and longitude as well as an interaction term (Table 1, Figure 2, $\mathrm{R}^{2}=0.5$ ). Similar to previous results, length-at-age decreased greatly with latitude and increased with longitude though to a lesser extent. The interaction term indicates greater size-at-age at equivalent latitudes as longitude increases.

Table 1. Results of regression analysis predicting length-at-age-25 ( $L_{25}$ ) from centred latitude (cLat) and centred longitude (cLon).

| Coefficient | Estimate | SE | p-value |
| :--- | :--- | :--- | :--- |
| Intercept | 1138.0 | 21.97 | $<0.0001$ |
| cLat | -53.34 | 7.51 | $<0.0001$ |
| cLon | 5.10 | 2.57 | 0.052 |
| cLat:cLon | 2.24 | 0.72 | 0.0032 |



Figure 2. Lake Sturgeon length-at-age-25 data plotted against location (left panel: latitude ( ${ }^{\circ} \mathrm{N}$ ); right panel: longitude ( ${ }$ W)) by DU. The lines represent the new model fit (Table 1; solid) and previous model (Fortin et al. 1996; dashed). Lines were predicted using the median longitude ( $81^{\circ} \mathrm{W}$, left) and latitude ( $47^{\circ} \mathrm{N}$, right).

Predicted $L_{25}$ was used to solve for $L_{\infty}$ through optimization of equation 3. To do this an estimate of $k$ and $t_{0}$ was required. For $t_{0}$ a constant hatch size for Lake Sturgeon of 10.25 mm (COSEWIC 2017) was assumed. The value of $k$ can be predicted from $L_{\infty}$ (Beverton and Holt 1959). Using the compiled dataset of VBGF parameters ( $n=36$, Appendix Table A1) a relationship was fit to predict $k$. The best fit relationship incorporated $\log \left(L_{\infty}\right)$ and $\log \left(L_{25}\right)$ (Figure $3, \mathrm{R}^{2}=0.87$ ):

$$
\begin{equation*}
k=0.232-0.172 \log \left(L_{\infty}\right)+0.154 \log \left(L_{25}\right) \tag{4}
\end{equation*}
$$

Population models for Lake Sturgeon with different growth curves were created. Location across Lake Sturgeon's Canadian distribution were selected and the VBGF predicted. This was achieved by predicting length-at-age-25 (Table 1), inserting Equation 4 into Equation 3 to solve for $L_{\infty}$, and then using Equation 4 to predict $k$. The resultant growth curves (Table 2, Figure 4) differed in the rate of growth (fast/slow) and the maximum size (large/medium/small). Five distinct growth curves were retained with a model created for each for use in simulation analysis. As there was significant residual variance in the relationships used to fit the VBGFs (i.e., Figure 2) it is likely that growth at a specific location may differ from that predicted. However, with the diversity of growth curves included in the analysis it is expected that most Lake Sturgeon populations in Canada will have been captured within the range of results presented across these analyses.


Figure 3. Relationship between VBGF parameters asymptotic length ( $L_{\infty}$ ) and growth coefficient (k) for Lake Sturgeon. The solid line represents the predicted values (Equation 4) using the mean $L_{25}$ value (1,194.6 mm).

Table 2. Estimated von Bertalanffy growth function parameter values for Lake Sturgeon populations. Various location across Lake Sturgeon's Canadian range were selected to provide a diversity of growth patterns for simulations. Growth patterns are defined by relative growth rate/maximum size.

| Growth Pattern | Lat. | Lon. | $\boldsymbol{T L}_{\mathbf{2 5}}$ | $\boldsymbol{L}_{\infty}$ | $\boldsymbol{k}$ | $\boldsymbol{t}_{\boldsymbol{0}}$ |
| :--- | :--- | ---: | :---: | ---: | :---: | :---: |
| fast/large | 42.5 | -82.7 | $1,432.8$ | $1,639.4$ | 0.083 | -0.076 |
| fast/medium | 45.4 | -73.9 | $1,276.9$ | $1,437.5$ | 0.087 | -0.082 |
| fast/small | 57.3 | -102.0 | $1,117.7$ | $1,240.8$ | 0.092 | -0.090 |
| slow/large | 49.1 | -95.0 | $1,163.1$ | $1,666.0$ | 0.048 | -0.129 |
| slow/small | 50.5 | -79.1 | 950.9 | $1,419.9$ | 0.044 | -0.164 |

Multiple length-weight relationships $(n=35)$ were available for Lake Sturgeon population throughout its distribution. There was no indication of location or habitat differences in the relationships. As a result, one length-weight relationship was used and applied it to all models; however, variability in length-at-age across DUs resulted in variability in predicted mean weight-at-age (Figure 4). There is a long-recognized correlation between the slope and intercept parameters from length-weight regressions (Peters 1983). A relationship between the slope (b) and the intercept (a) was fitted: $a=451.9 b^{-22.75}$ (Figure $5, \mathrm{R}^{2}=0.99$ ) and the mean relationship estimated by taking the arithmetic mean of the slope values and predicting the intercept, giving:

$$
\begin{equation*}
W_{t}=1.32 \times 10^{-9} L_{t}{ }^{3.21} \tag{5}
\end{equation*}
$$

Where $W$ is body weight in kg and $L$ is total length in mm .


Figure 4. Predicted length-at-age (mm), weight-at-age (kg) and fecundity-at-age (eggs/female) for Lake Sturgeon populations with various growth patterns (defined in Table 2).


Figure 5. Relationship between slope (b) and intercept (a) from Lake Sturgeon length-weight relationships.

## Reproduction

Lake Sturgeon spawning typically occurs from May to late June (Scott and Crossman 1973). Lake Sturgeon are highly fecund with large females capable of depositing over 1,000,000 eggs (Scott and Crossman 1973, COSEWIC 2006). Lake Sturgeon fecundity is best predicted from weight (kg; Bruch 2008). A synthesis of Lake Sturgeon fecundity data (Harkness and Dymond 1961, Fortin et al. 1992, Bruch et al. 2006) yielded the relationship (COSEWIC 2017):

$$
\begin{equation*}
f_{t}=8379 W_{t}^{1.097} \tag{6}
\end{equation*}
$$

which was used to predict age-specific fecundity (Figure 4).


Figure 6. The proportion of males in a Lake Sturgeon population across ages. Based on data from southern Quebec (Fortin et al. 1993).


Figure 7. Proportion of females mature-at-age. Based on data from Lake Winnebago, WI (Bruch 2008), locations in Quebec (Fortin et al. 1992), and the Ottawa River (Haxton 2008).

Female Lake Sturgeon spawn once every 2-7 years (COSEWIC 2017). The midpoint, 4.5 years, was used as a constant periodicity, $T$, value. The sex ratio is 1:1 at birth but becomes skewed towards females with age (Fortin et al. 1993). In southern Quebec, at age-of-maturity the sex ratio approaches 2:1, 6:1 by age 40, and 1:0 by age 80 (Figure 6; Fortin et al. 1993). As the model represents only females and $\varphi$ represent the proportion of offspring that are female $\varphi$ was therefore set to 0.5 . The sex ratio, later in life, only becomes a factor when estimating male population sizes.

Female Lake Sturgeon typically mature later than males; between ages 15 and 30 (COSEWIC 2017). Information on age-specific maturity were available from several sources (Figure 7). Age-at-50\%-maturity ( $t_{\text {mat }}$ ) ranged from 25-28 for populations in the Ottawa River and Quebec (Fortin et al. 1993, Haxton 2008) and 26 in Lake Winnebago, WI (Bruch 2008). Age 26 was used to represent the age-of-maturity which was held constant across population models.

## Mortality

Estimates of annual instantaneous mortality, $M$, were available for Lake Sturgeon populations in DU 2 and DU 4 with a single estimate from DU 3 from mark-recapture and catch curve analysis. Catch curve analysis typically resulted in lower $M$ estimates than mark-recapture studies (catch curve: mean $=0.070$, sd $=0.017$; mark-recapture: mean $=0.12$, sd=0.079). Estimates of adult instantaneous mortality, $M_{a}(\mathrm{n}=18)$, ranged from $0.020-0.236$ corresponding to annual survival rates, $\sigma_{a}$, of $98 \%$ and $79 \%$. Constant adult mortality rate (age $\geq t_{m a t}$ ) was assumed. $M_{a}$ was estimated with use of an empirical model from the literature to provide a growth pattern specific estimate. The estimates of $M_{a}$ (Table 3, Figure 8) were based on VBGF parameters (Table 2) which have been previously shown to relate to adult $M$ (Beverton and Holt 1959, Pauly 1980, Then et al. 2015); using the relationship (Then et al. 2015):

$$
\begin{equation*}
M_{a}=4.118 k^{0.73} L_{\infty}{ }^{-0.33} \tag{7}
\end{equation*}
$$

This gives estimates of $M_{a}$ (Table 3, Figure 8) that range between 0.053 and 0.091 ; corresponding to $\sigma_{a}$ values of $94.9 \%$ and $91.3 \%$. Populations with more rapid growth (larger $k$ ) were predicted to have greater mortality than populations with slow growth.

Table 3. Stage-specific survival rates for Lake Sturgeon populations with different growth patterns (Table 2) giving different average population growth rates ( $\lambda_{1}$ - stable; $\lambda_{\text {mean }}$ - growing; $\lambda_{\text {max }}$-booming). YOY survival rates are specific to each $\lambda$ while juvenile and adult rates apply across $\lambda$ s.

| Growth |  |  | YOY |  |  | Juvenile |  |
| :--- | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Pattern | $\boldsymbol{\lambda}_{\mathbf{1}}$ | $\boldsymbol{\lambda}_{\text {mean }}$ | $\boldsymbol{\lambda}_{\text {max }}$ | Young | Old | Young | Old |
| fast/large | 0.000015 | 0.000080 | 0.00062 | 0.810 | 0.916 | 0.925 | 0.925 |
| fast/medium | 0.000039 | 0.000207 | 0.00221 | 0.802 | 0.911 | 0.919 | 0.919 |
| fast/small | 0.000119 | 0.000562 | 0.00926 | 0.793 | 0.904 | 0.913 | 0.913 |
| slow/large | 0.000005 | 0.000038 | 0.00087 | 0.835 | 0.939 | 0.949 | 0.949 |
| slow/small | 0.000013 | 0.000097 | 0.00545 | 0.833 | 0.939 | 0.949 | 0.949 |

Juvenile mortality is expected to be greater than that of adults (Figure 8; Trested and Isely 2011, Pratt et al. 2014). Juvenile age-specific mortality was back-calculated from adult mortality assuming that mortality decreases as an inverse function of length (Lorenzen 2000):

$$
\begin{equation*}
M_{t}=m_{0} L_{t}^{-1} \tag{8}
\end{equation*}
$$

where $m_{0}$ is the mortality at a single unit of length. If $L_{t}$ is described by the von Bertalanffy growth curve (Equation 3), survival from age $t$ to $t+1$ can be calculated by combining Equations 3 and 8 and integrating (see Appendix A in van der Lee and Koops 2016); resulting in:

$$
\begin{equation*}
\sigma_{t}=\left[\frac{L_{t} e^{-k}}{L_{t+1}}\right]^{m_{0}} / k L_{\infty} \tag{9}
\end{equation*}
$$




Figure 8. Mortality (left) and survival (right) schedules for Lake Sturgeon populations with various growth patterns. The lines represent predicted rates and the whisker bars represent observed mean rates over an age range.

Equation 9 typically provides reasonable estimates of juvenile survival rate but often overestimates survival for young-of-the-year (YOY) fish. To determine YOY survival, $\sigma_{0}$, an optimization procedure was used to solve for the value of $\sigma_{0}$. Within stochastic simulations (see below) the mean $\sigma_{0}$ value that resulted in a predetermined average $\lambda$ value (the mean $\lambda$ across 1,000 replicates of 100 year simulations) was determined. This was done for each growth pattern model for three $\lambda$ values, $\lambda_{1}, \lambda_{\text {mean }}$, and $\lambda_{\text {max }}$ (Table 4). $A \lambda=\lambda_{1}$ indicates a stable population over time with a value of $1, \lambda_{\text {mean }}$ represents a growing population with a population growth rate that may be sustainable over a long time-frame, and $\lambda_{\max }$ represents a booming population or the maximum average rate of population growth attainable by a Lake Sturgeon population, this rate would likely only be possible at low population densities when a surplus of resources is available.

Few estimates of YOY mortality for Lake Sturgeon exist. Caroffino et al. (2010) quantified early life mortality over two years in the Peshtigo River, WI. Total instantaneous mortality, M, from egg to age-0 stage ( $\sim 3$ months of age) was 8.5 and 7.3 equivalent to survival rates of $0.02 \%$ and $0.07 \%$ (Caroffino et al. 2010). Crossman et al. (2009) estimated the overwinter survival rate of stocked YOY Lake Sturgeon (~ 6 months of age) in Black Lake, MI to be 40\%. Solved for estimates of YOY survival (Table 3) were greatest when $\lambda=\lambda_{\max }$ and least when $\lambda=\lambda_{1}$. Instantaneous annual YOY mortality, $M_{0}$, ranged from 9.0-12.2 when $\lambda=\lambda_{1}, 7.4-10.2$ when $\lambda=$ $\lambda_{\text {mean }}$, and 4.7-7.4 when $\lambda=\lambda_{\text {max }}$.

By estimating different $\sigma_{0}$ values to determine various rates of population growth (with $\lambda \geq 1$ ) this model implicitly assumed that population growth is regulated by density-dependence acting primarily on YOY fish; however, density-dependence is not explicitly incorporated into the model.

The value of $\lambda_{\text {mean }}$ was based on the geometric mean of a time series of estimated population growth rates from the lower Kettle River, MN between 1992 and 2007 ( $n=10$; Dieterman et al. 2010). Annual values were variable (range: 0.47-2.07); however, there was population growth over the sample period with a geometric mean of 1.035 . This value was used across growth patterns to represent mean population growth. This value is not meant to represent the sole $\lambda$ value possible by a growing population but instead as a representation of a reasonable level of
sustained growth. Other average $\lambda_{\text {mean }}$ values are possible and likely to occur; however, elasticity and allowable harm values would be similar in magnitude.

Table 4. Population status, generation time and reference population growth rates ( $\lambda$ ) for each growth pattern.

| Growth Pattern | Generation Time | $\boldsymbol{\lambda}_{\min }$ | $\boldsymbol{\lambda}_{\text {mean }}$ | $\boldsymbol{\lambda}_{\max }$ |
| :--- | :---: | :---: | :---: | :---: |
| fast/large | 37.1 | 0.95 | 1.035 | 1.088 |
| fast/medium | 33.1 | 0.95 | 1.035 | 1.100 |
| fast/small | 29.5 | 0.95 | 1.035 | 1.118 |
| slow/large | 53.9 | 0.95 | 1.035 | 1.111 |
| slow/small | 51.3 | 0.95 | 1.035 | 1.141 |

Maximum population growth rate was estimated from an allometric relationship (Randall and Minns 2000):

$$
\begin{equation*}
\lambda_{\max }=e^{2.64 W^{-0.35}} \tag{10}
\end{equation*}
$$

where $W$ represents weight at age-of-maturity. Across growth patterns, estimated $\lambda_{\max }$ ranged from 1.088-1.141 (Table 4). These values corresponded well to estimated maximum growth rates from Great Lakes populations of Lake Sturgeon which ranged from 1.088-1.174 (Haxton et al. 2014).

COSEWIC uses declines over a time frame of 10 years or 3 generations, whichever is longer, as one of the quantitative definition of species status. Endangered and Threatened species are defined as species that have experienced $70 \%$ and $50 \%$ declines over that time frame where the cause of decline has ceased or a $50 \%$ and $30 \%$ reduction if the decline is ongoing. There is no quantitative decline criterion for a Special Concern species. Lake Sturgeon have long generation times (Table 4) and have experience extensive declines in population size over the previous 3 generations due to overfishing, habitat alterations and other factors (COSEWIC 2017). Over the previous 3 generations populations in DU 1 have seen declines of $98 \%$, $90 \%$ in DU 2 and $99 \%$ in DU 4. The largest inferred decline ( $99 \%$ over 3 generations) was used as the value of minimum annual growth rate ( 0.95 , Table 4 ) across all models.

Four matrices were constructed with $\lambda=\lambda_{\text {min }}$ for each model. The vital rates from the matrix with $\lambda=\lambda_{1}$ were used and a scaler multiplier that would reduce the $\lambda$ value to equal $\lambda_{\text {min }}$ was solved for. This was done by reducing YOY survival ( $\sigma_{0}$ ), juvenile survival ( $\sigma_{j 1}$ and $\sigma_{j 2}$ ), adult survival ( $\sigma_{\mathrm{a} 1}$ and $\sigma_{\mathrm{a} 2}$ ), and fertility ( $F_{j}, F_{\mathrm{a} 1}$, and $F_{\mathrm{a} 2}$ ). Typically, large reductions in YOY survival and fertility were required to reach $\lambda_{\text {min }}$ while less significant reductions in juvenile and adult survival were required (Table 5).

Table 5. Scaler multiplier used to reduce individual vital rates to give average population growth rates ( $\lambda$ ) of $\lambda_{\text {min }}$ (Table 4)

| Growth Pattern | Survival |  |  |  |
| :--- | :--- | :--- | :--- | :--- |
|  | YOY | Juvenile | Adult |  |
| fast/large | 0.10 | 0.78 | 0.70 | 0.10 |
| fast/medium | 0.11 | 0.79 | 0.68 | 0.11 |
| fast/small | 0.12 | 0.79 | 0.65 | 0.12 |
| slow/large | 0.06 | 0.73 | 0.80 | 0.06 |
| slow/small | 0.06 | 0.73 | 0.80 | 0.06 |

## STAGE-SPECIFIC VITAL RATES

Stage-specific values of vital rates were estimated by combining age-specific estimates. Stage-specific fertility was calculated as the weighted mean of age-specific fertility value. The weights were based on the mean survival schedule (Figure 8, Equation 9) such that the influence of age-specific fertility on the stage mean was related to the mean stable age distribution (proportion of individuals by age).
Stage-specific survival rates were calculated from the mean of the annual age-specific instantaneous mortality rates within the stage:

$$
\begin{equation*}
\sigma_{i}=e^{-\frac{\sum_{x=t_{\min , i}}^{t_{\max , i}} M_{x}}{t_{\max , i}-t_{\min , i}}} \tag{11}
\end{equation*}
$$

where $t_{\text {max }, i}$ and $t_{\text {min, }, i}$ represent the minimum and maximum ages within stage $i$.

## STOCHASTICITY

Random, inter-annual variability was incorporated into simulations to account for the environmental stochasticity experienced by populations of Lake Sturgeon. Variability was incorporated at the age-specific level into growth (length) and mortality (Figure 9). Variability was not directly applied to fecundity; however, stochastic fecundity values resulted from inclusion of stochastic growth. Additionally, intra-annual correlation, within vital rates, was incorporated to account for the impacts of "good" or "bad" years for growth or mortality acting on the population as a whole.

## Length

Stochastic growth was simulated by applying variability to the inter-annual length increment, $L i_{t}$, between years. Mean $L i_{t}$ was calculated by subtracting $L_{t-1}$ from $L_{t}$ based on the von Bertalanffy growth curve (Equation 3, Table 2). Age-specific growth increments were varied following a stretched-beta distribution (a beta distribution rescaled to extend outside of the 0-1 range; Morris and Doak 2002). Use of a stretched-beta distribution allows the parameter to vary with the appropriate distributional shape while truncating the distribution so that the simulation is not impacted by extreme values of the tails of a typically distribution (e.g. normal) while maintaining the appropriate mean and standard deviation (Morris and Doak 2002). The inter-annual variation in growth increments within a Lake Sturgeon population was unknown. Lorenzen (2016) determined that the median temporal variation in length-at-age of wild fish stock was approximately $15 \%$. Variability was incorporated such that random growth increments had an approximately normal distribution with a coefficient of variation (CV) of 0.15. The random $\mathrm{Li}_{t}$ value was then added to the $L_{t-1}$ from the previous iteration to give the current $L_{t}$, with age-specific length probabilities. Initial $L_{t}$ values were randomly generated following the mean von Bertalanffy growth curve.

## Mortality

Age-specific instantaneous mortality was also varied following a stretched-beta distribution (Morris and Doak 2002). The inter-annual variability in mortality for Lake Sturgeon was unknown. Bradford (1992) found that across species and life-stages the variance in mortality increases as a function of $M\left(s d(M)=0.39 M^{1.12}\right)$. Mertz and Meyers (1995) determined that this variance estimate was likely inflated due to error from field estimates of $M$ and proposed that the inter-annual variability in $M$ could be represented by a constant CV of 0.2 . The stochastic distributions of $M$ were based on a normal distribution with means converted from Equation 9 and a CV of 0.2 for all age classes. The normal distribution for each age-specific mortality value
was converted to a stretched-beta distribution with unique ranges and distribution shapes (Figure 9). Although the mean mortality schedule was length dependent the stochastic mortality values were not impacted by stochastic length-at-age. This was done to maintain a constant mean mortality schedule across years and replicates with random variability around the mean trend rather than allowing for the mortality schedule itself to vary inter-annually.


Figure 9. Example of the probability density functions of stochastic parameters (length and mortality) incorporated into simulations. Values are representative of the fast/small growth pattern. NOTE: age increases along the $x$-axis from left to right for length but decreases from left to right for mortality and the scale of the instantaneous mortality axis had been $\log _{10}$ transformed.


Figure 10. Probability density function of population growth rate ( $\lambda$ ) resulting from incorporation of stochastic vital rates for annual and long term (100 year) simulations.

## Correlation

Individual intra-annual correlation structures were applied to growth and mortality such that age-specific variables were related to each other depending on the magnitude of the correlation coefficient, $r$. A correlation coefficient of 1.0 was used such that age-specific vital rates were completely correlated within years; except for YOY survival which was allowed to vary independently of other age classes with $r=0$. This assumes that different factors influence first year survival than other ages or that stochastic elements acting over short time frames may have a strong influence on YOY survival (i.e., the larval period (Ludsin et al. 2014), overwinter period (Hurst 2007) while the same time frame may not be impactful at later life stages.

Incorporating inter-annual stochasticity and intra-annual correlations of vital rates into projection matrices results in a probability distribution of $\lambda$ values. As population growth is a multiplicative process the distribution of annual population growth rates is log-normally distributed (Figure 10 left panel) with a $\log _{e}(s d)=0.05$. The long-term average population growth rate (geometric mean over 100 years) has an approximately normal distribution (Figure 10 right panel) with a standard deviation of 0.013 and approximate $99 \%$ confident intervals of $0.974-1.039$. Therefore, although on average the population experiences a growth rate of 1 , due to stochastic environmental variation over a period as long as 100 years there is a $1 \%$ chance individual populations may experience population declines of up to $3 \%$ or increases up to $4 \%$.

## SENSITVITY ANALYSIS

Sensitivity analysis of matrix population models determines the impact of changes to vital rates and lower level parameters on annual population growth rate ( $\lambda$ ). Sensitivities were quantified though estimation of elasticities $\left(\varepsilon_{v}\right)$ which describe the proportional change in $\lambda$ following a proportional perturbation in a vital rate ( $v$ ). For example, an elasticity value of 0.2 for adult survival $\left(\varepsilon_{a}\right)$ would indicate that a $20 \%$ increase in adult survival would result in a $4 \%(20 \% \times$ $0.2=4 \%$ ) increase to $\lambda$ (i.e., a $\lambda$ of 1.5 would increase to $1.56(1.5 \times(1+0.04)=1.56)$ ).

Elasticities are calculated by taking the scaled partial derivatives of $\lambda$ with respect to the vital rate:

$$
\begin{equation*}
\varepsilon_{v}=\frac{v}{\lambda} \sum_{i, j} \frac{\partial \lambda}{\partial a_{i, j}} \frac{\partial a_{i, j}}{\partial v} \tag{12}
\end{equation*}
$$

where $a_{i j}$ is the projection matrix element in row $i$ and column $j$. Elasticities are additive; as such, the effect of perturbations acting on multiple vital rates can be assessed by summing the elasticities of the affected vital rates.
Variation in model parameters was incorporated to determine effects on population responses from demographic perturbations (see Vélez-Espino and Koops 2007). Computer simulations were used to

1. generate 50,000 matrices with stochastic parameters ( $L_{t}$, and $\sigma_{t}$ ) drawn from distributions described above;
2. calculate the $\varepsilon_{v}$ of $\lambda$ with respect to $\sigma_{i}$ and $f_{i}$ for each matrix;
3. estimate mean stochastic elasticities and their $95 \%$ confidence intervals; and
4. repeat steps ito iii for matrices with $\lambda$ of $\lambda_{\text {min }}, \lambda_{1}, \lambda_{\text {mean }}$, and $\lambda_{\text {max }}$.

## ALLOWABLE HARM

Allowable harm defines the maximum harm to a population (decrease in vital rate(s)) that will not prevent population recovery for a given initial mean population growth rate. Allowable harm applies when a population has an initial $\lambda>1$. When a population experiences negative growth, $\lambda<1$, there is no scope for harm; however, recovery effort can be estimated, defined as the minimum changes to vital rate(s) required to allow for recovery to begin. Estimates of allowable chronic harm and transient harm are provided. Chronic harm refers to a permanent negative alteration to vital rate(s) while transient harm refers to a one time (temporary) mortality event impacting one or more life stage.
Allowable chronic harm and recovery effort can be estimated deterministically within a demographic framework following Vélez-Espino and Koops (2009):

$$
\begin{equation*}
\psi_{v} \text { or } \tau_{v}=\left(\frac{1}{\varepsilon_{v}}\right)\left(\frac{\lambda_{T}-\lambda}{\lambda}\right) \tag{13}
\end{equation*}
$$

Where $\varepsilon_{v}$ is the elasticity of vital rate $v, \lambda_{T}$ is the target lambda, and $\lambda$ is the initial population growth rate. Allowable harm and recovery effort are estimated with $\lambda_{T}=1$. If the recovery effort or harm impacts more than one vital rate it is calculated by summing the elasticity values $\left(\varepsilon_{v}\right)$ of each vital rate before inclusion in Equation 13.
In addition, simulation analysis was used to determine the risk associated with various levels of chronic harm to population recovery as follows:

1. generate 50,000 matrices with stochastic parameters ( $L_{t}$, and $\sigma_{t}$ ) drawn from distributions described above with an initial $\lambda=\lambda_{\text {mean }}$;
2. chronic harm, in the form of mortality, was applied as deaths per 100 individuals ranging from 0-99;
3. estimate average $\lambda$ after applying harm over various time frames (1 and 100 year(s));
4. calculate the proportion of replicates where $\lambda<1$ to assess the risk of population decline for each level of harm; and
5. repeat steps ii to iv for vital rates $\sigma_{\mathrm{YoY}}, \sigma_{\mathrm{j}}, \sigma_{\mathrm{a}}, \sigma_{1+}, \sigma$.

The effects of transient harm were estimated in a similar manner:

1. generate 50,000 matrices with stochastic parameters ( $L_{t}$, and $\sigma_{t}$ ) drawn from distributions described above with an initial $\lambda=\lambda_{\text {mean }}$;
2. Organize matrices into 5,000 replicates of 10 year simulations;
3. Transient harm, in the form of mortality, was applied as deaths per 100 individuals ranging from 0-99 to the first projection matrix, simulating a one-time removal of individuals;
4. estimate mean $\lambda$ after applying transient harm;
5. calculate the proportion of replicates where $\lambda<1$ to assess the risk of population decline for each level of harm; and
6. repeat steps iii to $v$ for vital rates $\sigma_{\mathrm{Yoy}}, \sigma_{\mathrm{j}}, \sigma_{\mathrm{a}}, \sigma_{1_{1}}, \sigma$.

## RECOVERY TARGETS

## Abundance: Minimum Viable Population (MVP)

To identify potential population recovery targets for Lake Sturgeon the concept of demographic sustainability simulated with population viability analysis (PVA; Morris and Doak 2002) was applied. From PVA arises the minimum viable population (MVP; Shaffer 1981). MVP is defined as the minimum adult female population size that results in a desired probability of persistence over a defined timeframe. Since population growth is not sustainable over time, the probability of persistence was simulated for a stable population over the long-term, $\lambda_{1}$.

Important elements incorporated in PVA include: the choice of timeframe over which persistence is determined, the severity and probability of a catastrophic event, and the quasiextinction threshold below which a population is deemed unviable. The choice of timeframe is arbitrary. It must be long enough to represent a significant amount of biological time for the species in question but also be a reasonable time frame over which to make management decisions. Typical time frames such as 100 years (Shaffer 1981) or 40 generations (Reed 2003b) were not appropriate as 100 years is not a significant amount of biological time for Lake Sturgeon (only 2.5 to 4 generation) and 40 generations represents between 1,000 and 1,600 years which is unreasonable for management considerations. Therefore, results are presented for timeframes of 250 years (MVP) and up to 500 years (persistence probabilities).
The rate and severity of catastrophic events occurring within Lake Sturgeon populations is not known. Reed et al. (2003a), through a meta-analysis, determined that among vertebrate populations catastrophic die-offs that result in a 1 year decrease in population size of $50 \%$ or greater occurred at a rate of $14 \% /$ generation on average. A rate of $0.15 /$ generation was incorporated in the PVA.

Quasi-extinction results from the compounding effects of Allee effects, demographic stochasticity and inbreeding depression (Lande 1988, Morris and Doak 2002) leading a population to extinction once the threshold is crossed. The quasi-extinction threshold is a simplifying assumption that allows for inclusion of these effects without having to explicitly parameterize them in the simulation model, which would require a number of unverifiable assumptions and increased computation time (Morris and Doak 2002). Use of a quasi-extinction threshold, rather than complete population extinction, improves model accuracy. Instead of attempting to predict the circumstances surrounding the death of the last individual in a population predictions are made based on the point when a population becomes small enough that the above mentioned genetic and ecological complications further increase the likelihood of
complete extinction (Morris and Doak 2002). The value on the quasi-extinction threshold cannot be empirically measured; therefore, 25 adult females was used as a reasonable approximation (Morris and Doak 2002).
Recovery targets were estimated as follows:

1. Random matrices were generated with stochastic parameters $\left(L_{t}\right.$, and $\left.\sigma_{t}\right)$ drawn from distributions described above with an initial $\lambda=\lambda_{1}$;
2. Population size was simulated over 500 years with initial population sizes ranging from 3015,000 female Lake Sturgeon;
3. Catastrophes were incorporated at a rate of $0.15 /$ generation and resulted in a $50 \%$ decline in total population abundance;
4. Population extinction occurred when adult female population size was $\leq 25$.

From these simulations, cumulative density functions for various population sizes were created to provide estimates of population extinction probability over time (up to 500 years). As well, the MVP (minimum female population size) required to provide a $5 \%$ and $1 \%$ probability of extinction over 250 years was calculated by fitting a logistic regression to binomial extinction data (1: extinct, 0: extant). Approximate male MVP values were also estimated from female MVPs based on the assumed sex ratio (Figure 6). As males mature earlier ( $50 \%$ maturity at age-20; Bruch 2008), stage classification (i.e., juvenile and adult) differed from females with juveniles defined as ages $1-19$ and adults bounded by ages 20 and 80 .

## Critical Habitat: Minimum Area for Population Viability (MAPV)

Following Vélez-Espino et al. (2010) and Young and Koops (2014), the minimum area for population viability (MAPV) was estimated as a first order quantification of the amount of habitat required to support a viable population. MAPV was estimated inclusive of both female and male Lake Sturgeon, and is calculated as:

$$
\begin{equation*}
M A P V=\sum_{t=0}^{t_{\max }} M V P_{t} \cdot A P I_{t} \tag{14}
\end{equation*}
$$

where $\mathrm{MVP}_{\mathrm{t}}$ is the age-specific minimum number of fish required to achieve the desired probability of persistence over 250 years, as estimated for the recovery target; and API $_{t}$ is the area required per individual of age $t$ (Minns 2003). Individuals were distributed among stage classes according to the stable stage distribution, which is represented by the dominant right eigenvector ( $w$ ) of the mean projection matrix based on $\lambda=1(A w=\lambda w$ ) (de Kroon et al. 1986). $\mathrm{API}_{t}$ was estimated by taking the inverse of density $\left(A P I_{t}=1 / D_{t}\right)$. Fish assemblage density (no ha' ${ }^{-1}$ ) has been found to relate to the mean weight, $W(\mathrm{~g})$, of the fish in the assemblage (Randall et al. 1995) following:

$$
\begin{equation*}
D=a_{D} W^{b_{D}} \tag{15}
\end{equation*}
$$

Density is habitat specific with fish in lotic environments maintaining greater densities than fish in lentic environments, at equal mean weights (Randall et al. 1995). Parameter values for Equation 16 are specific to habitat with $a_{D}=78,432$ and $b_{D}=-0.94$ in lotic environments and $a_{D}$ $=30,200$ and $b_{D}=-1.01$ in lentic environments. Although this allometry was initially estimated from measurements of fish assemblage densities and weights it is applied here to make species-specific estimates of density. As a result, MAPV values using these parameters provide estimates of Lake Sturgeon exclusive habitat requirements. In natural environments, interactions with other species will result in increased spatial requirements per individual fish.

Density data were available for a number of Lake Sturgeon populations in North America
(Figure 11, Appendix Table A2). Lake Sturgeon density data typically represented system wide
(lake or river) estimates of the number of fish (both sexes) $>1,000 \mathrm{~mm}$ in length. Density relationships were created for these data to provide estimates of habitat requirements inclusive of the wider fish assemblage. The slopes ( $b_{D}$ ) of the above relationships were conserved between the assemblage and species-specific levels but the intercept ( $a_{D}$ ) varied. Lake Sturgeon densities are low relative to historic levels (COSEWIC 2017). Therefore, measured densities for current population sizes may over-estimate spatial habitat requirements of a healthy population. Therefore, $a_{D}$ values were fit to Lake Sturgeon-specific density data for lotic and lentic environments to represent maximum and median density relationships. MAPV values estimated from the median density relationship represent the area requirements of an extant average size population. MAPV values estimated from the maximum density relationship may be a better representation of the area requirements of a healthy population.


Figure 11. Lake Sturgeon density data for lake and river populations plotted against mean population weight. Lines represent predictive density relationships from the literature (Randall et al. 1995) and based on median and maximum measure Lake Sturgeon densities (Appendix Table A2).

Space requirements of an age class can increase or decrease throughout the year depending on the age-specific mortality and growth rates. The required space for a given cohort is maximized at a given age ( $t_{\text {area }}$ ) which was estimated through simulations. The required space of a cohort increased until age $t_{\text {area }}$ and decreased afterwards. MAPV was estimated by finding the maximum age-specific MAPV (accounting for growth and mortality) and summing across age-classes. MAPV values were estimated by summing maximum MAPV ${ }_{t}$ values for age classes with lengths $\geq 1,000 \mathrm{~mm}$. This was done because most density estimates were made for whole systems (lakes or rivers) and only fish $\geq 1,000 \mathrm{~mm}$ were counted. It was assumed that the system would support the fish required to produce the counted fish and therefore no additional habitat is required.

## RECOVERY TIMES

Recovery times of Lake Sturgeon populations were estimated using simulation analysis. Rate of recovery is influenced primarily by the average population growth rate of the recovering populations. Population simulations were conducted with various initial population sizes (ranging from 50-1,000 females) at various mean $\lambda s$ (ranging from 1.01-1.15). $\lambda$ values were achieved by solving for the $\sigma_{\text {Yoy }}$ values that gave the desired $\lambda$. This, in effect, assumes population growth rate of Lake Sturgeon is primarily driven by density dependence acting on the YOY stage. If the
$\lambda s$ were achieved through alternative means (i.e., increasing adult survival rate) the results, in time to recovery, should be unaffected as long as the appropriate stable stage distribution is reached quickly.
Due to historic population declines many Lake Sturgeon populations have distributions skewed towards juveniles (T. Pratt, DFO, pers. comm.). The initial stage-distribution of Lake Sturgeon used in recovery simulations was based on the current juvenile-skewed distribution of the Lake Sturgeon population in Goulais Bay (T. Pratt, DFO, unpublished data). The sampled Goulais Bay populations consisted of $85.5 \%$ young juveniles, $13.5 \%$ old juveniles, $0.8 \%$ young adults, and $0 \%$ old adults. The proportion of YOY was determined using the stable-stage distribution of the matrix model for each growth rate.

Simulations were conducted in a similar manner to MVP analysis. Stochastic matrices were generated for 500 replicates of 1,500 year simulations. Annual population sizes were simulated and a population was considered recovered when it reached MVP (adult female) size. Recovery time was estimated by taking the $95^{\text {th }}$ percentile of time taken to reach MVP population size over replicates. A relationship was fit to predict time to a $95 \%$ probability of recovery (RT) from mean $\lambda$ as a hyperbolic curve $\left(R T=a_{r e c} /\left(\lambda+b_{r e c}\right)\right.$.

## RESULTS

## SENSITIVITY ANALYSIS

Elasticity patterns were similar across growth patterns (Table 6, Figure 12, Appendix Tables A3 and A4); though populations with fast growth were slightly more sensitive to younger life stages and less sensitive to older life stages than slow growth populations. Typically, the greatest elasticity values were produced from perturbations to young adult (age 26-62) survival rate. The exception to this was for populations experiencing declines where negative population growth was due to a reduction in adult survival $\left(\lambda_{\min }-F\right)$. In these instances $\lambda$ was most sensitive to perturbations of the survival rates of older juveniles (age 13-25).
Elasticities provide insight into how a population may react to perturbations to life history characteristics. For example, the model predicts Lake Sturgeon to have low sensitivity to reproduction and YOY survival with elasticities ranging from 0.008-0.07 across growth patterns and populations growth rates. This indicates that small perturbations to this aspect of life-history will not result in large changes to population state. Alternatively, assuming a growing population ( $\lambda=1.035$ ), elasticities of the young adult stage ranged from 0.44-0.50. Therefore a $10 \%$ decrease in survival rate (i.e., a $10 \%$ fishing mortality) would cause a $4.4-5 \%$ reduction in population growth rate (assuming other life history parameters remain constant) leading to population decline ( $\lambda \approx 0.98$ ).

Elasticity estimates assume other life history parameters remain constant. For example, elasticity analysis indicates that a population experiencing declines due to reduced adult survival (i.e., fishing mortality) will benefit most from increasing survival of older juveniles. This, however, assumes adult survival remains depressed. Instead if fishing mortality is reduced, initial population recovery may be slow (hence small elasticity values, Table 6), but once recovery begins and the population achieves the stable-stage distribution the elasticity patterns will begin to resemble that of $\lambda_{1}$ (blue bars) and recovery will accelerate.

As these estimates are stage-specific it is important to remember what the stages represent. Young adult survival produced the largest elasticity values partly because of its long stage length (38 years) where the young juvenile stage represented only 7 years. Therefore on a per age-class basis (assuming a stable populations, $\lambda=1$ ), pre-adult survival had elasticities
ranging from 0.033-0.046 while young adult survival elasticities ranged from 0.012-0.014. Preadult survival elasticities were approximately three times that of young adult survival per ageclass (i.e., if each age class was impacted separately). It is important to consider which ageclasses will be impacted by perturbations and how they align with stage divisions in this analysis. If overlap exists, elasticities must be adjusted to estimate the full impact of the perturbation.

Table 6. Mean values from stochastic sensitivity analysis of Lake Sturgeon population growth ( $\lambda$ ) rate to stage-specific perturbations of vital rates. The results are reported as elasticity ( $\varepsilon_{v}$ ) values and were estimated for various values of $\lambda$.

| Growth Pattern | $\lambda$ | Elasticity |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  |  | $\sigma_{0} / \mathbf{F}$ | $\sigma_{j 1}$ | $\sigma_{j 2}$ | $\sigma_{\text {a } 1}$ | $\sigma_{\text {a } 2}$ |
| fast/large | $\lambda_{\text {min }}-\sigma_{0}$ | 0.013 | 0.074 | 0.133 | 0.496 | 0.284 |
|  | $\lambda_{\text {min }}-\sigma_{j}$ | 0.016 | 0.043 | 0.051 | 0.559 | 0.330 |
|  | $\lambda_{\text {min }}-\sigma_{a}$ | 0.030 | 0.184 | 0.729 | 0.053 | 0.002 |
|  | $\lambda_{\text {min }}-F$ | 0.013 | 0.073 | 0.131 | 0.496 | 0.287 |
|  | $\lambda_{1}$ | 0.024 | 0.116 | 0.209 | 0.516 | 0.136 |
|  | $\lambda_{\text {mean }}$ | 0.035 | 0.158 | 0.282 | 0.484 | 0.041 |
|  | $\lambda_{\text {max }}$ | 0.052 | 0.210 | 0.352 | 0.382 | 0.003 |
| fast/medium | $\lambda_{\text {min }}-\sigma_{0}$ | 0.015 | 0.083 | 0.151 | 0.500 | 0.251 |
|  | $\lambda_{\text {min }}-\sigma_{j}$ | 0.019 | 0.051 | 0.061 | 0.572 | 0.297 |
|  | $\lambda_{\text {min }}-\sigma_{a}$ | 0.034 | 0.198 | 0.713 | 0.053 | 0.002 |
|  | $\lambda_{\text {min }}-F$ | 0.015 | 0.083 | 0.152 | 0.500 | 0.250 |
|  | $\lambda_{1}$ | 0.027 | 0.127 | 0.231 | 0.512 | 0.103 |
|  | $\lambda_{\text {mean }}$ | 0.039 | 0.171 | 0.303 | 0.462 | 0.025 |
|  | $\lambda_{\text {max }}$ | 0.061 | 0.229 | 0.371 | 0.339 | 0.001 |
| fast/small | $\lambda_{\text {min }}-\sigma_{0}$ | 0.018 | 0.096 | 0.176 | 0.503 | 0.208 |
|  | $\lambda_{\text {min }}-\sigma_{j}$ | 0.023 | 0.061 | 0.074 | 0.586 | 0.256 |
|  | $\lambda_{\text {min }}-\sigma_{a}$ | 0.038 | 0.215 | 0.693 | 0.052 | 0.001 |
|  | $\lambda_{\text {min }}-F$ | 0.018 | 0.095 | 0.175 | 0.503 | 0.209 |
|  | $\lambda_{1}$ | 0.031 | 0.141 | 0.257 | 0.501 | 0.070 |
|  | $\lambda_{\text {mean }}$ | 0.043 | 0.182 | 0.322 | 0.438 | 0.014 |
|  | $\lambda_{\text {max }}$ | 0.071 | 0.248 | 0.385 | 0.295 | 0.000 |
| slow/large | $\lambda_{\text {min }}-\sigma_{0}$ | 0.008 | 0.051 | 0.092 | 0.475 | 0.373 |
|  | $\lambda_{\text {min }}-\sigma_{j}$ | 0.010 | 0.025 | 0.029 | 0.524 | 0.412 |
|  | $\lambda_{\min }-\sigma_{a}$ | 0.023 | 0.161 | 0.727 | 0.077 | 0.011 |
|  | $\lambda_{\text {min }}-F$ | 0.008 | 0.051 | 0.092 | 0.475 | 0.374 |
|  | $\lambda_{1}$ | 0.017 | 0.089 | 0.156 | 0.496 | 0.242 |
|  | $\lambda_{\text {mean }}$ | 0.027 | 0.133 | 0.233 | 0.498 | 0.109 |
|  | $\lambda_{\text {max }}$ | 0.052 | 0.213 | 0.347 | 0.384 | 0.005 |
| slow/small | $\lambda_{\text {min }}-\sigma_{0}$ | 0.008 | 0.053 | 0.097 | 0.472 | 0.369 |
|  | $\lambda_{\text {min }}-\sigma_{j}$ | 0.010 | 0.026 | 0.030 | 0.524 | 0.409 |
|  | $\lambda_{\text {min }}-\sigma_{a}$ | 0.025 | 0.172 | 0.701 | 0.088 | 0.014 |
|  | $\lambda_{\text {min }}-F$ | 0.009 | 0.054 | 0.098 | 0.472 | 0.009 |
|  | $\lambda_{1}$ | 0.018 | 0.093 | 0.165 | 0.493 | 0.231 |
|  | $\lambda_{\text {mean }}$ | 0.029 | 0.140 | 0.245 | 0.489 | 0.097 |
|  | $\lambda_{\text {max }}$ | 0.064 | 0.239 | 0.367 | 0.329 | 0.001 |



Figure 12. Results of stochastic sensitivity analysis of Lake Sturgeon population growth rate ( $\lambda$ ) to perturbations in stage specific vital rates (survival ( $\sigma$ ) and fertility ( $F$ )). The results are reported as elasticities ( $\varepsilon_{v}$; mean, upper and lower confidence intervals) and were estimated for various values of $\lambda$.

## ALLOWABLE HARM

## Recovery effort and allowable chronic harm

Estimates of recovery effort and allowable harm are provided for two levels of population growth ( $\lambda_{\text {mean }}$ and $\lambda_{\text {max }}$ ) to provide a range of potential consequences of anthropogenic impacts on Lake Sturgeon populations. Realized population growth rates of natural populations will likely differ from those for which estimates were directly produced; however, the estimates may guide management on the impacts of harm (or recovery effort) on populations with growth rates within the range of those evaluated.

Table 7. Summary of recovery effort estimates of vital rates (stage-specific and aggregated across stages) for Lake Sturgeon populations in DU locations. Estimates were made for declining populations with differing causes of populations decline: reduce YOY survival ( $\sigma_{0}$ ); reduced juvenile survival ( $\sigma_{j}$ ); reduced adult survival ( $\sigma_{a}$ ); and reduced fertility ( $F$ ).

| Growth Pattern | $\lambda$ | Vital Rate |  |  |  |  |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  |  | $\boldsymbol{\sigma}_{0} / \mathbf{F}$ | $\sigma_{j 1}$ | $\sigma_{j 2}$ | $\sigma_{\text {a1 }}$ | $\sigma_{\mathrm{a} 2}$ | $\sigma_{j}$ | $\sigma_{a}$ | $\sigma_{1+}$ | $\sigma$ |
| fast/ large | $\lambda_{\text {min }}-\sigma_{0}$ | 62.74 | 9.50 | 3.54 | 0.14 | 0.90 | 2.57 | 0.12 | 0.06 | 0.05 |
|  | $\lambda_{\text {min }}-\sigma_{j}$ | 83.62 | 28.25 | 22.12 | 0.12 | 0.89 | 12.40 | 0.10 | 0.06 | 0.05 |
|  | $\lambda_{\text {min }}-\sigma_{a}$ | 78.69 | 7.74 | 0.12 | 30.40 | 656.34 | 0.07 | 28.96 | 0.06 | 0.05 |
|  | $\lambda_{\text {min }}-F$ | 63.97 | 9.71 | 3.60 | 0.14 | 0.89 | 2.62 | 0.12 | 0.06 | 0.05 |
| fast/ medium | $\lambda_{\text {min }}-\sigma_{0}$ | 43.83 | 6.84 | 2.47 | 0.14 | 1.11 | 1.80 | 0.13 | 0.06 | 0.05 |
|  | $\lambda_{\text {min }}-\sigma_{j}$ | 59.32 | 19.82 | 15.17 | 0.12 | 1.12 | 8.59 | 0.11 | 0.06 | 0.05 |
|  | $\lambda_{\text {min }}-\sigma_{a}$ | 43.96 | 4.61 | 0.12 | 19.40 | 569.31 | 0.07 | 18.70 | 0.06 | 0.05 |
|  | $\lambda_{\text {min }}-F$ | 44.05 | 6.89 | 2.47 | 0.14 | 1.07 | 1.81 | 0.12 | 0.06 | 0.05 |
| fast/ small | $\lambda_{\text {min }}-\sigma_{0}$ | 28.61 | 4.69 | 1.60 | 0.15 | 1.36 | 1.19 | 0.13 | 0.06 | 0.05 |
|  | $\lambda_{\text {min }}-\sigma_{j}$ | 39.83 | 13.29 | 9.93 | 0.12 | 1.40 | 5.68 | 0.11 | 0.06 | 0.05 |
|  | $\lambda_{\text {min }}-\sigma_{a}$ | 22.49 | 2.54 | 0.12 | 11.87 | 513.77 | 0.07 | 11.55 | 0.06 | 0.05 |
|  | $\lambda_{\text {min }}-F$ | 28.37 | 4.64 | 1.59 | 0.15 | 1.35 | 1.18 | 1.18 | 0.06 | 0.05 |
| slow/ large | $\lambda_{\text {min }}-\sigma_{0}$ | 107.69 | 15.15 | 6.23 | 0.13 | 0.36 | 4.40 | 0.10 | 0.05 | 0.05 |
|  | $\lambda_{\text {min }}-\sigma_{j}$ | 135.08 | 49.77 | 41.68 | 0.11 | 0.32 | 22.68 | 0.08 | 0.06 | 0.05 |
|  | $\lambda_{\text {min }}-\sigma_{a}$ | 152.35 | 13.13 | 0.13 | 28.49 | 155.02 | 0.07 | 24.02 | 0.06 | 0.05 |
|  | $\lambda_{\text {min }}-F$ | 108.80 | 15.32 | 6.30 | 0.13 | 0.36 | 4.45 | 0.10 | 0.05 | 0.05 |
| slow/ small | $\lambda_{\text {min }}-\sigma_{0}$ | 80.35 | 11.59 | 4.70 | 0.13 | 0.34 | 3.34 | 0.10 | 0.05 | 0.05 |
|  | $\lambda_{\text {min }}-\sigma_{j}$ | 99.14 | 37.08 | 31.05 | 0.11 | 0.30 | 16.90 | 0.08 | 0.06 | 0.05 |
|  | $\lambda_{\text {min }}-\sigma_{a}$ | 82.32 | 7.34 | 0.13 | 14.44 | 71.64 | 0.07 | 11.98 | 0.06 | 0.05 |
|  | $\lambda_{\text {min }}-F$ | 78.04 | 11.26 | 4.57 | 0.13 | 0.34 | 3.24 | 0.10 | 0.05 | 0.05 |

Recovery effort and allowable chronic harm estimates (Tables 7 and 8) were based on the lower and upper confidence intervals of stage-specific elasticity values from stochastic sensitivity analysis respectively (Appendix A3 and A4), following a precautionary approach. Values represent the proportional change to vital rates that would result in $\lambda=1$. Values below -1 indicate a lack of significant impacts of harm to that vital rate if all others are held constant for a given level of population growth. Recovery effort was estimated for populations with mean $\lambda \mathrm{s}$ of $\lambda_{\text {min }}$ and allowable harm was estimated for populations with mean $\lambda \mathrm{s}$ of $\lambda_{\text {mean }}$ and $\lambda_{\text {max }}$.
In most instances, improvements in $\lambda$ to values $\geq 1$ were achieved most easily with improvements to adult survival rates when compared to other stage-specific vital rates (Table 7). The exception to this was for populations with initially reduced adult survival where improvements to juvenile survival provided the best way to reach $\lambda$ to values $\geq 1$. The YOY
stage typically required significant improvement to allow for a $\lambda \geq 1$ and, therefore, may not represent a suitable avenue for anthropogenic intervention to initiate population growth.

Similarly, growing populations were more susceptible to harm to the adult stage, especially young adults, followed by the juvenile stage (Table 8). Populations were more resilient to harm to the YOY stage and at large growth rates $\left(\lambda_{\max }\right)$ were not susceptible to harm to old adults.

Table 8. Summary of allowable chronic harm estimates of vital rates (stage-specific and aggregated across stages) for Lake Sturgeon populations in DU locations. Estimates were made for population experiencing average rates of population growth of $\lambda_{\text {mean }}$ and $\lambda_{\max }$ (Table 3).

| Growth Pattern | $\lambda$ | Vital Rate |  |  |  |  |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  |  | $\sigma_{0} / \mathbf{F}$ | $\sigma_{j 1}$ | $\sigma_{j 2}$ | $\sigma_{\text {a } 1}$ | $\sigma_{\mathrm{a} 2}$ | $\sigma_{j}$ | $\sigma_{a}$ | $\sigma_{1+}$ | $\sigma$ |
| fast/ | $\lambda_{\text {mean }}$ | -0.358 | -0.119 | -0.094 | -0.048 | -0.176 | -0.053 | -0.042 | -0.034 | -0.034 |
| large | $\lambda_{\text {max }}$ | -0.712 | -0.263 | -0.210 | -0.13 | <-1 | -0.134 | -0.174 | -0.093 | -0.091 |
| fast/ | $\lambda_{\text {mean }}$ | -0.353 | -0.119 | -0.093 | -0.048 | -0.295 | -0.053 | -0.042 | -0.034 | -0.034 |
| medium | $\lambda_{\text {max }}$ | -0.777 | -0.294 | -0.232 | -0.175 | <-1 | -0.134 | -0.174 | -0.093 | -0.091 |
| fast/ | $\lambda_{\text {mean }}$ | -0.349 | -0.118 | -0.092 | -0.050 | -0.562 | -0.053 | -0.046 | -0.034 | -0.034 |
| small | $\lambda_{\text {max }}$ | -0.863 | -0.333 | -0.259 | -0.243 | <-1 | -0.152 | -0.243 | -0.109 | -0.105 |
| slow/ | $\lambda_{\text {mean }}$ | -0.416 | -0.126 | -0.099 | -0.054 | -0.098 | -0.056 | -0.037 | -0.034 | -0.034 |
| large | $\lambda_{\text {max }}$ | -0.925 | -0.331 | -0.263 | -0.168 | <-1 | -0.151 | -0.164 | -0.101 | -0.100 |
| slow/ | $\lambda_{\text {mean }}$ | -0.427 | -0.127 | -0.100 | -0.054 | -0.109 | -0.056 | -0.038 | -0.034 | -0.034 |
| small | $\lambda_{\text {max }}$ | <-1 | -0.403 | -0.323 | -0.262 | <-1 | -0.184 | -0.128 | -0.127 | -0.123 |



Figure 13. Example of the probability density function of population growth rate ( $\lambda$ ) from a Lake Sturgeon population experiencing $\lambda_{\text {mean }}$ conditions and no harm as well as maximum allowable harm annually and over 100 years (long term).

To examine the risks to population growth associated with harm simulations were conducted to determine the probability of population decline at various levels and types of harm (in the form of stage-specific deaths per 100 individuals). The distributions of $\lambda$ from annual and long-term
(100 year) simulations (Figure 13) display the likelihood of given rates of population growth for an unharmed ( $\lambda_{\text {mean }}$ ) population and a population experiencing maximum allowable harm ( $\lambda \approx 1$ ). An unharmed population had an annual probability of population decline of $\sim 56 \%$ and $0 \%$ over 100 years. At maximum allowable harm the likelihood of population decline increased to $76 \%$ annually and $31 \%$ over 100 years.

The probability of population decline, to various life stages (YOY, juvenile, adult, age-1+, and all age-class), increases with the level of harm (Figures 14 and 15). From Figures 14 and 15 the risk (in the form of probability of decline) associated with rates of fish death can be determined on an annual and 100 year time frame. Risk increases sharply with harm to juvenile, age-1+, and all age classes. Even small amounts of chronic harm ( $\sim 5$ deaths per 100 individuals) could ensure long term population decline. The probability of population decline following deaths to adult fish reaches $100 \%$ at $\sim 30$ deaths per 100 fish over 100 years. The populations were less susceptible to deaths to YOY fish. There was a $50 \%$ risk of population decline following between 64 and 80 deaths per 100 YOY fish over 100 years.


Figure 14. The probability of annual Lake Sturgeon population decline ( $\lambda<1$ ) after experiencing increasing levels of chronic harm (deaths per 100 individuals per year) to various life stages annually.


Figure 15. The probability of 100 year Lake Sturgeon population decline ( $\lambda<1$ ) after experiencing increasing levels of chronic harm (deaths per 100 individuals per year) to various life stages over 100 years.

## Transient allowable harm

Transient harm was simulated as a one-time death event (deaths per 100 fish) with the effects measured over a 10-year period. The risk to population growth from transient harm was estimated similarly to chronic harm by calculating the likelihood of population decline (probability of $\lambda<1$ ) over 10 years (Figure 16). Across growth patterns, transient harm to age- $1+$ and all age classes had a similar effect reaching a $50 \%$ probability of population decline at between 26 and 30 deaths per 100 fish. Transient harm to YOY fish did not significantly affect population growth over 10 years. A one-time event of 99 deaths per 100 YOY fish only resulted in between
a $13 \%$ and $19 \%$ probability of population decline over 10 years. The effects of transient harm to juvenile and adult fish were growth pattern dependent. Populations with slow growth patterns were more affected by harm to adults reaching a 50\% decline probability between 49 and 52 deaths per 100 fish. Populations with fast growth were less affected by adult deaths; requiring between 64 and 71 deaths per 100 fish to reach a $50 \%$ decline probability. The effects of juvenile deaths were reversed with populations with fast growth populations affected slightly more by juvenile deaths; $50 \%$ decline probability after 49-51 deaths compared to 56 and 57 deaths for slow growth populations.


Figure 16. The probability of Lake Sturgeon population decline ( $\lambda<1$ ) after experiencing increasing levels of transient harm (one time fish death per 100 individuals) to various life stages annual.

## RECOVERY TARGETS

## Abundance: Minimum Viable Population (MVP)

The probability of extinction ( $P$ [ext.]) over 250 years was modeled as a function of female adult population size $\left(P_{o p}\right)$ using a logistic regression (Figure 17): $P[$ ext. $]=\frac{1}{1+e^{-\left(a_{M V P}+b_{M V P} \log _{10}\left(\text { Poo }_{F}\right)\right)}}$, $a_{M V P}$ and $b_{M V P}$ are fitted parameter values (Table 9). The equation can be rearranged and used to estimate minimum recovery target for adult female Lake Sturgeon for a desired probability of persistence over 250 years given the pre-defined population size, catastrophe and quasi-extinction criteria.

Table 9. Logistic regression parameter values fitted to PVA simulation results to predict the probability of population extinction from adult female population size.

| Growth Pattern | $\boldsymbol{a}_{\text {MVP }}$ | $\boldsymbol{b}_{\text {MVP }}$ |
| :--- | :--- | :--- |
| fast/large | 6.145 | -2.927 |
| fast/medium | 6.386 | -2.980 |
| fast/small | 6.315 | -2.972 |
| slow/large | 7.206 | -3.734 |
| slow/small | 7.166 | -3.796 |

In choosing recovery targets, the risks associated with extinction probability must be balanced with the costs associated with an increased target (increased recovery effort, longer time to recovery, etc.). Recovery target values are presented for a $5 \%$ and $1 \%$ risk of extinction using simulation criteria of populations affected by a catastrophe rate of $0.15 /$ generation with a quasi-extinction threshold of 25 adult females (Table 10). Additional targets, those with different extinction risks, can be estimated with use of the functional relationships (Table 9). In addition, estimates of population size of younger stages required to support an MVP sized population of females are provided (Table 10), as well as, the equivalent number of males based on the assumed age-specific sex ratio (Figure 6). As males mature earlier than females the adult stage for males was defined as those $\geq 20$ years of age while females were defined as those $\geq 26$ years of age. As well, estimates of MVP are reported as the number of females and males $>1,000 \mathrm{~mm}$ in length as this is a common threshold used in Lake Sturgeon population estimates.
Estimates of MVP differ across populations with different growth patterns. Populations with slow growth had much lower MVP estimates (1,446 and 1,255 adult females) than fast growth populations (MVP ranged from 4,664 to 4,836 adult females). This was largely due to the difference in generation time (Table 4) and frequency of catastrophic population decline (50\% reduction) incorporated into simulations. At the incorporated rate of $15 \%$ chance of catastrophe on average a catastrophic population declines occurred once every 169-247 years for fast growth sturgeon and every 342-359 for slow growth Lake Sturgeon.

In addition to the fitted MVP relationships (Table 9), the cumulative density functions of extinction probability of populations of different sizes (adult females) over time (up to 500 years) were generate (Figure 18). This allows for the determination of persistence probability of various adult female population sizes for additional time frames for each growth pattern. For example, based on Figure 18, a Lake Sturgeon population with fast growth that achieves medium maximum size (fast/medium) with a population size of 100 adult females has an approximately $35 \%$ probability of going extinct within 100 years. This risk increases to approximately $70 \%$ over 500 years.


Figure 17. PVA model results for Lake Surgeon populations with various growth patterns. Predicted probability of extinction over 250 years for various adult female population sizes assuming a $0.15 /$ generation probability of catastrophe and a quasi-extinction threshold of 25 adult females. Results were fitted as a logistic regression (Table 9). Reference lines indicate a 5\% and 1\% probability of extinction (Table 10).


Figure 18. Cumulative extinction probability for Lake Sturgeon populations with different growth patterns for various female population sizes over time. Values indicate the probability a population will go extinct within a given timeframe.

Table 10. Estimates of the minimum viable population (MVP) for Lake Sturgeon to achieve a 5\% and 1\% probability of persistence over 250 years; assuming a $0.15 /$ generation probability of catastrophe and a quasi-extinction threshold of 25 adult females. Adult female values were predicted from logistic regression models (Table 9), younger age classes were calculated based on the mean survival schedule (Figure 8), and male population sizes were calculated based on the sex ratio (Figure 6). Adult females represent age classes 26+; adult males represent age classes 20+.

| Growth <br> Pattern | Stage | Female |  | Male |  |
| :---: | :---: | ---: | ---: | ---: | ---: |
|  |  | $P[$ ext $=5 \%$ | $P[$ ext $=\mathbf{1 \%}$ | $P[$ ext $=5 \%$ | $P[$ ext $=\mathbf{1 \%}$ |
| fast/large | YOY | $36,645,747$ | $134,245,382$ | $36,645,747$ | $134,245,382$ |
|  | Juvenile | 20,427 | 74,831 | 9,892 | 36,239 |
|  | $>1000 \mathrm{~mm}$ | 3,623 | 13,271 | 2,365 | 8,662 |
|  | Adult | 1,273 | 4,664 | 974 | 3,567 |
|  | YOY | $24,977,685$ | $89,417,810$ | $24,977,685$ | $89,417,810$ |
| fast/medium | Juvenile | 27,420 | 98,161 | 12,991 | 46,506 |
|  | 1000 mm | 3,482 | 12,465 | 2,208 | 7,904 |
|  | Adult | 1,351 | 4,836 | 1,095 | 3,921 |
|  | YOY | $14,609,815$ | $52,481,895$ | $14,609,815$ | $52,481,895$ |
|  | Juvenile | 34,280 | 123,141 | 15,838 | 56,895 |
| fast/small | $>1000 \mathrm{~mm}$ | 2,484 | 8,922 | 1,442 | 5,181 |
|  | Adult | 1,304 | 4,685 | 1,125 | 4,040 |
|  | YOY | $11,254,148$ | $31,141,019$ | $11,254,148$ | $31,141,019$ |
| slow/large | Juvenile | 3,475 | 9,615 | 1,771 | 4,901 |
|  | $>1000 \mathrm{~mm}$ | 685 | 1,896 | 305 | 844 |
|  | Adult | 522 | 1,446 | 305 | 844 |
|  | YOY | $5,218,219$ | $14,204,218$ | $5,218,219$ | $14,204,218$ |
|  | Juvenile | 3,095 | 8,425 | 1,569 | 4,272 |
| slow/small | $>1000 \mathrm{~mm}$ | 394 | 1,072 | 124 | 337 |
|  | Adult | 461 | 1,255 | 268 | 729 |

## Critical Habitat: Minimum Area for Population Viability (MAPV)

MAPV was estimated inclusive of whole Lake Sturgeon populations (females and males of all life stages). MAPV is a first order approximation of the habitat required to sustain an MVP sized population based on population densities. Three estimates of MAPV are provided for lake and river habitats. The first based on an allometry from the literature of fish assemblage densities. MAPV values using these density estimates were estimated by summing area requirements across all age classes. Two more estimates were made by adjusting the allometry to Lake Sturgeon specific densities based on the maximum and median values. As measured, Lake Sturgeon densities typically represented system wide estimates of individuals $>1,000 \mathrm{~mm}$, MAPVs using these density estimates were made by summing the area requirements of individuals $>1,000 \mathrm{~mm}$ in length and assuming these values are inclusive of the habitat required by smaller fish. This assumption is valid if younger age classes are present in the systems where population size of fish $>1,000 \mathrm{~mm}$ were estimated.
Space requirements of a cohort changes with time due to the interactions of somatic growth and mortality and depended on habitat type and MVP estimate (Figure 19). MAPV was estimated by summing the maximum age-specific space requirements (Figure 19) across age classes (all age classes when densities were estimated based on Randall et al. (1995) and age classes with
length $\geq 1,000 \mathrm{~mm}$ when densities were estimated from Lake Sturgeon populations). MAPV estimates were made for each MVP simulation (Table 11); as well, MAPV estimates were made for a range of potential female population sizes (Figure 20, population sizes relate only to females although MAPV estimates encompass the entire population).


Figure 19. Space occupied (ha) by a cohort over time for populations with differing growth patterns occupying lakes or rivers. Estimated as the product of daily MVP size and API.

Table 11. Estimate of the minimum area for population viability (MAPV, ha) of the entire population (females and males) at MVP population size (Table 10) in lakes and rivers. Estimates are based on an allometry (Randall et al. 1995) and median and maximum Lake Sturgeon densities.

| Growth Pattern | P[ext] | Randall Density |  | Maximum Density |  | Median Density |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  |  | Lake | River | Lake | River | Lake | River |
| fast/small | 5\% | 1,304.7 | 267.1 | 1,319.2 | 397.65 | 4,841.5 | 2,845.54 |
|  | 1\% | 4,686.7 | 956.6 | 4,738.8 | 1,428.46 | 17,391.8 | 10,221.84 |
| fast/medium | 5\% | 2,528.0 | 506.5 | 2,556.1 | 754.04 | 9,381.1 | 5,395.76 |
|  | 1\% | 9,050.1 | 1,813.4 | 9,150.7 | 2,699.38 | 33,583.6 | 19,316.33 |
| fast/large | 5\% | 3,540.5 | 694.1 | 3,579.8 | 1,033.20 | 13,138.1 | 7,393.43 |
|  | 1\% | 12,969.8 | 2,542.6 | 13,114.0 | 3,784.95 | 48,129.0 | 27,084.55 |
| slow/large | 5\% | 541.9 | 106.5 | 548.0 | 158.52 | 2,011.0 | 1,134.31 |
|  | 1\% | 1,499.6 | 294.7 | 1,516.2 | 438.62 | 5,564.7 | 3,138.72 |
| slow/small | 5\% | 199.4 | 40.2 | 201.6 | 59.85 | 739.8 | 428.30 |
|  | 1\% | 542.7 | 109.5 | 548.7 | 162.92 | 2,013.8 | 1,165.85 |



Figure 20. Estimate of required habitat (ha) of the entire population (females and males) at various female population sizes (MVP). Estimates based on an allometry (Randall et al. 1995) and median and maximum observed Lake Sturgeon densities.

## RECOVERY TIMES

Recovery times were estimated as the time taken to reach MVP size from various initial female population sizes at annual rates of population increase ( $\lambda$ ) ranging from 1.01-1.15. Time to recovery depended on the initial population size and the rate of population growth, decreasing as population size and growth rate increased. Simulation results produced a range (distribution) of recovery times. For example, within populations with a fast/small growth pattern with an initial population of 500 female Lake Sturgeon recovery time distributions were broad and increased in breadth for lower population growth rates (Figure 21). For example, if mean population growth rate was 1.15 the $95 \%$ confidence interval for recovery times ranged from 33-54 years. Alternatively if mean population growth rate was only 1.01 the $95 \%$ confidence interval for recovery times ranged from 110-1,165 years.

A relationship was fit to simulation results to predict time to a $95 \%$ likelihood of recovery from mean population growth rate (i.e., the time taken for $95 \%$ of simulations to reach MVP size) from each initial population size (Figure 22). From this, the time to recovery can be predicted if the population size and the average population growth rate of the recovering population are known. The changes to vital rates needed to improve population growth rate to allow populations to recover can be estimated from elasticity analysis results.


Figure 21. Example probability density functions for recovery time simulations with an initial population size of 500 adult female Lake Sturgeon under various average population growth rates ( $\lambda$ ). Distributions represent the likelihood of recovery (reaching MVP) over time. The rug plot indicates the $95^{\text {th }}$ percentile and the values used in Figure 22.


Figure 22. Estimated time to $95 \%$ likelihood of recovery (reaching MVP) for Lake Sturgeon populations with various initial female population sizes experiencing various average population growth rates.

## DISCUSSION

## ELEMENTS

## Element 3: Estimate the current or recent life-history parameters for Lake Sturgeon

The best available data were assembled to provide life-history parameters for Lake Sturgeon. The value for each life-history parameter used in modelling is presented in Tables 2-5, Figures $2-9$, and Equations 3-11. Details regarding how the parameters were estimated and source data used are outlined in the Methods section of this report.
Element 12: Propose candidate abundance and distribution target(s) for recovery
Estimates of minimum viable population (MVP) for Lake Sturgeon assume a probability of catastrophe of $0.15 /$ generation, a 25 -adult female quasi-extinction threshold, and a probability of extinction of $5 \%$ and $1 \%$ over 250 years (Table 10). Estimates depended on growth pattern where fast growth, greater adult mortality, and resultant shorter generation times lead to larger MVP estimates: 4,600-4,800 females and 3,600-4,000 males compared to 1,250-1,450 females and 750-850 males (at a 1\% extinction probability).

The choice of recovery target is not limited to the scenarios presented. MVP estimates with additional persistence probabilities can be made using the parameter values listed in Table 9.

According to Reed et al. (2003a), catastrophic events (a one-time decline in abundance of 50\% or more) occur at a probability of $0.14 /$ generation in vertebrates. It is uncertain at what frequency catastrophic events occur for Lake Sturgeon populations, therefore, modelling of recovery targets included the most conservative catastrophe scenario, based on Reed et al. (2003a), of $15 \%$.

Recovery targets based on MVP can be easily misinterpreted as a reference point for exploitation or allowable harm. A recovery target is neither of these things because it pertains exclusively to a minimum abundance level for which the probability of long-term persistence within a recovery framework is high. Therefore, abundance-based recovery targets are particularly applicable to populations that are below this threshold, and are useful for optimizing efforts and resources by selecting those populations that are in the greatest need of recovery. These MVP targets refer to adult numbers only. If juveniles are being included in abundance estimates, then the MVP must include these age classes as well.
Additionally, MVP estimates for Lake Sturgeon were made using a post-breeding matrix model. This means that abundance estimates were made directly after spawning has occurred and before age-specific mortality has acted. Therefore, abundance estimates from MVP analysis represent maximum annual abundances for a given population. When compared to field observations of abundance sampling date relative to spawning date should be considered and the expected mortality rate over this time period accounted for.
Element 13: Project expected population trajectories over a scientifically reasonable time frame (minimum 10 years), and trajectories over to the potential recovery target(s), given current Lake Sturgeon population dynamics parameters.
Current trajectories of Lake Sturgeon populations are variable across its Canadian distribution ranging from decline to growing; however, specific estimates of population growth rate are lacking.

Element 14: Provide advice on the degree to which supply of suitable habitat meets the demands of the species both at present and when the species reaches the potential recovery target(s) identified in element 12.
Lake Sturgeon have a wide distribution throughout Canada occupying lakes and rivers across four freshwater biogeographic zones. Much of the habitat across Lake Sturgeon's distribution has become fragmented due to dam construction limiting migration and access to spawning habitat.

As a first order approximation of Lake Sturgeon habitat requirements the minimum area for population viability (MAPV) or the quantity of habitat required to support an MVP sized population was estimated. This provides an estimate of the amount of physical space an MVP sized population would occupy but does not take into account additional habitat required to complete aspects of the life cycle such as spawning habitat or migration corridors. MAPV estimates were made for each MVP value presented for lake and river habitat (Table 11).

MAPV estimates are provided from three different estimates of population density. A density allometry from the literature (Randall et al. 1995) was utilized that describes the relationship between fish assemblage density and average weight. This allometry, when applied to Lake Sturgeon habitat estimates, provides a value of Lake Sturgeon exclusive habitat (independent of other species from the fish assemblage) and is consistent with previous RPA models. As well, estimates of Lake Sturgeon habitat requirements from measurements of Lake Sturgeon densities based on median and maximum densities are reported. MAPV estimates from median Lake Sturgeon densities may represent average habitat requirements of extant Lake Sturgeon populations; however, as most populations are at low density and below carrying capacity this may overestimate required habitat. As a result, MAPV estimates were made from maximum density values which may better represent habitat requirements of healthy populations.

## Element 15: Assess the probability that the potential recovery target(s) can be achieved under the current rates of population dynamics, and how that probability would vary with different mortality (especially lower) and productivity (especially higher) parameters.

Current population trajectories of Lake Sturgeon populations are variable across its Canadian distribution ranging from declining to growing; however, specific estimates of population growth rate are lacking. Time to $95 \%$ likelihood of recovery was estimated for Lake Sturgeon populations of various initial abundances growing at various rates (Figure 22). From Figure 22 the likely time to recovery can be determined if $\lambda$ and adult female population size are known. The approximate improvements in vital rates required to allow a population to grow at any specific rate can be estimated from elasticity estimates and Equation 11.
Specific estimates of the minimum recovery effort required to allow for population growth were made (Table 7). These estimates are specific to the initial rate of population growth, $\lambda_{\text {min }}$. The results indicate that the cause of population decline has an impact on the best method with which to stimulate recovery. Under most scenarios, population decline caused by a reduction in YOY survival, juvenile survival or fertility, population growth could be most easily stimulated with improvements to adult survival (based on stage-specific vital rates). If, however, the population was in decline because of reduced adult survival (i.e., from fishing mortality) population growth was most easily stimulated by improving juvenile survival rates (note: this assumes adult survival rate remains depressed). Therefore it is important to consider current conditions and population status when determining the best method to initiate recovery.

Element 19: Estimate the reduction in mortality rate expected by each of the mitigation measures or alternatives in element 16 and the increase in productivity or survivorship associated with each measure in element 17.

No clear links have been identified between the mitigation measures and Lake Sturgeon mortality rates or productivity.
Element 20: Project expected population trajectory (and uncertainties) over a scientifically reasonable time frame and to the time of reaching recovery targets, given mortality rates and productivities associated with the specific measures identified for exploration in element 19. Include those that provide as high a probability of survivorship and recovery as possible for biologically realistic parameter values.

Without a direct link between mitigation measures and Lake Sturgeon mortality rates or productivity, this information cannot be provided.
Element 21: Recommend parameter values for population productivity and starting mortality rates and, where necessary, specialized features of population models that would be required to allow exploration of additional scenarios as part of the assessment of economic, social, and cultural impacts in support of the listing process.

The parameter values incorporated in the population models are based on the best available data for Lake Sturgeon in Canada and should be used for any future population modelling. Details regarding how the parameters were estimated and source data used are outlined in the Methods section of this report.

Element 22: Evaluate maximum human-induced mortality and habitat destruction that the species can sustain without jeopardizing its survival or recovery.

Allowable harm analysis was conducted for each growth pattern for population growth rates of $\lambda_{\text {mean }}$ and $\lambda_{\text {max }}$ (Table 8). Estimates of chronic allowable harm are provided at the stage-level as well as combined across stages. With a moderate level of population growth ( $\lambda_{\text {mean }}$ ) only small decreases in survival to juvenile and/or older Lake Sturgeon could be sustained without risking population decline. For example, for populations with fast/large growth patterns a proportional decrease in adult survival of more than $3.9 \%$ would cause population decline. Perturbations to YOY survival or fertility were less impactful with allowable chronic harm estimates of 35-43\% for a moderate rate of population growth.
The amount of risk associated with chronic harm was explored through simulation analysis. Risk, in the form of probability of population decline, was estimated on an annual and 100-year basis by calculating the probability that $\lambda$ would be $<1$ over each time frame (Figures 14 and 15). Harm was applied as a number of stage-specific death per 100 individuals. With a mean population growth rate of $\lambda_{\text {mean }}(1.035)$ there was a significant risk of population decline with no harm applied ( $\sim 55 \%$ ). Although, over the long term (100 years) there was no risk of population decline without harm applied. The risks of population decline increased sharply, annually and over the long term, when harm was applied to the juvenile stage, age-1+ and to all age classes. Harm was less impactful when applied to the adult stage and had little effect when applied to YOY fish unless a significant amount of harm (i.e., > 50\%) was applied. The specific amount of risk associated with harm will depend on the average $\lambda$ of the population experiencing harm.
The effects of transient harm were estimated similarly; however, for transient harm, harm was applied to only 1 year and the impacts over 10-year simulations were estimated. From Figure 16 the risk of population decline (i.e., smaller population size relative to before harm occurred)
following deaths resulting from transient harm to various life stages can be determined. The YOY stage was relatively insensitive to transient harm while other life stages saw significant risk of population decline following moderate levels of harm. The effects were similar across growth patterns.

## UNCERTAINTIES

Lake Sturgeon populations were modelled with five distinct growth patterns. Specific locations (Table 2) were chosen and predictive relationships were used to generate the growth patterns; however, due to the significant amount of residual variability in the predictive relationships it is unlikely that the models generated represent high-quality location-specific models. Instead, models were intended to represent the diversity of life history characteristics among Canadian populations. Therefore, when interpreting the results one must consider the ranges reported and then try, if possible, to determine where the population of interest may fall.

Although Lake Sturgeon is a well-studied species there is still uncertainty in many model parameters. Foremost, the same maturation schedule was used across all population models. Lake Sturgeon maturity is often presented as a range (e.g., 15-30 years; COSEWIC 2017) which is approximated by the maturation schedule incorporated (Figure 7). Although the four examples of female maturity schedules available were similar they did not span the Canadian distribution well. One observation was from Wisconsin, USA and three from Quebec. As growth in Lake Sturgeon shows significant differences it might be expected that age-at-50\%-maturity differs as well (Lester et al. 2004). Differences in age-at-50\%-maturity would likely have significant impacts on model predictions. More research to produce additional maturation schedules at other locations is required to determine how maturation may differ across its range and inform potential predictive relationships.

The estimates of MVP that resulted from the viability analysis may have been influenced by the incorporated population structure which was limited by data availability. Lake Sturgeon was modelled as a single panmictic population with density-independent population growth. Both of these assumptions can inflate estimates of MVP when compared to a meta-population structure (van der Lee et al. 2020) and density-dependent population growth (Roberts et al. 2016). van der Lee et al. (2020) conducted population viability analysis for Redside Dace (Clintostomus elongatus) and found the results were highly dependent on the assumed meta-population structure. MVP estimates assuming a single randomly mating population structure were up to more than five times that of a defined meta-population, depending on impacts of catastrophic die-offs, with all other life history characteristics held constant. Roberts et al. (2016) estimated MVP values for Roanoke Logperch (Percina rex) with inclusion of density-dependent and density-independent population growth. MVP estimates with density-dependence ranged from 200-4,200 adults while with density-independence greatly exceeded 1,000,000 adults. Both of these analyses incorporated data that were not available for Lake Sturgeon populations. van der Lee et al. (2020) incorporated movement data between defined
sub-populations and Roberts et al (2016) had access to long (17 year) population size time series from which population growth rate and density-dependence effects could be estimated. Further research into population size and structure of Lake Sturgeon populations will allow for refinement of model structure and improvement of population viability analysis.

The frequency and impacts of catastrophic events for Lake Sturgeon were unknown. Simulations were conducted assuming a catastrophe rate of 0.15/generation consistent with the mean rate across vertebrate taxa (Reed et al. 2003a). The specific rate at which catastrophic events affect Lake Sturgeon population is unknown and the rate at which catastrophes occur has been shown to greatly influence MVP estimates (Vélez-Espino and Koops 2012). As well,
catastrophes always result in a $50 \%$ reduction in abundance across all age classes. It is likely that the impacts of catastrophes have a range of effects (Reed et al. 2003a) which likely would influence model results. Research that identifies the magnitude and frequency of catastrophic events at the population level would greatly reduce uncertainty in estimates of MVP size, and is recommendation for the conservation of Lake Sturgeon.

In general, a conservative approach to estimating MVP and MAPV was taken leading to larger estimates. Simulations incorporated conservative criteria, 250 year time horizon, a $0.15 /$ generation catastrophe rate and a 25 -adult female quasi-extinction threshold. As well, strong intra-annual correlations within stochastic simulations were incorporated and the populations were simulated as isolated with no migration. Finally, no density dependence effects were incorporated to allow a population to rebound following catastrophes. Each of these assumptions results in increased estimates of MVP. Alternatives to these assumptions could be included into Lake Sturgeon population simulations which would likely lead to decreases in the estimated MVP; however, due to data limitations it is unclear the magnitude of the effects. For example, the strength of density-dependence or the rate and magnitude of migration between populations are unknown and would greatly affect model outputs. Therefore, it is pragmatic to make estimates based on the more conservative assumptions.

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## APPENDIX. DATA TABLES

Table A1. Von Bertalanffy growth function (VBGF) parameters and length-at-age-25 (TL ${ }_{25}$ ) data used in predictive relationships (Figures 2 and 3). Length values are in mm . Source represents the reference from which data were extracted not necessarily the initial publication.

| $L_{\infty}$ | k | to | TL25 | N | Length Range | Age Range | Location | Lat. | Lon. | Source |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| 1,403.8 | 0.11 | -0.56 | 1,319.4 | 237 | 830-1,660 | 8-59 | Rainy Lake, MI/ON | 48.64 | -93.03 | Adams et al. 2006 |
| 1,675.0 | 0.050 | -0.12 | 1,200.1 | - | 138-1,742 | 1-64 | Lake Michigan | 44 | -87 | Block 2001 |
| 1,308.0 | 0.095 | 0 | 1,186.3 | - | 25-1,250 | 1-34 | Round Lake | 50.62 | -95.77 | Block 2001 |
| 1,138.5 | 0.087 | 0 | 1,009.2 | - | 50-1,350 | 4-72 | Slave Falls-Point du Bios | 50.23 | -95.59 | Block 2001 |
| 1,664.0 | 0.052 | -4.53 | 1,305.6 | 4,248 | 107-1,680 | 13--69 | Lake Winnebago | 44 | -88.4 | Bruch 2008 |
| 1,889.0 | 0.045 | -2.48 | 1,340.5 | 5,916 | 108-1,940 | 13-96 | Lake Winnebago | 44 | -88.4 | Bruch 2008 |
| 1,979.2 | 0.038 | -1.10 | 1,245.1 | - | - | - | St. Francis River | 45.90 | -71.16 | Power and McKinley 1997 |
| 1,105.8 | 0.115 | -0.08 | 1,044.0 | - | 413-1,233 | 3-23 | Lake St. Francis | 45.17 | -74.37 | Block 2001 |
| 2,088.8 | 0.031 | -9.39 | 1,369.4 | 79 | 750-1,775 | 8-36 | White River, ON | 48.56 | -86.24 | Ecclestone 2012 |
| - | - | - | 1,254.0 | - | - | - | St. Lawrence River | 46.63 | -71.93 | Fortin et al. 1996 |
| - | - | - | 1,177.0 | - | - | - | Baskatong Reservoir, QC | 46.8 | -75.8 | Fortin et al. 1996 |
| - | - | - | 1,040.0 | - | - | - | Bell River, QC | 49 | -77.5 | Fortin et al. 1996 |
| - | - | - | 659.0 | - | - | - | Eastmain-Opinaca Rivers | 52.24 | -78.01 | Fortin et al. 1996 |
| - | - | - | 1,078.0 | - | - | - | Harricana, River | 50.53 | -79.11 | Fortin et al. 1996 |
| - | - | - | 1,118.0 | - | - | - | Lac Gueguen | 48.10 | -77.23 | Fortin et al. 1996 |
| - | - | - | 989.0 | - | - | - | Lower Ottawa River | 46 | -77.3 | Fortin et al. 1996 |
| - | - | - | 1,023.0 | - | - | - | Megiscan, Lake E | 48.6 | -75.85 | Fortin et al. 1996 |
| - | - | - | 1,003.0 | - | - | - | Megiscan, Lake W. | 48.56 | -75.92 | Fortin et al. 1996 |
| - | - | - | 1,040.0 | - | - | - | Nottaway, River | 50.05 | -77.47 | Fortin et al. 1996 |
| - | - | - | 1,058.0 | - | - | - | Rupert, River | 50.95 | -73.7 | Fortin et al. 1996 |
| - | - | - | 1,072.0 | - | - | - | Lake Temiscamingue | 47.33 | -79.5 | Fortin et al. 1996 |
| 1,120.8 | 0.129 | -0.07 | 1,076.6 | - | 355-1,165 | 2-25 | Lac Des Duex Montangues | 45.45 | -74 | Fortin et al. 1993 |
| 1,338.8 | 0.098 | -0.078 | 1,225.3 | - | 452-1,369 | 6-34 | Lac St. Louis | 45.40 | -73.81 | Fortin et al. 1993 |
| 1,350.0 | 0.091 | -0.083 | 1,213.8 | - | 634-1,313 | 7-23 | Lac St. Pierre | 46.20 | -72.83 | Fortin et al. 1993 |
| 1,570.0 | 0.044 | -1.182 | 1,073.8 | - | - | - | Lake Nipigon | 49.83 | -88.5 | Power and McKinley 1997 |
| 1,807.2 | 0.093 | -0.902 | 1,644.7 | 180 | 250-1,750 | 0-35 | Muskegon River/Lake, MI | 43.23 | -86.3 | Harris et al. 2017 |


| L | k | to | TL25 | N | Length Range | Age Range | Location | Lat. | Lon. | Source |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| 1,404.5 | 0.094 | -0.078 | 1,271.9 | - | 298-1,588 | 1-65 | Saskatchewan River, SK | 50.04 | -110.7 | Block 2001 |
| 1,289.0 | 0.061 | -3.0 | 1,055.4 | - | 40-1,000 | 2-46 | Ottawa River, harvested | 46.07 | -76.76 | Haxton and Findlay 2008 |
| 1,473.0 | 0.066 | -0.85 | 1,205.5 | - | 400-1,400 | 4-30 | Ottawa River, impounded | 46.07 | -76.76 | Haxton and Findlay 2008 |
| - | - | - | 1,517.2 | 61 | 300-1,600 | 1-23 | Lower Niagara River, NY | 43.21 | -79.06 | Hughes et al. 2005 |
| - | - | - | 1,262.0 | - | - | - | Upper St. Lawrence River | 44.54 | -75.75 | Fortin et al. 1996 |
| - | - | - | 1,491.0 | - | - | - | Lake Wisconsin | 43.31 | -89.72 | Fortin et al. 1996 |
| - | - | - | 1,166.0 | - | - | - | St. Lawrence River | 46.81 | -71.20 | Fortin et al. 1996 |
| 1,479.5 | 0.04 | -3.338 | 1,003.3 | - | - | - | Lake Waswanipi | 49.55 | -76.45 | Power and McKinley 1997 |
| 926.2 | 0.059 | -0.188 | 718.4 | - | 660-876 | 20-35 | La Grande River | 53.83 | -79.07 | Block 2001 |
| - | - | - | 1,143.0 | - | - | - | Lac Des Duex Montangues | 45.45 | -74 | Fortin et al. 1996 |
| 1,421.0 | 0.085 | -0.031 | 1,251.7 | NA | - | - | Lake of the Woods | 49.25 | -94.75 | Adams et al. 2006 |
| 1,561.9 | 0.043 | 4.901 | 903.8 | - | 155-1,580 | 1-69 | Groundhog/Mattagami River | 49.5 | -81.97 | Noakes et al. 1999 |
| 1,326.3 | 0.069 | -0.111 | 1,096.2 | - | 538-1,430 | 9-49 | Mattagami River | 48.01 | -81.56 | Block 2001 |
| - | - | - | 1,383.0 | - | - | - | Lake Nipissing | 46.27 | -79.79 | Fortin et al. 1996 |
| 1,382.5 | 0.036 | -0.205 | 828.5 | - | 282-1,513 | 2-48 | Kenogami River | 51.11 | -84.48 | Block 2001 |
| 1,583.7 | 0.047 | -0.135 | 1,106.8 | - | 860-1,767 | 12-87 | Sipiwesk Lake | 55.09 | -97.58 | Block 2001 |
| 2,028.0 | 0.031 | -6.5 | 1,264.2 | - | 365-1,510 | 1-30 | Goulais Bay, Lake Superior | 46.71 | -84.45 | Pratt et al. 2014 |
| 1,534.4 | 0.059 | -0.113 | 1,188.2 | - | 138-1,490 | 1-46 | Menominee River | 45.09 | -87.59 | Block 2001 |
| 1,852.1 | 0.055 | -0.146 | 1,387.5 | - |  |  | Lake Poygan, WI | 44.15 | -88.75 | Power and McKinley 1997 |
| 1,423.2 | 0.061 | -0.117 | 1,120.6 | - | 155-1,453 | 1-50 | Lake Winnebago | 44 | -88.4 | Block 2001 |
| 1,335.9 | 0.088 | -0.088 | 1,189.0 | - | 194-1,563 | 1-51 | Saskatchewan River, AB | 53.96 | -102.4 | Block 2001 |
| 1,395.6 | 0.064 | -1.388 | 1,137.8 | - |  |  | Mattagami River | 48.01 | -81.56 | Power and McKinley 1997 |
| 1,424.7 | 0.057 | -0.127 | 1,083.5 | - | 630-1,387 | 9-41 | Flambeau River | 45.30 | -91.24 | Block 2001 |
| 1,768.9 | 0.091 | 0.355 | 1,581.1 | - | 1,120-1,870 | 12-70 | Upper Black River, MI | 45.46 | -84.27 | Smith and Baker 2005 |
| 1,319.6 | 0.057 | -0.136 | 1,008.0 | - | 825-1,362 | 14-70 | Sipiwesk Lake | 55.09 | -97.58 | Block 2001 |
| 1,373.6 | 0.093 | -0.080 | 1241.0 | - | 187-1,570 | 1-62 | Nelson River | 55 | -102 | Sunde 1961 |
| 1,128.5 | 0.078 | -0.117 | 969.4 | - | 371-1,119 | 1-36 | Moose River | 51.35 | -80.4 | Block 2001 |
| 1,913 | 0.094 | -9.569 | 1,839.8 | 195 | 400-1,400 | 3-32 | Grasse River, NY | 44.99 | -74.77 | Trested and Isely 2011 |
| - | - | - | 1,015 | - | - | - | Saskatchewan River | 53.6 | -100.8 | Fortin et al. 1996 |
| 1,212.6 | 0.073 | -0.117 | 1,017.2 | 533 | 301-1,562 | 3-34 | Kettle River MN | 46.23 | -92.85 | Dieterman et al. 2010 |

Table A2. Lake Sturgeon density data used to estimate population area requirements (Figure 11).

| Density <br> (fish ha $^{-1}$ ) | Weight (kg) | Habitat | Location | Source |
| :--- | :--- | :--- | :--- | :--- |
| 3.38 | 6.4 | Lake | Round Lake, MB | Block 2001 |
| 2.03 | 2.1 | Lake | Round Lake, MB | Block 2001 |
| 2.37 | 7.7 | Lake | Round Lake, MB | Block 2001 |
| 5.41 | 2 | Lake | Round Lake, MB | Block 2001 |
| 0.12 | 0 | River | Seven Sisters-Slave Falls, MB | Block 2001 |
| 0 | 0 | River | Seven Sisters-Slave Falls, MB | Block 2001 |
| 0 | 0 | River | Seven Sisters-Slave Falls, MB | Block 2001 |
| 0.25 | 32.5 | River | Seven Sisters-Slave Falls, MB | Block 2001 |
| 0 | 0 | River | Seven Sisters-Slave Falls, MB | Block 2001 |
| 0.12 | 5.6 | River | Seven Sisters-Slave Falls, MB | Block 2001 |
| 0 | 0 | River | Seven Sisters-Slave Falls, MB | Block 2001 |
| 0.74 | 14.8 | River | Seven Sisters-Slave Falls, MB | Block 2001 |
| 0.74 | 7.4 | River | Seven Sisters-Slave Falls, MB | Block 2001 |
| 5.91 | 7.1 | River | Seven Sisters-Slave Falls, MB | Block 2001 |
| 11.21 | 2.3 | River | Seven Sisters-Slave Falls, MB | Block 2001 |
| 8.75 | 2 | River | Seven Sisters-Slave Falls, MB | Block 2001 |
| 0.3 | 2 | River | Slave Falls-Point du Bois, MB | Block 2001 |
| 1.49 | 2.6 | River | Slave Falls-Point du Bois, MB | Block 2001 |
| 1.64 | 7.9 | River | Slave Falls-Point du Bois, MB | Block 2001 |
| 1.86 | 8.2 | River | Slave Falls-Point du Bois, MB | Block 2001 |
| 7.9 | 11.7 | River | Namakan River, ON | McLeod 2008 |
| 0.24 | 11.98 | Lake | Lake of the Woods | Heinrich and Friday (2014) |
| 1.2 | 10.4 | River | Kaministiquia River | Ontario Ministry of Natural |
| 5 | 10.4 | River | Kaministiquia River | Resources (OMNR) 2009 |
| 3.37 | 8.1 | Lake | Slave Falls Reservoir | OMNR 2009 |
| 0.018 | 9.7 | Lake | Lac Deschenes | Manitoba Hydro unpublished |
| 0.3 | 12.7 | Lake | Black Lake, Ml | Haxton 2006 |
|  |  |  | Baker and Borgeson 1999 |  |

Table A3. Lower confidence intervals from stochastic sensitivity analysis of Lake Sturgeon population growth rate ( $\lambda$ ) to stage-specific perturbations of vital rates. The results are reported as elasticity $\left(\varepsilon_{v}\right)$ values and were estimated for various values of $\lambda$.

| Growth Pattern | $\lambda$ | Elasticity |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  |  | $\boldsymbol{\sigma}_{0} / \mathbf{F}$ | $\sigma_{j 1}$ | $\sigma_{j 2}$ | $\sigma_{\text {a } 1}$ | $\sigma_{\text {a } 2}$ |
| fast/large | $\lambda_{\text {min }}-\sigma_{0}$ | 0.001 | 0.006 | 0.015 | 0.371 | 0.059 |
|  | $\lambda_{\text {min }}-\sigma_{j}$ | 0.001 | 0.002 | 0.002 | 0.452 | 0.059 |
|  | $\lambda_{\text {min }}-\sigma_{a}$ | 0.001 | 0.007 | 0.431 | 0.002 | 0.000 |
|  | $\lambda_{\text {min }}-F$ | 0.001 | 0.005 | 0.015 | 0.371 | 0.059 |
|  | $\lambda_{1}$ | 0.001 | 0.009 | 0.027 | 0.299 | 0.009 |
|  | $\lambda_{\text {mean }}$ | 0.004 | 0.027 | 0.089 | 0.261 | 0.002 |
|  | $\lambda_{\text {max }}$ | 0.014 | 0.085 | 0.269 | 0.214 | 0.000 |
| fast/medium | $\lambda_{\text {min }}-\sigma_{0}$ | 0.001 | 0.008 | 0.021 | 0.363 | 0.048 |
|  | $\lambda_{\text {min }}-\sigma_{j}$ | 0.001 | 0.003 | 0.003 | 0.442 | 0.047 |
|  | $\lambda_{\text {min }}-\sigma_{a}$ | 0.001 | 0.011 | 0.435 | 0.003 | 0.000 |
|  | $\lambda_{\text {min }}-F$ | 0.001 | 0.008 | 0.021 | 0.366 | 0.049 |
|  | $\lambda_{1}$ | 0.002 | 0.013 | 0.042 | 0.294 | 0.007 |
|  | $\lambda_{\text {mean }}$ | 0.007 | 0.040 | 0.136 | 0.257 | 0.002 |
|  | $\lambda_{\text {max }}$ | 0.022 | 0.118 | 0.333 | 0.201 | 0.000 |
| fast/small | $\lambda_{\text {min }}-\sigma_{0}$ | 0.002 | 0.011 | 0.033 | 0.358 | 0.039 |
|  | $\lambda_{\text {min }}-\sigma_{j}$ | 0.001 | 0.004 | 0.005 | 0.430 | 0.038 |
|  | $\lambda_{\text {min }}-\sigma_{a}$ | 0.002 | 0.021 | 0.438 | 0.004 | 0.000 |
|  | $\lambda_{\text {min }}-F$ | 0.002 | 0.011 | 0.033 | 0.358 | 0.039 |
|  | $\lambda_{1}$ | 0.003 | 0.021 | 0.068 | 0.291 | 0.005 |
|  | $\lambda_{\text {mean }}$ | 0.010 | 0.058 | 0.194 | 0.254 | 0.001 |
|  | $\lambda_{\text {max }}$ | 0.033 | 0.152 | 0.365 | 0.187 | 0.000 |
| slow/large | $\lambda_{\text {min }}-\sigma_{0}$ | 0.000 | 0.003 | 0.008 | 0.392 | 0.148 |
|  | $\lambda_{\text {min }}-\sigma_{j}$ | 0.000 | 0.001 | 0.001 | 0.492 | 0.166 |
|  | $\lambda_{\text {min }}-\sigma_{a}$ | 0.000 | 0.004 | 0.413 | 0.002 | 0.000 |
|  | $\lambda_{\text {min }}-F$ | 0.000 | 0.003 | 0.008 | 0.391 | 0.147 |
|  | $\lambda_{1}$ | 0.001 | 0.006 | 0.015 | 0.338 | 0.030 |
|  | $\lambda_{\text {mean }}$ | 0.002 | 0.017 | 0.048 | 0.297 | 0.008 |
|  | $\lambda_{\text {max }}$ | 0.016 | 0.097 | 0.275 | 0.231 | 0.001 |
| slow/small | $\lambda_{\text {min }}-\sigma_{0}$ | 0.001 | 0.005 | 0.011 | 0.392 | 0.156 |
|  | $\lambda_{\text {min }}-\sigma_{j}$ | 0.001 | 0.001 | 0.002 | 0.494 | 0.177 |
|  | $\lambda_{\text {min }}-\sigma_{a}$ | 0.001 | 0.007 | 0.411 | 0.004 | 0.001 |
|  | $\lambda_{\text {min }}-F$ | 0.001 | 0.005 | 0.012 | 0.391 | 0.155 |
|  | $\lambda_{1}$ | 0.001 | 0.008 | 0.022 | 0.342 | 0.033 |
|  | $\lambda_{\text {mean }}$ | 0.003 | 0.023 | 0.066 | 0.303 | 0.010 |
|  | $\lambda_{\text {max }}$ | 0.029 | 0.147 | 0.346 | 0.218 | 0.000 |

Table A4. Upper confidence intervals from stochastic sensitivity analysis of Lake Sturgeon population growth rate ( $\lambda$ ) to stage-specific perturbations of vital rates. The results are reported as elasticity $\left(\varepsilon_{v}\right)$ values and were estimated for various values of $\lambda$.

| Growth Pattern | $\boldsymbol{\lambda}$ | Elasticity |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  |  | Go/F | $\sigma_{j 1}$ | $\sigma_{j 2}$ | $\sigma$ a1 | $\sigma_{\text {a } 2}$ |
| fast/large | $\lambda_{\text {min }}-\sigma_{0}$ | 0.049 | 0.216 | 0.301 | 0.559 | 0.468 |
|  | $\lambda_{\text {min }}-\sigma_{j}$ | 0.080 | 0.189 | 0.213 | 0.641 | 0.482 |
|  | $\lambda_{\text {min }}-\sigma_{a}$ | 0.105 | 0.351 | 0.991 | 0.114 | 0.006 |
|  | $\lambda_{\text {min }}-F$ | 0.050 | 0.216 | 0.301 | 0.559 | 0.468 |
|  | $\lambda_{1}$ | 0.081 | 0.268 | 0.346 | 0.652 | 0.397 |
|  | $\lambda_{\text {mean }}$ | 0.095 | 0.287 | 0.362 | 0.708 | 0.194 |
|  | $\lambda_{\text {max }}$ | 0.113 | 0.306 | 0.383 | 0.617 | 0.014 |
| fast/medium | $\lambda_{\text {min }}-\sigma_{0}$ | 0.053 | 0.224 | 0.309 | 0.576 | 0.450 |
|  | $\lambda_{\text {min }}-\sigma_{j}$ | 0.084 | 0.198 | 0.225 | 0.660 | 0.470 |
|  | $\lambda_{\text {min }}-\sigma_{a}$ | 0.106 | 0.354 | 0.984 | 0.107 | 0.005 |
|  | $\lambda_{\text {min }}-F$ | 0.052 | 0.222 | 0.308 | 0.576 | 0.450 |
|  | $\lambda_{1}$ | 0.083 | 0.271 | 0.350 | 0.670 | 0.343 |
|  | $\lambda_{\text {mean }}$ | 0.097 | 0.288 | 0.366 | 0.708 | 0.116 |
|  | $\lambda_{\text {max }}$ | 0.117 | 0.311 | 0.393 | 0.520 | 0.004 |
| fast/small | $\lambda_{\text {min }}-\sigma_{0}$ | 0.056 | 0.229 | 0.316 | 0.596 | 0.417 |
|  | $\lambda_{\text {min }}-\sigma_{j}$ | 0.088 | 0.206 | 0.235 | 0.683 | 0.450 |
|  | $\lambda_{\text {min }}-\sigma_{a}$ | 0.110 | 0.357 | 0.972 | 0.097 | 0.004 |
|  | $\lambda_{\min }-F$ | 0.056 | 0.229 | 0.315 | 0.596 | 0.417 |
|  | $\lambda_{1}$ | 0.084 | 0.272 | 0.354 | 0.687 | 0.266 |
|  | $\lambda_{\text {mean }}$ | 0.098 | 0.289 | 0.370 | 0.681 | 0.061 |
|  | $\lambda_{\text {max }}$ | 0.122 | 0.316 | 0.406 | 0.432 | 0.001 |
| slow/large | $\lambda_{\text {min }}-\sigma_{0}$ | 0.033 | 0.173 | 0.252 | 0.504 | 0.488 |
|  | $\lambda_{\text {min }}-\sigma_{j}$ | 0.054 | 0.127 | 0.142 | 0.567 | 0.494 |
|  | $\lambda_{\text {min }}-\sigma_{a}$ | 0.085 | 0.327 | 0.993 | 0.164 | 0.023 |
|  | $\lambda_{\text {min }}-F$ | 0.033 | 0.174 | 0.252 | 0.504 | 0.488 |
|  | $\lambda_{1}$ | 0.065 | 0.244 | 0.323 | 0.567 | 0.463 |
|  | $\lambda_{\text {mean }}$ | 0.082 | 0.271 | 0.345 | 0.632 | 0.349 |
|  | $\lambda_{\text {max }}$ | 0.108 | 0.301 | 0.373 | 0.592 | 0.018 |
| slow/small | $\lambda_{\text {min }}-\sigma_{0}$ | 0.032 | 0.169 | 0.248 | 0.503 | 0.485 |
|  | $\lambda_{\text {min }}-\sigma_{j}$ | 0.052 | 0.122 | 0.137 | 0.563 | 0.492 |
|  | $\lambda_{\text {min }}-\sigma_{a}$ | 0.083 | 0.324 | 0.988 | 0.170 | 0.026 |
|  | $\lambda_{\text {min }}-F$ | 0.032 | 0.170 | 0.249 | 0.502 | 0.484 |
|  | $\lambda_{1}$ | 0.063 | 0.241 | 0.321 | 0.564 | 0.450 |
|  | $\lambda_{\text {mean }}$ | 0.080 | 0.268 | 0.343 | 0.627 | 0.314 |
|  | $\lambda_{\text {max }}$ | 0.111 | 0.306 | 0.381 | 0.470 | 0.003 |

