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**An Assessment of White Sturgeon
Stock Status and Trends in the Lower
Fraser River**

**Évaluation de l'état du stock
d'esturgeon blanc et des tendances
dans le bas-Fraser**

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ABSTRACT

We used a variety of information sources to estimate natural and fishing mortality rates and to assess recent trends and current abundance of white sturgeon in the lower Fraser River from Yale to the Strait of Georgia. These sources included:

- 1) estimates of current population structure and recruitment-mortality rates over the last few decades, from age composition sampling in 1995-99 and size composition sampling since 1999;
- 2) estimates of capture and recapture rates of over 18100 sturgeon tagged with PIT tags since 1999, which provide information on growth, mortality rate, total abundance, and movement;
- 3) estimates of historical harvests since 1880, which can be used to back-calculate how large the stock must have been before fishing and how much it must have declined in order to be near the current size indicated by the PIT tag data.
- 4) indices of recent abundance trend from the gill net test fishery at Albion.

The age and size composition data suggest a stock that has relatively healthy numbers of older fish and is either stable or increasing over time. The PIT tag data indicate that natural mortality plus emigration rates may be as low as 7%/yr or as high as 14%/yr. Higher rate estimates are obtained when the data are analyzed with simple aggregate abundance models while lower rates are indicated by models which recognize that data come from a complex spatial sampling regime with variable marking and capture rates among sections of the lower river. Reconstructions of the stock history from catch data indicate that the stock is probably now at about 10% of its natural annual egg production, and may have been impacted most severely over its history by gill net fishing in the 1960-90 period rather than the pre-1920 period. Catches in the Albion test fishery suggest a very rapid increase in abundance of small fish (ages 1-7) since 2000, and this apparent rapid increase is also evident in the size structure of fish captured for PIT tagging. Although the stock is most likely increasing in abundance, it is also most likely

to be well below its most productive level for harvest management, and its recovery to that level could be severely delayed by allowing any retention fisheries or through increases in by-catch mortality associated with catch and release angling or commercial and first nations gillnet fisheries.

RÉSUMÉ

À partir de diverses sources d'information, nous avons estimé le taux de mortalité naturelle et par pêche et évalué les récentes tendances ainsi que l'abondance actuelle de l'esturgeon blanc dans le cours inférieur du Fraser, entre Yale et le détroit de Georgia.

Ces sources comprennent :

- 5) des estimations de la structure actuelle de la population et des taux de recrutement-mortalité au cours des dernières décennies, à partir de la composition par âge dans les échantillons entre 1995 et 1999 et de la composition selon la taille dans les échantillons depuis 1999;
- 6) des estimations des taux de capture et de recapture de plus de 18 100 esturgeons marqués avec des étiquettes à transpondeur passif intégré (TPI) depuis 1999, fournissant des données sur la croissance, le taux de mortalité, l'abondance totale et les déplacements;
- 7) des estimations de prises historiques, depuis 1880, qui peuvent servir à rétro-calculer l'importance du stock avant la pêche et l'ampleur du déclin pour qu'il arrive à la taille actuelle selon les données fournies par les étiquettes à TPI;
- 8) des indices de la récente tendance de l'abondance évaluée à partir de la pêche expérimentale au filet maillant pratiquée à Albion.

Les données sur la composition par âge et par taille semblent indiquer que le stock compte un nombre relativement sain de poissons âgés et qu'il est soit stable ou qu'il augmente avec le temps. Les données fournies par les étiquettes à TPI montrent que le taux de mortalité naturelle plus l'émigration pourrait se situer entre un minimum de 7 % par année ou un maximum de 14 % par année. On obtient un taux plus élevé lorsqu'on analyse les données avec des modèles simples de l'abondance globale, et un taux plus faible avec des modèles qui tiennent compte du fait que les données proviennent d'un régime d'échantillonnage spatial complexe, comportant des taux variables de capture et de marquage selon la section du cours inférieur du Fraser. La reconstitution de l'historique du stock à partir des données sur les prises montre que le stock est

probablement maintenant à environ 10 % de son taux de ponte naturelle annuel et pourrait avoir plus particulièrement souffert, au cours de son histoire, de la pêche aux filets maillants pratiquée au cours de la période de 1960 à 1990, que de la pêche d'avant les années 1920. Les prises résultant de la pêche expérimentale à Albion semble indiquer une très rapide augmentation de l'abondance des petits poissons (1 à 7 ans) depuis 2000, hausse qui est aussi évidente dans la structure de taille des poissons capturés pour le marquage à TPI. Bien que l'abondance du stock augmente vraisemblablement, il est aussi à peu près certain qu'il est bien en deçà de son niveau le plus productif pour une bonne gestion de l'exploitation, et que son rétablissement pourrait être grandement retardé si l'on autorisait une pêche normale ou si la mortalité accidentelle associée à la pêche de capture et remise à l'eau ou à la pêche commerciale au filet maillant des Premières nations augmentait.

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TABLE OF CONTENTS

ABSTRACT	ii
RÉSUMÉ.....	iv
ACKNOWLEDGEMENTS	vi
TABLE OF CONTENTS	vii
1.0 INTRODUCTION	1
2.0 ASSESSMENT OF CURRENT POPULATION STRUCTURE AND RECENT TRENDS USING LENGTH FREQUENCY DATA.....	3
3.0 ANALYSIS OF THE FRSCS PIT TAG DATA	8
3.1 Cormack-Jolly-Seber estimates of survival rate and abundance of larger fish	8
3.2 Checks for tagging and catch-release mortality.....	14
3.3 Estimation of spatial movement and mortality rate	16
3.4 Estimation of average monthly movement as a Markov process	24
3.5 A simple check on survival rate estimates using tag recaptures over time.....	25
4.1 SRA model structure.....	27
4.2 Historical data sources	29
4.3 Baseline SRA Results	31
4.3.1 <i>The importance of uncertainties about numbers versus weight of fish caught</i>	<i>31</i>
4.3.2 <i>Was the natural stock size all that much larger than the current stock?..</i>	<i>34</i>
4.3.3 <i>Is a strong rebound in recruitment occurring, and if so what is causing it? </i>	<i>35</i>
4.3.4 <i>Is it possible that the population is considerably larger than indicated by mark-recapture data, and is stable rather than growing?</i>	<i>39</i>
4.3.5 <i>How severe are current fishery impacts, especially on stock rebuilding rates?</i>	<i>40</i>
4.4 A Stochastic Approach to Stock Reduction Analysis	45
5.0 DISCUSSION.....	51
6.0 REFERENCES	55

1.0 INTRODUCTION

The Committee on the Status of Endangered Wildlife in Canada (COSEWIC) classified white sturgeon as endangered in November 2003 (COSEWIC 2003). A decision with respect to its legal listing under the Canadian Species-At-Risk Act (SARA) is not expected until August 2006. If this species is classified as endangered throughout its range, then a variety of activities which potentially affect it would require permits under SARA. The procedure for establishing permissible activities includes the completion of an allowable harm assessment, and information on stock status, population trajectory, and estimates of natural and human-induced mortality rates. Fraser River white sturgeon have been segregated into four separate stocks with the most abundance stock being that in the Lower Fraser River between the Strait of Georgia and Hell's Gate. Given the variety of direct and incidental factors that may affect the population in the Lower Fraser River, the development of information in support of an allowable harm assessment was initiated for this stock.

Until recently, there was little information on the population trends and dynamics for the lower Fraser stock. Echols (1995) synthesized historical harvest data and used limited tagging and recapture data to reconstruct the effect of the fishery and estimate sustainable harvest rates. RL&L (2000) conducted a mark-recapture study from 1995-2000 and estimated a population of 976 adult and subadult sturgeon between Yale and Hope and 17,259 fish between Hope to Mission. The relative low number of tags released and the extremely low recapture rates from the RL&L study resulted in wide confidence limits in the population estimates. In addition, no information on fish downstream of Mission was provided. As a result of these data gaps, the Fraser River Sturgeon Conservation Society (FRSCS) received funding from the provincial government to support a volunteer mark-recapture program using PIT tags to provide a precise population estimate for the entire lower Fraser stock (Strait of Georgia to Yale) and to improve information on habitat use, movement, and distribution in this area. Recently published results from this effort (Nelson et al. 2004) suggest there are approximately 60,000 fish between 40 and 220 cm

(as of Feb. 2004) with evidence of an increasing population over the 2000-2004 study period.

One of the principle criteria required in an allowable harm assessment is the estimation of mortality rates, both natural and anthropogenic, and estimation of population trajectories into the future. In this report we have used a variety of information sources to estimate natural and fishing mortality rates, the historical trend in abundance, and the current population size of white sturgeon in the lower Fraser River from Yale to the Strait of Georgia. These sources included:

- 1) estimates of current population structure and recruitment-mortality rates over the last few decades, from age composition sampling in 1995-99 and size composition sampling since 1999;
- 2) estimates of capture and recapture rates of over 18,100 sturgeon tagged with PIT tags since 1999, which provide information on growth, mortality rate, total abundance, and movement;
- 3) estimates of historical harvests since 1880, which can be used to back-calculate how large the stock must have been before fishing and how much it must have declined in order to be near the current size indicated by the PIT tag data; and
- 4) indices of recent abundance trend from gill net test fisheries at Albion.

Section 2.0 provides an assessment of the current population structure and recent trends using the length frequency data. Section 3.0 summarizes the methods and results of various mark-recapture models used to estimate the total apparent mortality rate, population size, and recent recruitment trends. Section 4.0 uses the mortality rate and population estimates in conjunction with historical harvest data to reconstruct the population trend and assess the impacts of historic directed and current incidental harvests. Conclusions and uncertainties from all the analyses are summarized in Section 5.0. Appendix 1 combines information from this analysis with outputs from the deterministic model to address specific management questions.

2.0 ASSESSMENT OF CURRENT POPULATION STRUCTURE AND RECENT TRENDS USING LENGTH FREQUENCY DATA

In a previous analysis, Korman and Walters (2001) concluded from examination of age composition data collected over 1995-99 that the lower Fraser stock was likely stable or growing slowly prior to 2000. Fig. 1 shows a catch curve constructed by converting the observed length frequency distribution of FRSCS tagged fish for 1999-2004 to ages using the Von Bertalanffy growth parameters estimated by Nelson et al. (2004). The length distribution was converted to an age distribution using the following simple procedure. For any length interval L to $L+d$ cm., expected ages a_L and a_{L+d} at the beginning and end of the interval were calculated from the inverse Von Bertalanffy model $a = -1/K \ln(1 - L/L_{inf})$, using $K=0.0231$, $L_{inf}=412$ as estimated by Nelson et al. using a modified Fabens method with the tagging data. Then the number of fish per age (n_a) for the interval was calculated as $n_a = n_L / (a_{L+d} - a_L)$ where n_L is number of fish observed in the length interval.

For fully-recruited ages (age ≥ 20 yrs. forklength ≥ 150 cm), the slope of the $\ln(n_a)$ curve indicates an apparent total mortality rate Z^* of around 0.2. Nelson et al.'s (2004) open population estimates for wider length intervals estimated a somewhat lower apparent Z^* of 0.13 with 95% confidence limits of 0.12-0.14. But in both cases, Z is considerably higher than the range 0.05-0.08 that would be required to maintain a population with a small number of very old (100+ yrs) fish. There are three possible explanations for this discrepancy:

- 1) the lower Fraser stock has suffered higher mortality rates over the past 20-40 yr due to fishing, i.e. $Z^* = F + M$ and average F has been at least 0.12;
- 2) the lower Fraser is a "nursery area" from which there is continued emigration as fish age, at instantaneous annual rates $E = Z^* - (F + M)$ averaging 0.1 or higher;
- 3) recruitment to the stock has been increasing over time for the past 20-40 yr at an instantaneous rate of $Z^* - (F + M)$ that might have been as high as 0.1 yr^{-1} , depending on the actual $F + M$.

Sampled number of sturgeon per year of age, 1999-2004
Apparent $Z=0.19-0.21$

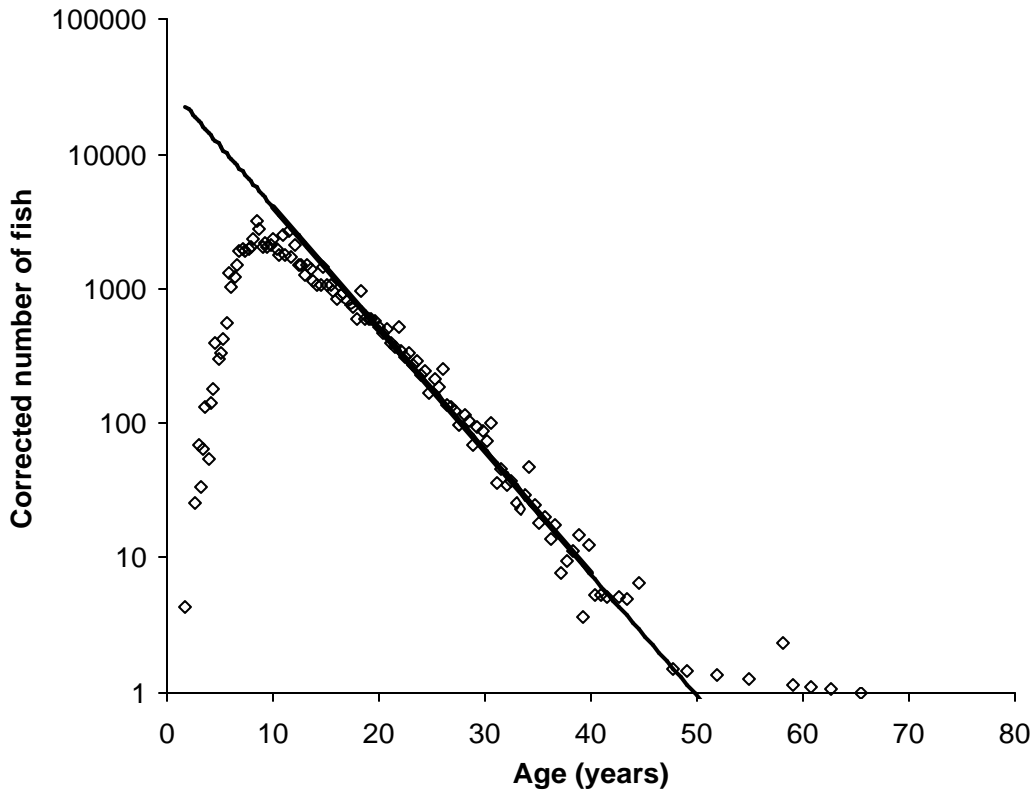


Figure 1. Number of fish by age category with the best-fit mortality rate estimate that explains the decrease in the abundance with increasing age.

To distinguish among these hypotheses, we obviously need independent estimates of the net apparent mortality rate ($F+M+E$) and the fishing mortality rate. The former can be estimated directly from tagging data while the latter can be estimated by stock reduction models.

The data in Fig. 1 can also be used to estimate age-size selectivity in capture probabilities, and to provide correction factors for growth effects on such capture

probabilities in analyses of survival rates for the release cohorts created by tagging at various times and places. If we fit a log-linear model to the n_a data for older “fully vulnerable” fish, this model can be used to back extrapolate the likely number of fish present in the population for younger ages. Then dividing the sampled number of younger fish by this likely number gives a simple estimate of v_a , the average relative vulnerability of age a fish to the tagging capture process ($v_a=1$ for fully vulnerable fish). As shown in Fig. 2, the v_a estimated by this procedure has a logistic shape that is very common for vulnerability curves in fisheries assessment, and fish apparently have increasing vulnerability over a wide range of ages (likely due to increases with length in average swimming speeds and encounter rates with gear, combined with length-dependent ability of fish to attack baits successfully).

Relative vulnerabilities of sturgeon as indicated by average length composition of 1999-2004 tagged fish

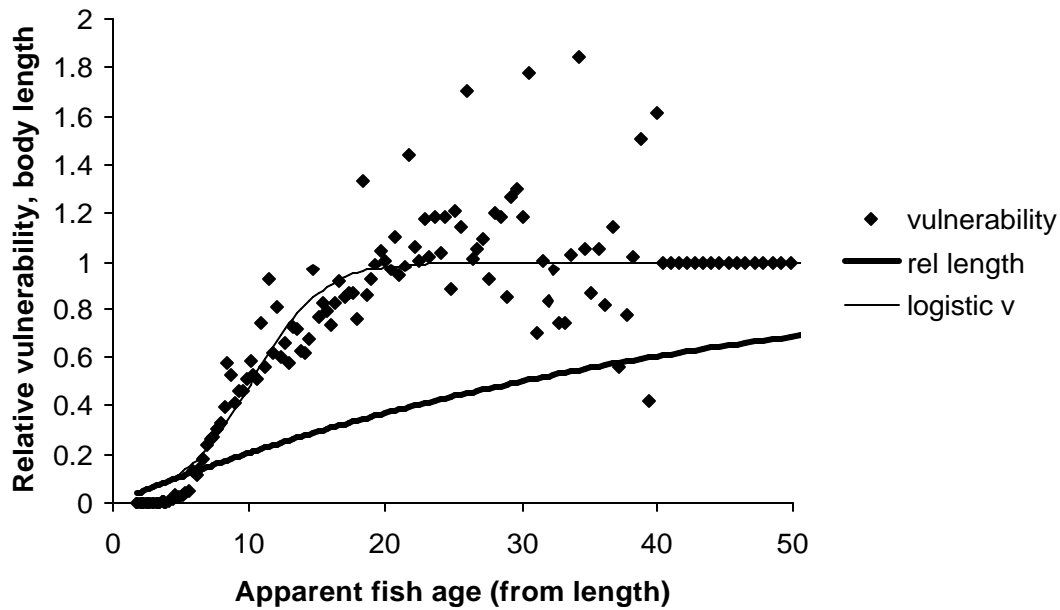


Figure 2. Relative vulnerability as a function of age showing the best-fit relationship (logistic V) and relative length (mean forklength by age divided by asymptotic length).

If we average the logistic vulnerability relationship in Fig. 2 over the sample age distribution (weighting each v_a by sample numbers at age n_a), we estimate the average vulnerability over the sampled ages to be 0.57. Using the same logistic relationship to predict v_{a+1} , we can predict how much the average vulnerability should have increased per year, for at least the first few years after tagging, for any tagged cohort that had the same age distribution as Fig. 1 at the time of tagging. That annual increase in mean vulnerability should have been about 0.063 yr^{-1} . We have used this correction factor in all analyses of the tag recapture data, to account for effects on capture probabilities of growth over time. It would be possible to calculate cohort-specific correction factors by using the observed length frequency distribution of the fish in each cohort at the time of tagging, but this refinement would have little effect on survival estimates since most tagged cohorts used in the analysis had a fairly representative size distribution.

Note in Fig. 1 that there is no clear evidence of an apparent increase in mortality rate (steepness of the catch curve) for fish older than around 30. Those fish represent cohorts that were 5 yrs old and older before 1980, and so would have been large enough to be intercepted in historical gill net fisheries in DFO statistical area 29D (New Westminster, so-called “dogpatch” area where Chinook salmon were targeted before 1980) near the core area of sturgeon distribution, as well as gill net fisheries that have continued further downstream.

There is an important caveat to this analysis. The estimates of apparent mortality rate Z^* depend critically on the von Bertalanffy parameter estimates used to estimate age composition from length composition, and much lower Z^* estimates (and hence lower population growth rates) are indicated by some reasonable growth parameter estimates. Other methods of analysis, including an alternative method based on regressing growth rate from the tags on length at tagging, and growth parameters estimated directly from aged fish collected in 1995-1999 by RL&L (2000), indicate somewhat higher K values and lower L_{inf} values:

Method	K	L_{inf}
Nelson et al (2004)-Fabens method	0.0231	412.8
Screened Fabens	0.0185	487.4
RL&L age data	0.032	228.0
?L/?t vs L regression	0.024	368.0

While all of these methods predict similar growth rates for young fish, they predict highly variable asymptotic lengths (the range of sizes-ages sampled does not include many fish near the growth curve asymptote). When used to convert length to age, the high K-low L_{inf} parameter combinations result in assigning more fish to older ages, and hence to considerably lower Z* values. For example, the RL&L age composition sample and K, L_{inf} parameters result in Z*=0.1, i.e. a population that is either stable or growing only very slowly. The calculated age composition from ?L/?t vs L regression estimates results in Z*=0.15-0.17 depending on the age range assumed to have equal vulnerability to fishing.

3.0 ANALYSIS OF THE FRSCS PIT TAG DATA

Mark-recapture data cannot be analyzed at all without making some assumptions about representative sampling and recapture processes and similarity among fish in vulnerability to capture. But it is never clear just what assumptions are reasonable; there is a precision-accuracy tradeoff between trying to account for (and avoid bias due to) more causes of variation (e.g. fish size, season, location of tagging) by trying to estimate more “local” parameters from the data, versus obtaining more precise but possibly biased estimates by pretending that only a few parameters were important. There are statistical criteria for selecting “best” models (e.g. Akaike Information Criterion), but these can be very misleading when the assumed error structure is incorrect.

Our approach in the following analysis is to simply present results from a variety of statistical models that aggregate the data in various ways (over fish sizes, time, and place). Our aim is to demonstrate that the estimates for a few key parameters, particularly total mortality (plus emigration) rates, are quite robust to alternative assumptions about how far it is safe to go in aggregating the data.

3.1 Cormack-Jolly-Seber estimates of survival rate and abundance of larger fish

To obtain an overview of survival and abundance trends from the mark-recapture data, we first developed spatially aggregated “Cormack-Jolly-Seber” models (Seber 1982). These models compare observed and predicted recapture sequences for the release cohorts of tagged fish created by tagging over time, with the aim of estimating both the survival rates and time-specific estimates of the “capture probability”, i.e. the proportion of fish that were at risk at any time that were actually captured. These capture probabilities can then be used to estimate the number of unmarked fish present at each time, basically as the number of unmarked fish caught divided by the capture probability.

Suppose that marking in month (or year) tt created a cohort with initial number of fish T_{tt} , and that over subsequent months (or years) tr , numbers of recaptures $r_{tt,tr}$ were made of these fish. Assuming constant monthly (or annual) survival rate S , the expected number of recaptures of these fish is given by

$$\hat{r}_{tt,tr} = P_{tr} T_{tt} S^{tr-tt} V(tr-tt) \quad (1)$$

where P_{tr} is the probability that any fish alive at time tr will be captured (at least once) during that sampling period, and $V(tr-tt)=1+0.063(tr-tt)/12$ is a correction for growth in vulnerability of the fish over time (see Section 2.0 text and Fig. 2; omit /12 if annual data). If we assume that the observed number of recaptures for each tag cohort in period tr represents a sample from a Poisson distribution with mean $\hat{r}_{tt,tr}$, then the log-likelihood for the recapture data can be represented simply (ignoring terms that do not depend on parameters) as

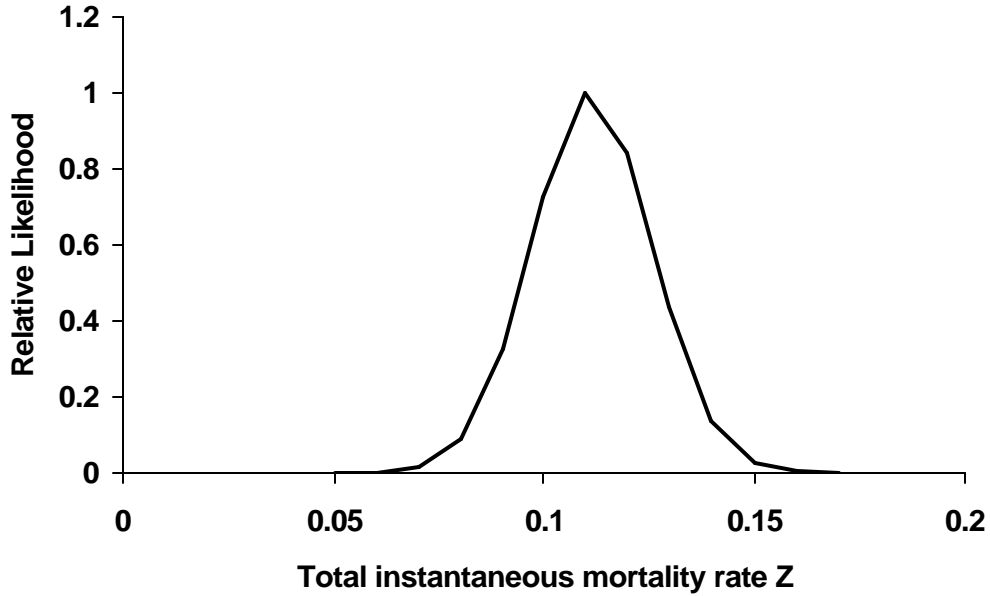
$$L = \sum_{tt} \sum_{tr>tt} [-\hat{r}_{tt,tr} + r_{tt,tr} \ln(\hat{r}_{tt,tr})] \quad (2).$$

It is easily seen by differentiating this likelihood function with respect to P_{tr} that conditional (on S and V) maximum likelihood estimates of the capture probabilities are given by

$$\hat{P}_{tr} = \frac{\sum_{tt<tr} r_{tt,tr}}{\sum_{tt<tr} T_{tt} S^{tr-tt} V(tr-tt)} \quad (3).$$

Using these conditional estimates, estimation of S then becomes a simple one-variable nonlinear search problem: maximize L by varying S , while calculating the predicted r 's using the P 's from (3). Note that the denominator of (3) is a conditional (on S) estimate of the total number of marked fish present at time tr .

Likelihood for average annual mortality rate Z, Jolly-Seber estimation for 60+cm fish, monthly data



Apparent total population of 60+ cm sturgeon, using Jolly-Seber estimation with size vulnerability correction

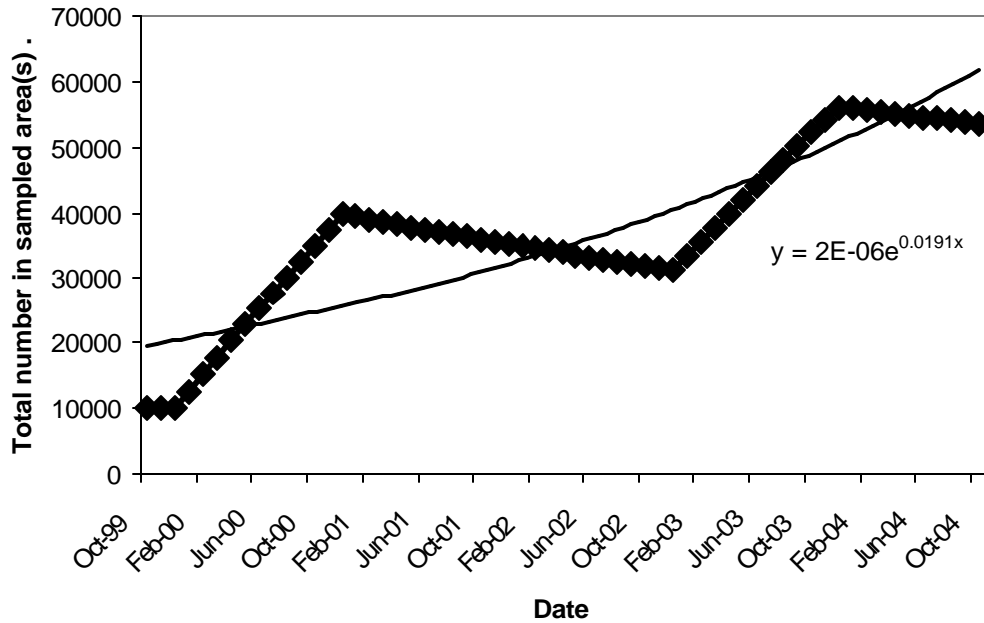
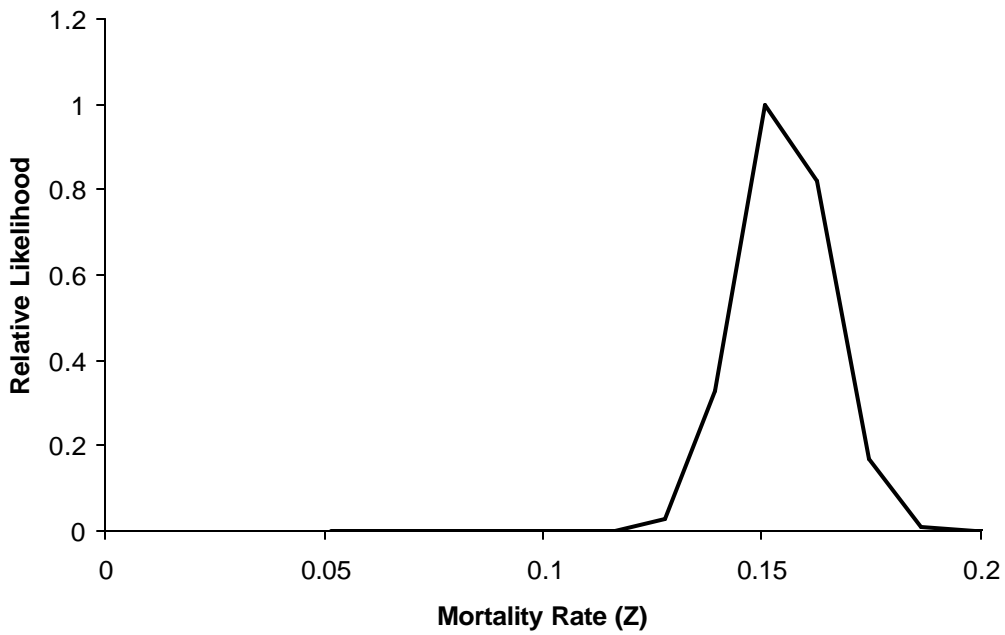


Figure 3. Likelihood profile (top) of the total mortality rate and population trend (bottom) based on monthly aggregation of mark-recapture data.

**Likelihood profile for Z, using annual Jolly-Seber
model corrected for growth effect on vulnerability, data
for 60+ cm and larger fish only**



**Apparent total 60cm+ sturgeon population, from
annual Jolly-Seber estimation with correction for size
vulnerability**

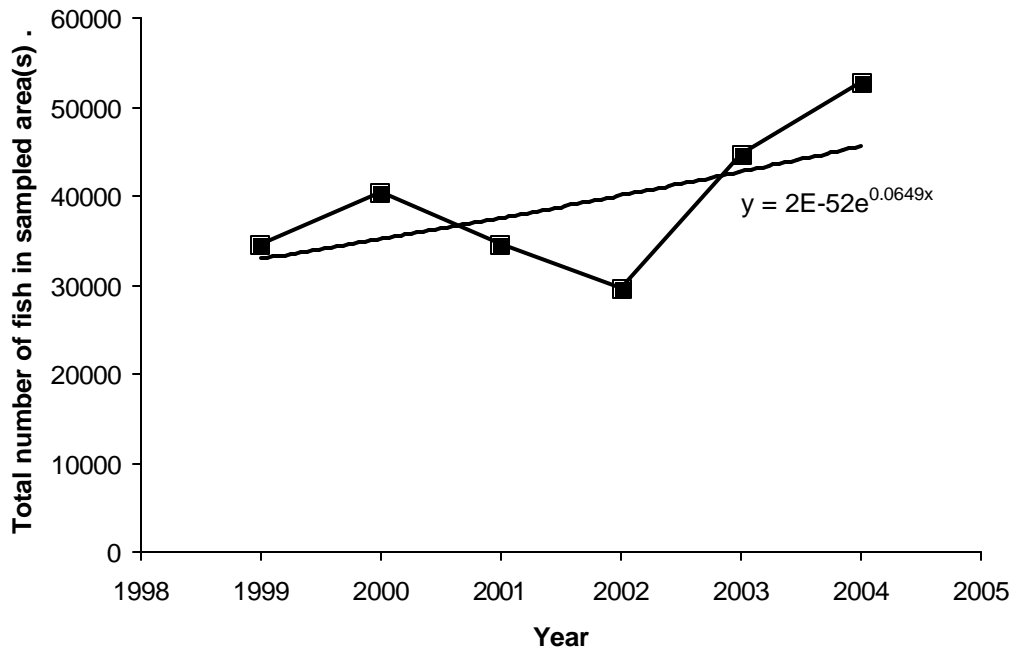


Figure 4. Likelihood profile (top) of the total apparent mortality rate and the population trend (bottom) based on annual aggregation of mark-recapture data.

Likelihood profiles for S using this Jolly-Seber method are shown in Fig. 3-4, for monthly and annual aggregation of the data, and monthly capture probability estimates are shown in Fig. 5. Both methods indicate that $Z=F+M+E$ ($Z=-\ln(S)$) is most likely in the range 0.11-0.15, i.e., total mortality rate due to natural causes, fishing, and emigration is apparently lower than the decay rate Z^* evident from size distribution data. On the basis of these estimates, we would conclude that the population has most likely been growing at a rate of 5%-10% per year over the last few decades.

Monthly capture probabilities for 60+cm fish, Jolly-Seber model with growth-corrected vulnerabilities

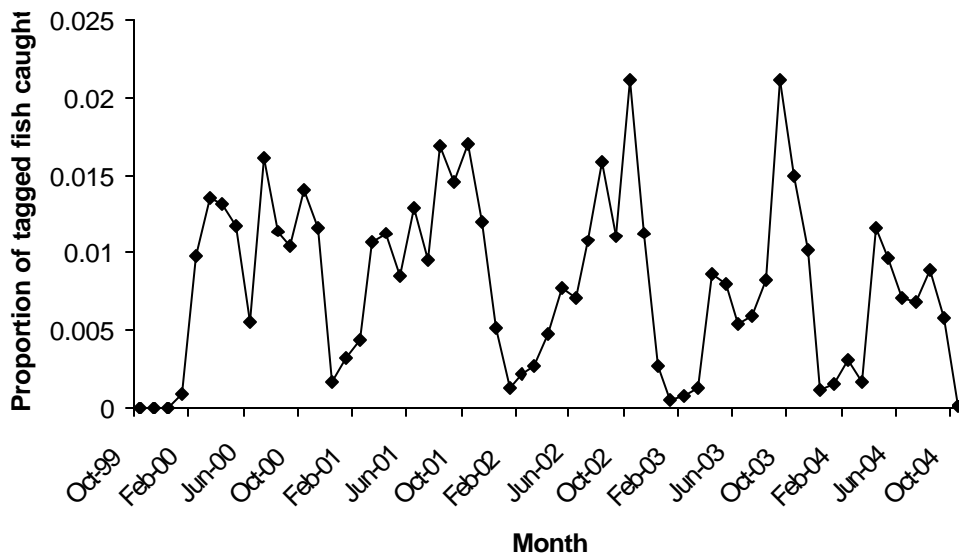


Figure 5. Maximum likelihood capture probabilities conditional on estimates of survival rate and vulnerability.

While the capture probabilities from the survival estimation can be used directly in conjunction with numbers of unmarked fish u_r captured by month to estimate total population size over time, the resulting estimates are very noisy. Better estimates can be obtained by assuming that total unmarked numbers U_{tr} varied as

$$U_{tr} = S(U_{tr-1} - M_{tr-1}) + R_{tr} \quad (4)$$

Where M_{tr-1} is number of fish marked in month $tr-1$ and R_{tr} is a monthly (or annual) recruitment rate; in the monthly case, we have assumed $R_{tr} = R_{year}/12$, and have only tried to estimate annual recruitments R_{year} for years 2000-2004. Note that when we use data for 60+cm fish only, R represents the number of fish reaching this size, not the number of younger fish recruited to the overall sturgeon population. Just as for marked fish, we have assumed that the unmarked fish caught in month tr were sampled from a Poisson distribution with expected capture rate

$$\hat{u}_{tr} = P_{tr} U_{tr} \quad (5).$$

These predictions and u observations can be included in the likelihood function (3) as additional, independent observations with likelihood terms $-\hat{u}_{tr} + u_{tr} \ln(\hat{u}_{tr})$. The P_{tr} estimate is modified to include u_{tr} catches in the numerator, and U_{tr} numbers at risk in the denominator. Parameter estimation proceeds by maximizing the extended likelihood function with respect to the parameter set $(S, U_0, R_{2000}, R_{2001}, R_{2002}, R_{2003}, R_{2004})$.

This extended estimation results in essentially the same estimates for S (unmarked fish carry no information about S when recruitments are assumed variable), and in the estimates shown in Fig. 4 for total abundance of 60+cm fish (marked plus unmarked, marked from denominator of eq. 3) over time. Both annual and monthly analysis indicate that the population has been increasing since 2000, and the abundance estimates agree well with those obtained for larger fish by Nelson et al. (2004) using an open population mark-recapture model with assumed $S=0.9$.

The annual model results in Fig. 4 indicate (weakly, with such few population estimates) an annual growth rate of $6.5\% \text{ yr}^{-1}$ for the population of 60+cm fish, while the monthly model results indicate a slightly higher growth rate of $12\% \text{ yr}^{-1}$. These estimates of population growth rate are well in line with the expectation of around $10\% \text{ yr}^{-1}$ from the

length frequency analysis combined with the Z estimates above, and certainly support the hypothesis that Z^* in the length frequency data is high because of population growth rather than high total mortality rate.

An independent check on the recruitment estimates from the Cormack-Jolly-Seber models is to compare their estimates of R/N , annual new recruits per 60+ cm (age 6-7+) fish, to the annual recruitment per fish evident from the age distribution in Fig. 1. The mark-recapture models give annual R/N estimates of 0.19-0.22. From the data in Fig. 1 and the growth rate estimates of Nelson et al. (2004), the length-frequency data indicate $R/N=0.20$, i.e. the two methods for estimating recruitment rate agree remarkably well.

3.2 Checks for tagging and catch-release mortality

It is not generally possible to estimate mortality caused by the tagging process in Jolly-Seber type models (any reduction in recapture rates caused by such mortality can be equally well explained by lower capture probabilities), unless the capture probability estimates are constrained by using other information such as fishing effort and/or capture rates of unmarked fish in conjunction with estimates of the total numbers of unmarked fish at risk to capture. When a survival rate through tagging is included in the eq. 1 prediction of expected recaptures, and the eq. 3 estimate of capture probability includes recaptures and numbers of unmarked fish, the maximum likelihood estimate of that survival rate is 1.0, i.e. the estimate has to be constrained not to exceed 100% survival.

If significant mortality is caused by each catch-release event subsequent to initial tagging, we would expect the observed number of multiple recaptures to be less than predicted from the overall survival and capture probability estimates. We can predict the number of fish that should have been caught two or more times from eq. 1, simply by substituting the observed number of recaptures for each time period (r_{tt}) in place of the number initially tagged (T_{tt}); the predicted number of re-recaptures of these fish (predicted number caught more than twice) can then be substituted into eq. 1 to predict the number

of fish that should have been caught three or more times, and so forth. For the monthly Cormack-Jolly-Seber model, this method results in the following recapture pattern for the 16,187 60+ cm fish that have been tagged since 1999:

Recaptures:	Observed	Predicted
One or more times	2493	3131
Two or more times	408	426
Three or more times	61	43.

These results indicate that if anything, fish that have been caught once are more prone to recapture. There is certainly no direct evidence of cumulative mortality due to repeated catch and release associated with the recreational fishery.

A warning about these results is that statistical analysis of data from a mark-recapture study can never be used to convincingly demonstrate tagging or recapture mortality. Any unfavorable result (indicating high mortality) can always be plausibly explained as a recapture avoidance effect, i.e. capture events may trigger dispersal out of fishing areas or avoidance of gear. Any favorable result can likewise be explained by heterogeneity in capture probabilities, i.e. concentration of fishing in areas where there are more tagged fish and/or increased attraction to gear because of bait reward, etc. Such problems can be partly avoided by using different gears for marking and recapture, but ultimately the best way to measure mortality rates caused by capture is to track individual fish following release, using radio or acoustic tags.

3.3 Estimation of spatial movement and mortality rate

Analysis of spatially aggregated mark-recapture data for estimation of survival and abundance is always somewhat dangerous, and can be subject to bias caused by shifts in the distribution of marking and recapture effort over time. In the Fraser case, most early tagging (1999) was in statistical area 8 near Mission (see Fig. 3 of Nelson et al. 2004), and effort later spread out to cover other areas more evenly. Such spreading, and even seasonal concentration of effort in particularly productive fishing locations, can cause a negative bias in survival estimates (survival estimates from spatially restricted sample period are low because tagged fish moving out of area appear as higher mortality) and apparent upward trends in population size (essentially, the study population increases as fish from more areas become subject to marking). For these reasons, we attempted a spatial mark-recapture analysis.

Suppose tagged fish are classified into tag cohorts of $T_{tt,it}$ individuals each, where tt is the month of tagging ($1=Oct\ 99$) and “ it ” is the area of tagging (“ it ” might represent the 14 sturgeon statistical zones, or some coarser aggregation of those zones). The expected number of recaptures of these fish at any later recapture month tr , in recapture areas ir , can be represented assuming constant monthly survival rate S as

$$\hat{r}_{tt,tr,it,ir} = p_{tt,tr,it,ir} T_{tt,it} S^{tr-tt} P_{tr,ir} V(tr-tt) \quad (6)$$

where $p_{tt,tr,it,ir}$ is the proportion of fish from the (tt,it) tag cohort that are in area ir in month tr , $P_{tr,ir}$ is the capture probability for fish in that area in month tr , and $V(tr-tt) = 1 + 0.06(tt-tr)/12$ is a linear correction for average growth over time in vulnerability to capture (see length frequency analysis and Figs. 1, 2). Assuming that actual recaptures $r_{tt,tr,it,ir}$ are distributed as a Poisson with mean given by $\hat{r}_{tt,tr,it,ir}$ from (6), conditional maximum likelihood estimates of the capture probabilities are given by

$$\hat{P}_{tr,ir} = \frac{\sum_{tt < tr} \sum_{it} r_{tt,tr,it,ir}}{\sum_{tt < tr} \sum_{it} P_{tt,tr,it,ir} T_{tt,it} S^{tr-tt} V(tt - tr)} \quad (7)$$

The numerator here is just the observed total number of recaptures in month tr in area ir , and the denominator is the expected total number of fish at risk to recapture, summed over all tag cohorts that were created before month tr , in all areas, corrected for vulnerability and survival and in proportions assigned to area ir using $p_{tt,tr,it,ir}$. Likewise, conditional maximum likelihood estimates of the location proportions are given under Poisson sampling by

$$\hat{p}_{tt,tr,it,ir} = r_{tt,tr,it,ir} / [T_{tt,it} S^{tr-tt} \hat{P}_{tr,ir} V(tr-tt)] \quad (8).$$

A simple way to think of this calculation is as $\hat{p} = (r / \hat{P}) / N$, where r / \hat{P} is an estimate of the number of fish present in the recapture area and N is the number at risk to recapture (corrected for growth in vulnerability). In order that S be estimable, the p 's must be constrained or averaged in some way, e.g. by requiring the sum of p 's over recovery areas ir to be 1.0.

Equations (7) and (8) are a large, interdependent equation system, with estimates of the p 's depending on the P 's and vice versa. This equation system can be solved efficiently by a simple relaxation procedure: (1) provide an initial estimate of the p 's by assuming all fish are recaptured only where they were tagged ($p_{tt,tr,it,ir} = 1$ for $ir = it$, 0 otherwise), and solve eq. 7 with these estimates; (2) use the resulting estimates of $P_{tr,ir}$ to solve eq. (8) for the p 's, and renormalize the resulting estimates to sum to 1.0 over recovery areas ir for each tt, tr, it ; (3) return to step (1) and repeat estimation of P 's using the p 's from step (2), continuing this iteration until the p 's stop changing.

This iterative procedure is valid only for time-area aggregations of the data large enough to insure $P_{tr,ir} > 0$ for all recapture times tr and areas ir . For times-areas not fished when the data are analyzed at fine scales, the r/P calculation fails, and the normalization of the

p 's over those areas that were fished gives nonsense estimates of location proportions (fish are just assigned to those areas that happened to be fished). However, even ignoring this problem the procedure appears to give reasonable estimates for Z , somewhat lower than the Cormack-Jolly-Seber methods (Fig. 6). An alternative for analysis of the finer-scale data is to assume some functional structure for p variation, e.g. by season and time since tagging; the p 's are expected to reflect seasonal upstream-downstream movements along with "diffusive" spreading over time.

It has been common in the literature on mark-recapture analysis to represent "movement rates" using markov transition matrices where a vector N_t of tagged fish is "moved" according to $N_{t+1} = MN_t$ where M is a matrix with element M_{ij} representing the probability of an animal that is in area j at time t moving to area i (and surviving) as of time $t+1$. In our view, such transition probabilities are a misrepresentation of what can be learned from tagging data. What the data can tell us is the proportion of area i animals that were in area j as of time $t+1$, i.e. something about the $p_{t,ir,it,ir}$ proportion, but not how much the animals moved between these times. A Fraser sturgeon is quite capable of moving from the river mouth upstream to Hope, then back down to the mouth, in a week or two. Recapturing it back at the mouth a month after tagging does not imply that it stayed there for a month and hence was not exposed to the various risks (like gill net interception) that a moving fish might encounter. We simply cannot assess such movements, and associated risks, with tagging data especially when each fish is recaptured only infrequently.

**Likelihood profile for total mortality rate using
space-time monthly model, 14 statistical areas,
60cm+ fish**

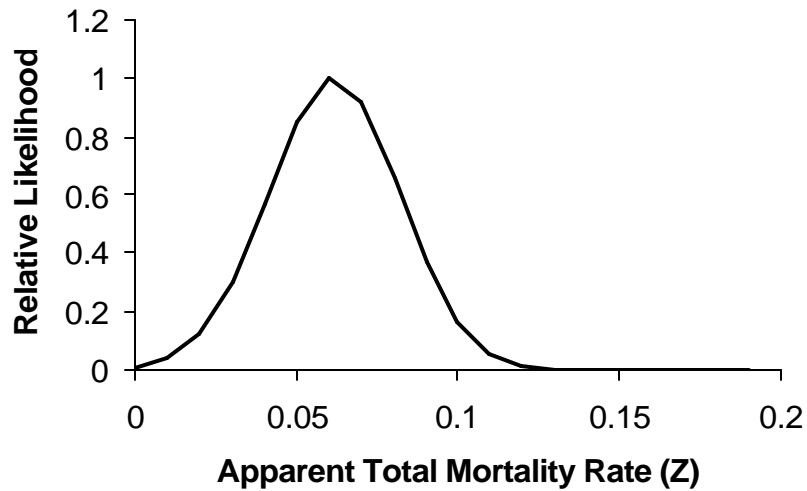


Figure 6. Likelihood profile of the total mortality rate based on spatial mark-recapture model assuming no spatial or seasonal structure in location proportions.

But we certainly should be able to estimate some structure in the $p_{tt, tr, it, ir}$ proportions, reflective at least of strong seasonal movements that might distort population estimates, and use this structure to avoid possible bias due to assuming $p_{tt, tr, it, ir}=0$ for areas that did not happen to be sampled or for which there were few observed recaptures due to limited fishing effort. Any proposed aggregation scheme or structure can be represented as replacing the elementary proportions $p_{tt, tr, it, ir}$ in the likelihood function for the recapture data with a value from some smaller array $p_{tr(tt, tr), ts(tr), it, ir}$ where the tag and recapture times tt, tr are replaced with a time-since-tagging function $tr(tt, tr)$ and a month-of-year function $ts(tr)$ that assumes some interannual stationarity in seasonal movement patterns (e.g. be independent of tt alone, and depend instead only on time since capture $tt-tr$ and the seasonal time of year). Note that the dependence of p on tagging location (it) cannot be omitted, without assuming some instantaneous redistribution of fish among areas independent of that location. For any aggregation choice, the iterative scheme described

above can be used to obtain conditional maximum likelihood estimates of the $p_{tr(tt,tr),ts(tr),it,ir}$, simply by replacing the numerator and denominator of eq. (8) with sums of recaptures and numbers at risk over those tt, tr, it, ir elements that are assumed to have had the same $p_{tr(tt,tr),ts(tr),it,ir}$ value.

Using a location p aggregation that assumes effects of time since tagging last up to 12 months ($tr(tt, tr)$ function equals $\min(tr-tt, 12)$) and monthly variation in distributions ($ts(tr)$ counts months from 1 to 12 then repeats), we obtain a maximum likelihood estimate of $Z=0.08$, remarkably close to the value reported for other sturgeon stocks by Beamesderfer et al. (1995), and the likelihood profile for Z shown in Fig. 7.

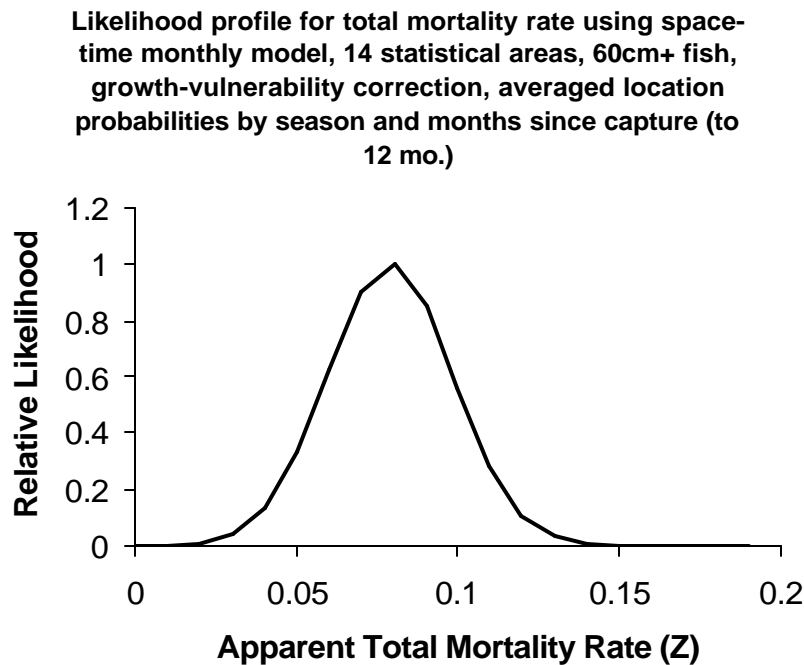


Figure 7. Likelihood profile of the total mortality rate based on the spatial mark-recapture model assuming effects of time since tagging and season on location probabilities.

Estimated equilibrium ($tr > tt + 12$) seasonal distributions assuming $Z=0.08$ are shown for a few major statistical areas in Fig. 8. An obvious feature of these distributions is that long-term location patterns are not strongly dependent on tagging location, i.e. fish

appear to seasonally concentrate in a few statistical areas (particularly area 8) wherever they were originally tagged. However, there is a tendency for fish captured downstream (area S) to be more concentrated in that same area in summer, and for fish captured upstream (area 12) to be more concentrated in area 12 in summer. Further, there is a pattern of apparent seasonal movement, with fish appearing to concentrate in areas 6-8 in mid-winter, and in downstream areas in mid-summer. Fish from area 14 are seldom caught outside that area. In most years, there has been almost no catch except in area 8 during December and January, and the estimation procedure described above assigns zero residence proportion in any case where both $P_{tr,ir}$ and total recaps are zero. For the months with largest total catches (July-October), both catch and estimated residence proportions are widely distributed across statistical areas.

The likelihood profile of the apparent total mortality rate based on the spatial model (Fig.7) is not at all sensitive to which ages-sizes of fish are included in the analysis, nor is it sensitive to spatial aggregation of the statistical areas. To check for such sensitivity, we ran the analysis including all fish (30+cm), and aggregated the statistical areas into 5 zones:

Zone 1: areas O,M,S,3,4

Zone 2: areas 5,6

Zone 3: areas 7,8

Zone 4: areas 10-12

Zone 5: areas 13,14.

These changes caused the maximum likelihood Z to increase slightly (to 0.09), and resulted in somewhat smoother spatial distribution patterns (Fig. 9). Interestingly, spatial aggregation to larger zones revealed more clearly that fish tagged near the ends of the distribution (Zones 1, 5) are more likely to found later in the same zone, especially for the upstream Zone 5. However, the apparent winter concentration in Zone 3 may be purely an artifact of having almost all of the (very low) December-January fishing concentrated in statistical area 8.

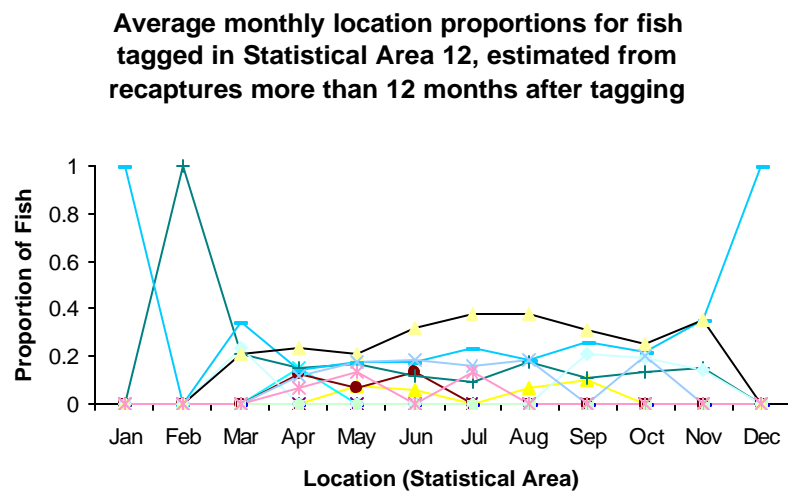
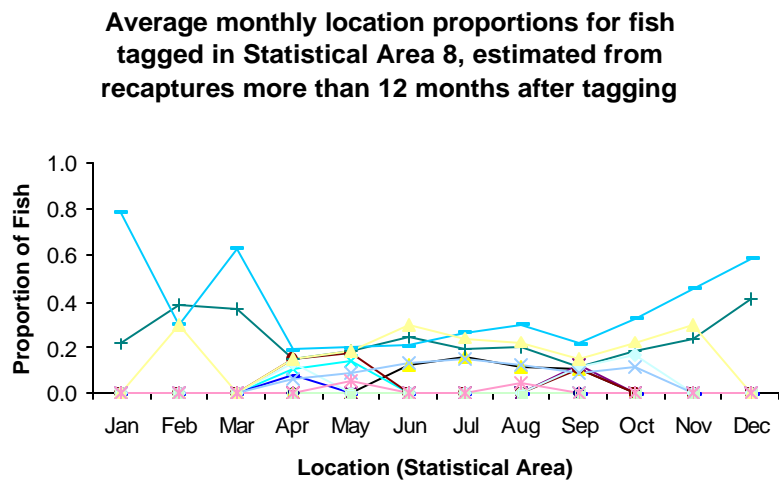
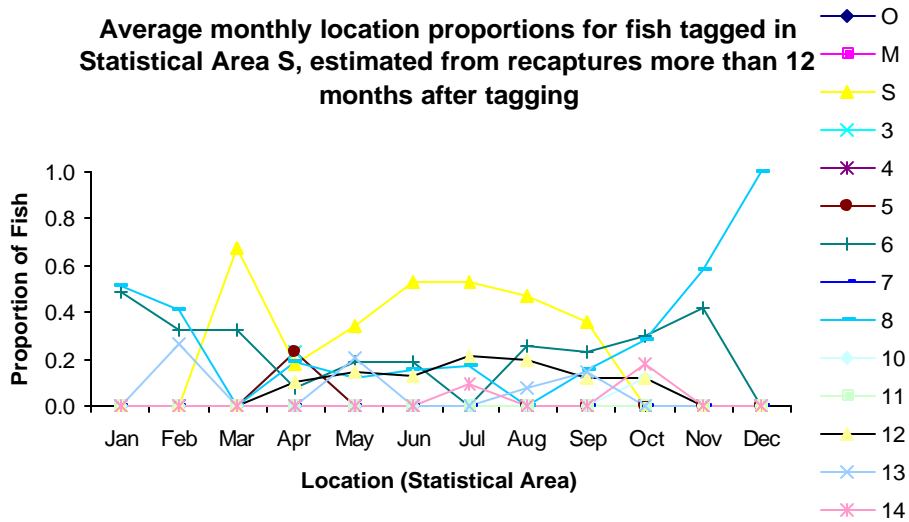


Figure 8. Estimates of the proportion of fish tagged in Area S (top), 8 (middle), and 12 (bottom) fish available for recapture in other locations based on the spatial mark-recapture model assuming effects of time since tagging and season on location probabilities.

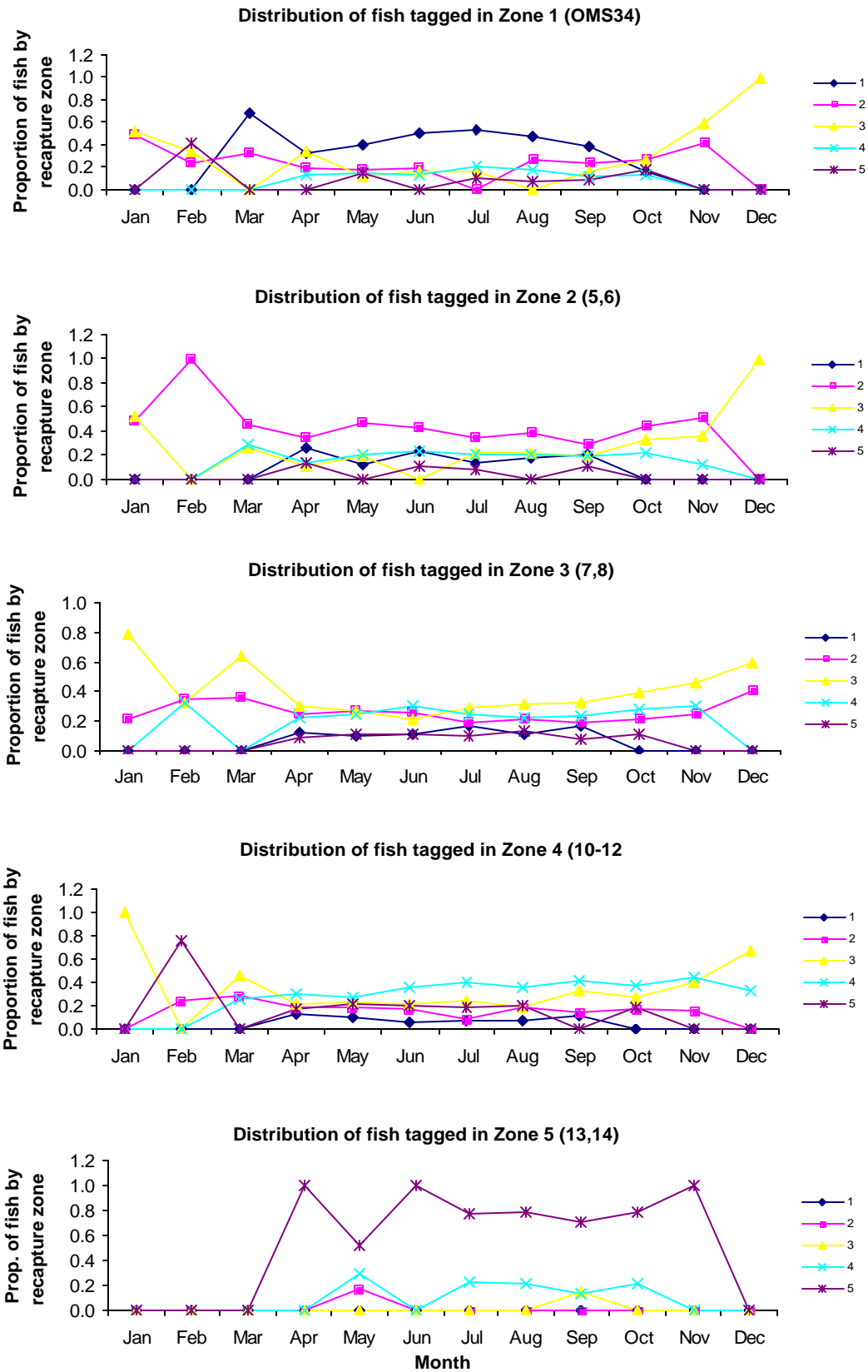


Figure 9. Estimates of the proportion of fish tagged in various locations available for recapture in other locations using an aggregated spatial structure.

3.4 Estimation of average monthly movement as a Markov process

The relative complex model above for estimation of location proportions can be biased due to lack of recapture efforts, especially in winter, in some statistical areas. We therefore decided to also estimate a single Markov movement matrix representing the average proportion of fish that were in area j at time t that moved to area i as of time $t+1$. In terms of the above notation, this is accomplished simply by setting $\hat{p}_{tt,ir}$ in eq. 8 equal to $m_{ir,it,t+1}$, where $m_{ir,it,t+1}$ is the ir,it element of M^t where M is the single Markov movement matrix. This approach resulted in a maximum likelihood estimate $Z=0.071$, and the following M matrix:

		From				
		Zone 1	Zone 2	Zone 3	Zone 4	Zone 5
To	Zone 1	0.928	0.010	0.006	0.002	0.001
	Zone 2	0.029	0.924	0.041	0.010	0.000
	Zone 3	0.035	0.053	0.918	0.059	0.001
	Zone 4	0.006	0.012	0.035	0.918	0.016
	Zone 5	0.001	0.001	0.000	0.011	0.982

These movement proportions are quite reasonable, indicating that each fish in Zones 1-4 has about a 90% chance of staying in the same zone for one month, with the 10% of fish that do move mainly end up in Zones 1-4. As noted for the more complex models, fish from Zone 5 are most likely (98%) to stay in that zone, with the few that move going mainly to zone 4.

As for the other movement models, the assumption of Markov movement gives lower Z estimates than do analyses based on spatially aggregated data. This lends further support to our suspicion that spatial (and seasonal) heterogeneity in recapture effort may tend to cause upward bias in apparent Z values (fish that disperse into lightly fished areas are

more likely to be seen in the aggregated data as apparent mortalities or emigrants from the study region).

3.5 A simple check on survival rate estimates using tag recaptures over time

As a check on the more complex estimation methods described above, we simply plotted the number of recaptures per fish tagged each month, against number of months since tagging. The average of these values over cohorts also has to be standardized by the number of cohorts that were at risk for each possible number of months since tagging. The result is shown in Fig. 10, plotted on a log scale to demonstrate that Z is apparently stable for at least the first four years after tagging, and according to this method averages a bit over 0.12. We suspect that lower Z estimates from the spatial analysis reflect a tendency for fish to disperse more widely over time, with many ending up in areas with less fishing pressure so as to make their numbers of recaptures go down a bit more quickly than the actual death rate.

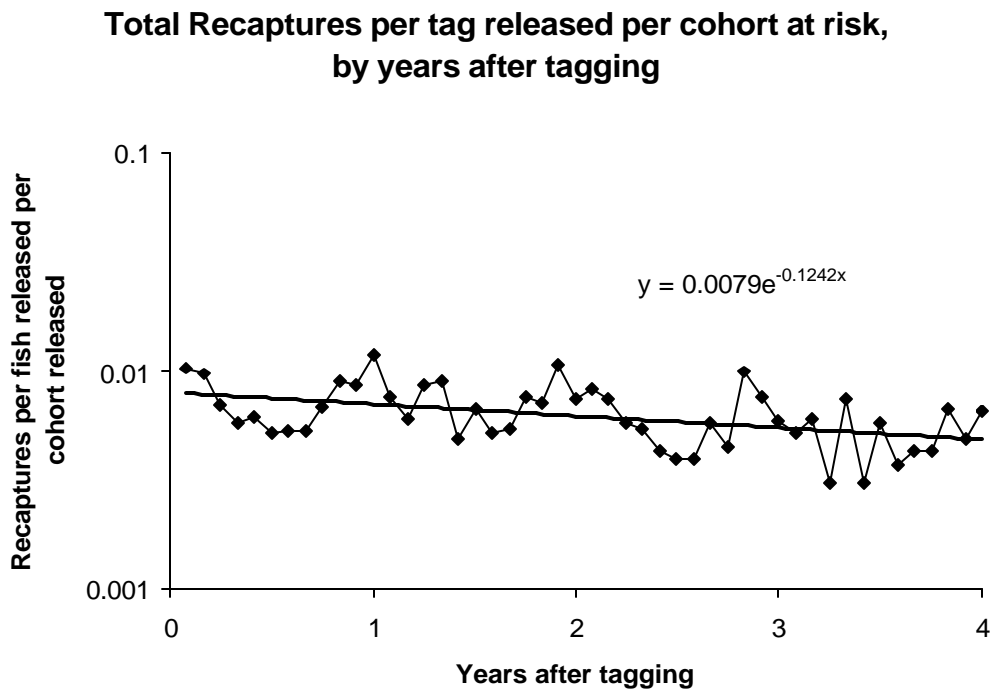


Figure 10. The expected number of recaptures per fish each month as a function of the amount of time since tagging (corrected recaps monthly estimation 60.xls).

4.0 RECONSTRUCTION OF HISTORICAL FISHERY IMPACTS AND STOCK STATUS USING STOCK REDUCTION ANALYSIS

We used the approach called “stock reduction analysis” or SRA (Kimura, et al 1984) to estimate how much the stock has been impacted by historical harvest removals, and how large it must have been before fishing in order to have produced those removals and still be as abundant as the tagging data indicate for recent years. The estimated historical stock sizes provide possible reference points for stock rebuilding plans, and in particular how much larger recruitment rates must have been in the unfished stock than they are today (i.e. has there been recruitment overfishing?).

The essential idea in stock reduction analysis is to build a population dynamics model of the same structure as would be used to interpret recent data and make projections (age structured representation of recruitment, survival, and growth), but to “drive” this model by essentially subtracting historical catches over time. That is, the model simulates forward from a historical reference point near the beginning of the fishery (1880), removing historical catches along the way while adding and subtracting estimates of recruitment and mortality, so as to produce a cumulative prediction of current (2003) stock status. The recruitment estimates in particular are then adjusted until the simulations produce current numbers near those estimated from tagging or other recent assessment method; when the model is provided recruitment estimates that are too low to have sustained historical catches, it predicts that the population should have disappeared long ago. When the model is provided recruitments that are too high, it predicts too little historical fishing impact and a current population size that is too large. So in essence we “fit” the reconstruction model to historical data just by adjusting its recruitment parameters so as to give a simulated stock history that both incorporates historical fishing impacts and ends up near the current stock size.

Formally, we “fit” the SRA model by varying assumed natural recruitment rate or unfished abundance so as to maximize the likelihood of the 1999-2004 mark-recapture data (for 60cm+ fish). That is, the SRA model predicts total abundance for each year, and from these abundances and calculated abundances of marked fish (adjusted for marks

added and survival rate from year to year), we predict the number of marked and unmarked fish that should have been captured each year, and compare these to the observed numbers using the same Poisson likelihood function as used for the mark-recapture analyses.

4.1 SRA model structure

We developed a standard age-structured population accounting model in Excel (based on Korman and Walters 2001) with 100 ages. Numbers were assumed to change over age a and years t according to the survival equation

$$N_{a+1,t+1} = N_{a,t} S(1 - v_{a,t} U_t) \quad a=1 \dots 100, t=1880-2004 \quad (9)$$

where $S = e^{-M}$ is the natural survival rate (mortality rate $M=0.08$ which is based on the spatially aggregated model that accounts for effects of movement on mortality rate estimates, Fig. 7)), $v_{a,t}$ is the relative vulnerability of an age a fish in year t to fishing, and U_t is an estimate of the exploitation rate on fully vulnerable fish ($v_a=1$) in year t . To run this model, we needed to estimate or predict S (from tagging), $v_{a,t}$ from analysis of harvest information, the U_t time series, and recruitment rates $N_{1,t}$ for all years after 1880.

For years prior to 1900, when targeted commercial fishing used a variety of gears to take sturgeon of a wide range of sizes, we assumed the same $v_{a,t}$ schedule as estimated for recent recreational harvest sampling by volunteer PIT taggers (see Figure 2). Since 1920, the bulk of the catch has been gill-net bycatch, which selectively takes a narrower range of sturgeon sizes mostly between 40 and 90 cm. Using the size data from gill net catches in the FRSCS database and assuming a vulnerability schedule shaped like a normal distribution, we estimated a second $v_{a,t}$ schedule for the post 1920 period. For the 1900-1920 period, we assumed a smooth (linear) shift between the two vulnerability schedules, from 0 to 90% gill net type.

For each year, the overall exploitation rate U_t was calculated from historical total catches C_t (in kg) as

$$U_t = C_t / [\sum N_{a,t} v_{a,t} w_a] \quad \text{except } U_t < 1.0 \quad (10)$$

where w_a is the average weight of an age 'a' sturgeon. The denominator in this equation is the predicted total vulnerable biomass in year t, summed over all ages a in that year. Note that a good indication of bad $N_{a,t}$ estimates is to have this calculation predict some $U_t > 1.0$, i.e. historical catches larger than the biomass of fish predicted to have been available.

Recruitment rates $N_{1,t}$ were predicted from estimates of annual egg production $E_t = \sum N_{a,t} f_a$ (f_a = relative fecundity at age a) using a Beverton-Holt recruitment relationship. The Beverton-Holt relationship is almost universally seen in empirical fish stock-recruitment relationships (now around 300 cases; see Myers and Barrowman 1996; Myers et al. 1999), for reasons reviewed in Walters and Martell (2004). We used the compensation ratio or a form of the relationship,

$$N_{1,t} = \frac{K(R_0 / E_0)E_t}{1 + (\frac{K-1}{E_0})E_t} \quad (11)$$

where R_0 is an unfished recruitment rate to be estimated, E_0 is average annual egg production for an unfished population with recruitment rate R_0 , and K is the Goodyear compensation ratio (Myers a) representing the steepness of the stock recruitment curve at low stock sizes as the ratio of juvenile survival rate at low stock to juvenile survival rate in the unfished population. We assumed $K=5$, which is roughly the average value reported for a wide range of populations by Myers et al. (1999). Sensitivity of the historical reconstructions to various (R_0, K) possibilities was explored by varying K from 2.0 (near the minimum observed in other populations) to 1000 and examining how this affected the R_0 estimates. In the end, we found that uncertainty about K is apparently far

less important in interpreting the historical data than uncertainty in effective fecundities at age f_a (see section below on recruitment rebound) and the natural mortality rate M .

4.2 Historical data sources

Semakula and Larkin (1968) provide reconstructions of annual historic commercial catches from 1880-1963. Echols (1995) updated this record to the end of the commercial fishery in 1991. Commercial landings of sturgeon from the Fraser River were only recorded as total weight. Green and white sturgeon were both caught in the fishery but the landings were not recorded by species so the biomass of white sturgeon is overestimated to an unknown extent. As in Echols, we assumed that harvests from poaching was 50% of the total legal harvest. Echols also compiled estimates of sturgeon harvested by gill nets in First Nation (FN) fisheries in the Lower Fraser River from 1983-1993 prior to the moratorium on retention in food and recreational fisheries implemented in 1994. FN harvests were estimated by DFO as the product of the average number of fish caught in a sub sample of sets times the total number of nets observed. Harvest may be overestimated because not all fish caught were necessarily retained. Catches were multiplied by an assumed average weight in the gillnet catch of 3.3 kg to estimate the FN harvest biomass. We assumed that the harvest prior to 1983 was equal to the average harvest between 1983 and 1993. Recreational harvest was estimated based on the angling catch records from 1965-1994 summarized in Echols (1995). The number of fish caught per year was multiplied by an average weight in the recreational catch of 25 kg to determine the total recreational harvested biomass. Prior to 1964 we assumed that 300 fish were killed per year as in Echols.

A critical problem in using these catch estimates for SRA is that the commercial data, which form the vast majority of the catch, are reported only in weight units, so we must rely upon the model calculations of changes in average body size (see eq. 10) to essentially “convert” the biomass harvest to numbers of fish killed over time. These changes are of two types: (1) decreases in mean body size in the field population due to

cumulative removal of older, larger fish; and (2) effects of gear mix (eg hook and line vs gill nets) on the vulnerability schedules $v_{a,t}$. Based on growth and survival schedules, we would for example expect the current recreational vulnerability schedule to result in a mean weight of fish caught of around 35 kg (Beamesderfer et al., 1995), if the unfished stock structure (much higher proportions of larger, older fish) were available to that fishery; even with the present “eroded” age structure, the average weight of fish PIT tagged in 1999-2004 that were caught by angling has been 25 kg (based on mode of length frequency converted to weight using weight-length relationship from RL&L 2000). In contrast, the average weight of fish caught by gill nets is around 3.3 kg based on mode of length frequency and weight-length relationship from RL&L 2000). That is, the average body weight of fish in the catch may well have declined by a factor of more than 10, implying that the early historical fishery may only have been taking the same number of fish per year as the recent (1950-80) gill net fishery, even when it was taking tonnages 10 or more times as high (Fig. 11).

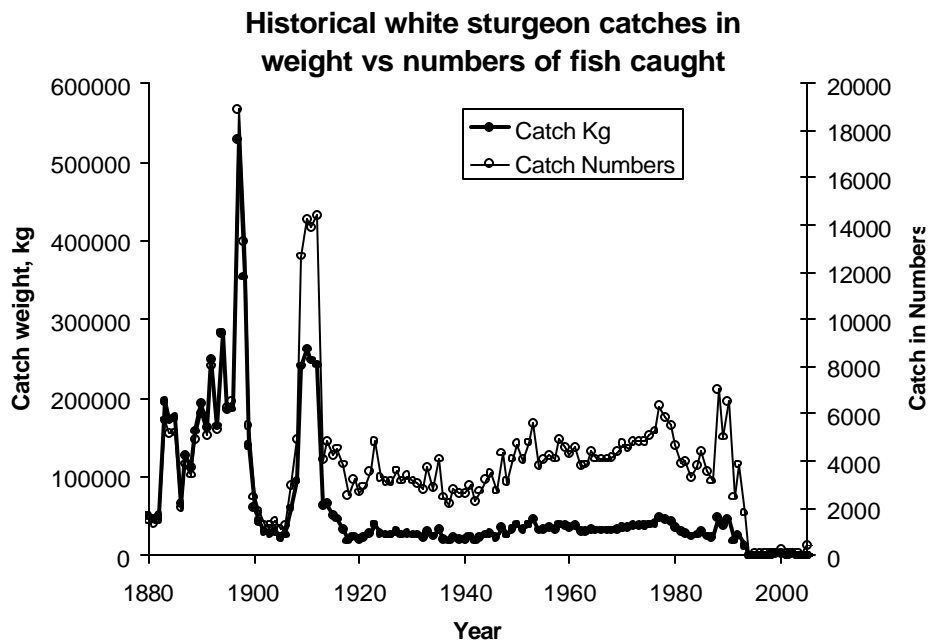


Figure 11. Historical catches of white sturgeon in the Fraser River. Catch weights from historical reports (Semakula and Larkin 1968, Echols 1995). Catch numbers converted from weight by using estimates from population models of average weight of fish vulnerable to gears used at different times. Pre 1920 catches were of much larger fish (20+ kg average) than afterward when the majority of the catch was taken by gill nets with a narrower size selectivity (3.3 kg average).

4.3 Baseline SRA Results

A baseline estimate of the lower Fraser abundance history from SRA is shown in Fig. 12, using the default survival and vulnerability parameter estimated in the length-frequency and mark-recapture components of this study, respectively. This history is very similar to that estimated using a simpler (delay-difference) model by Echols (1995). For this estimate, the Beverton-Holt R_0 (unfished natural recruitment, number of 1 yr olds per year) was varied so as to give a maximum likelihood estimate of current vulnerable abundance (to recreational fishing) of 40000-60000 fish, and recent trends similar to those obtained from mark-recapture analysis but smoother over time due to the Beverton-Holt recruitment assumption in the SRA calculation (Fig. 13). Assuming compensation $K=5$, this current abundance required $R_0=21600$, and the Beverton-Holt model predicts recent (1990s) average annual recruitments of around 7500 yearlings, i.e. about 1/3 of the natural level.

Calculated exploitation rates (U_t) from the baseline scenario (Fig. 14) indicate that the high-tonnage fishery before 1900 did not generate very high exploitation rates (the tonnage represented relatively few large fish). Rather, peak commercial exploitation rates are estimated to have occurred during the 1960-90 period of highest gill net by-catch.

4.3.1 *The importance of uncertainties about numbers versus weight of fish caught*

If the early fishery did take fish averaging 35 kg, then in order to have survived the pre-1920s fishery the stock must have started out with at least 127,000 vulnerable fish (17,600 annual age 1 recruits) if recruitment compensation was strong ($K=100$), and even more (194,000, 26,900 annual recruits) if recruitment compensation was very weak ($K=2$). But if the average weight of fish caught did decline dramatically when the early fishery collapsed around 1920, then the gill net catches taken later must in fact have generated even higher exploitation rates than those which resulted in the early fishery collapse.

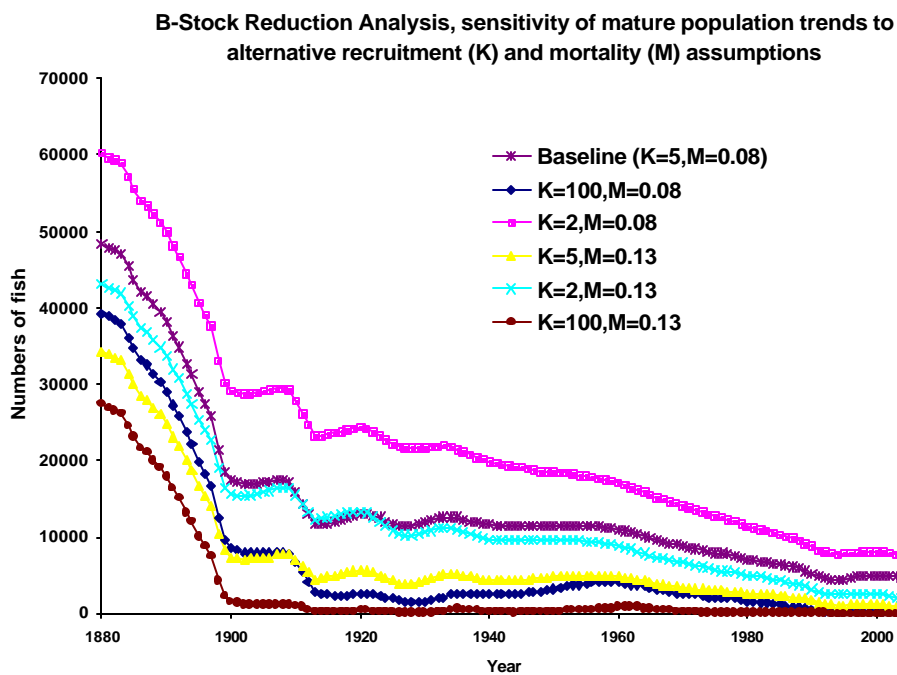
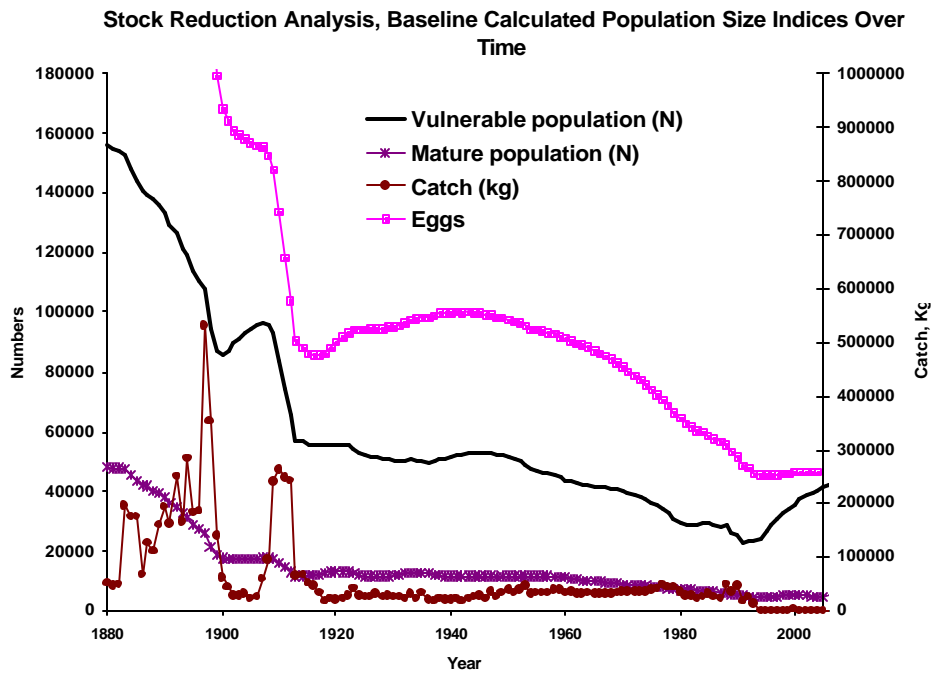


Figure 12. Stock reduction analysis (SRA) model estimates of historical white sturgeon stock sizes needed to have withstood historical catches and give recent abundance consistent with mark-recapture data. Vulnerable population is to line fishing gear, where vulnerability increases from zero for fish less than age 5 to full (1.0) at age 15. Mature population is number of age-20 and older fish. A-baseline run with recruitment compensation $K=5$, natural mortality rate $M=0.08$; B-alternative scenarios showing sensitivity of estimated adult population numbers to a range of reasonable assumptions about K and M .

60cm+ Population size from SRA model vs Mark-Recapture Models with annual recruitment changes

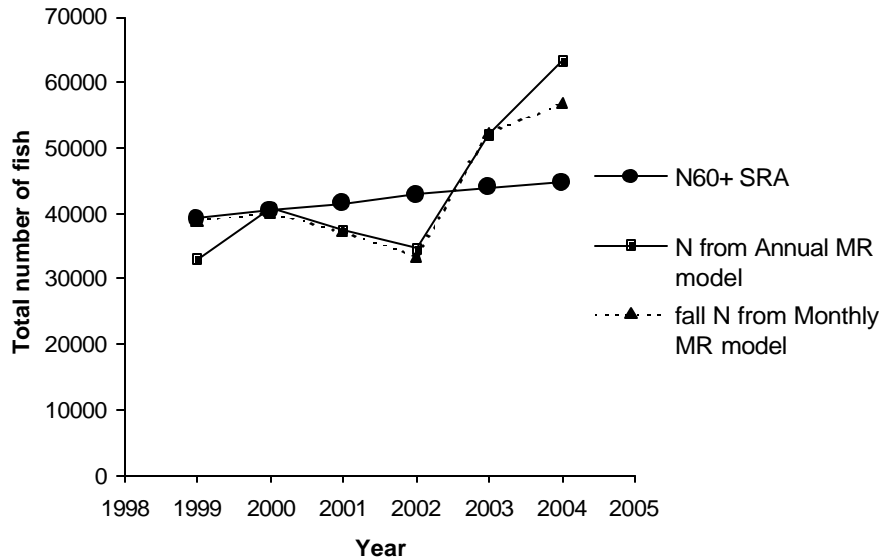


Figure 13. Comparison of recent abundance of 60 cm+ fish calculated by the SRA model to abundances estimated from mark-recapture analyses that did not assume similar recruitment rates from year to year.

Calculated exploitation rates as proportions of vulnerable population to gear used over time and as proportions of 60cm+ population

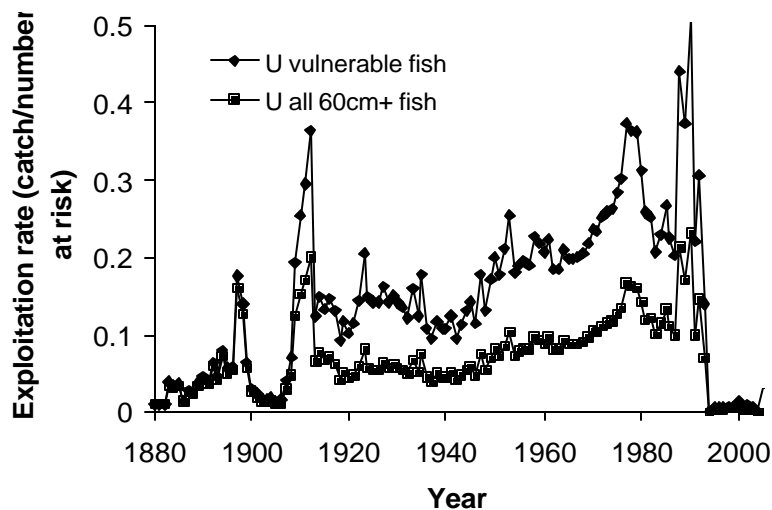


Figure 14. Calculated exploitation rates over time from the best-fitting SRA model. Note that exploitation rates as proportions of numbers of fish vulnerable to gill net fishing may be unrealistically high for recent years, and may reflect underestimate of the number of fish vulnerable to gill net harvest.

For most stock reduction scenarios the model indicates that the highest historical exploitation rates likely occurred not in the late 1800s when catches were highest, but rather in the 1960-80 period when the Area 29D (“dogpatch”) commercial chinook gill net fishery was most intense. During the heyday of gill net fishing, it is quite possible that very few fish survived through the gill net size window (40-100cm), so that recruitment to the adult population was virtually stopped. But this calculation rests heavily on the assumption of a large decrease in average size of fish harvested in the early fishery versus the later gill net fishery.

When changes in body size are ignored in the SRA (catches converted to numbers, and exploitation rates calculated accordingly, under the assumption that body size in the catch was always near its current average of 3-3.5 kg), the model indicates that the early historical fishery must have reduced stock size by at least 95%, and more likely by 99%. But as indicated in the next subsection, we reach just the opposite conclusion, that in fact stock size was always relatively healthy, when widely divergent body sizes are assumed for the early vs late gill net fisheries. This means that changes in average body size in the catch are a critical uncertainty in the overall analysis, and further studies should place particular emphasis on trying to recover information from historical sources about how much the average size of fish in the catch has actually changed over the last century.

The predicted temporal pattern of decrease in mean body size in the best-fitting SRA models is not nearly so violent and rapid as estimated by Rieman and Beamesderfer (1991) for the Columbia River. Apparently exploitation rates increased much faster in the Columbia River fishery than they did on the Fraser, and must have nearly wiped out the adult stock in less than one decade whereas large catches were taken on the Fraser over at least 20 yrs.

4.3.2 Was the natural stock size all that much larger than the current stock?

It has been typical in SRA analyses to find that early (e.g. 1800s) large fisheries severely reduced stock size, so that many stocks have been much smaller throughout the 20th

century than they could have been. To our surprise, we found that the early fishery may not have reduced the stock all that much, since (as noted in the previous section) the large tonnage catches may have been a relatively small number of large fish, and that the biggest impact on the stock may in fact have been in the 1960-80 period of intense gill net interception.

Under the assumed large decrease in body size of fish caught after 1920, the SRA model in fact gives best fits (gives current population sizes closest to mark-recapture estimates) with unfished recruitments and stock biomasses only about 3 to 5 times higher than at present (i.e., stock is 20-30% of its original level), and with spawning populations (or more precisely, annual egg production) on order 10x higher than at present.

4.3.3 Is a strong rebound in recruitment occurring, and if so what is causing it?

Two data sources indicate a very strong surge of recruitment to younger ages (4-10) over the last several years, 1998-2004 (Fig. 15). First, white sturgeon catches in the Chinook and chum gill net test fishery at Albion have increased greatly in the last few years (mainly in the late chum fishing), and this increase would be mainly of fish spawned in the early to mid 1990s. Second, the 1999-2004 size composition from the sturgeon tagging program has relatively large numbers of fish of apparent ages 7-10 (Fig. 16), which translates into calculated recruitment increases in the early to mid 1990s when age composition estimated from length composition is used to backcalculate recruitment by the $R_{t-a} = C_a e^{Ma}$ method described by Korman and Walters (2001).

One explanation for the recent recruitment increase is that there have been just one or a few unusually strong year classes, and these are recruiting to the sampling fisheries over an extended time period. Annual length-frequency patterns in the fish sampled for PIT tagging (Fig. 16) do suggest a shift in modal size from 1999-2004 of about the magnitude (7-10 cm/yr) expected from growth of a single cohort over time. If this explanation is correct, there will soon be a substantial decrease in Albion test fishing c.p.u.e. (as fish

grow out of gill net vulnerable sizes), and a substantial increase in the average size of unmarked fish captured in the tagging program.

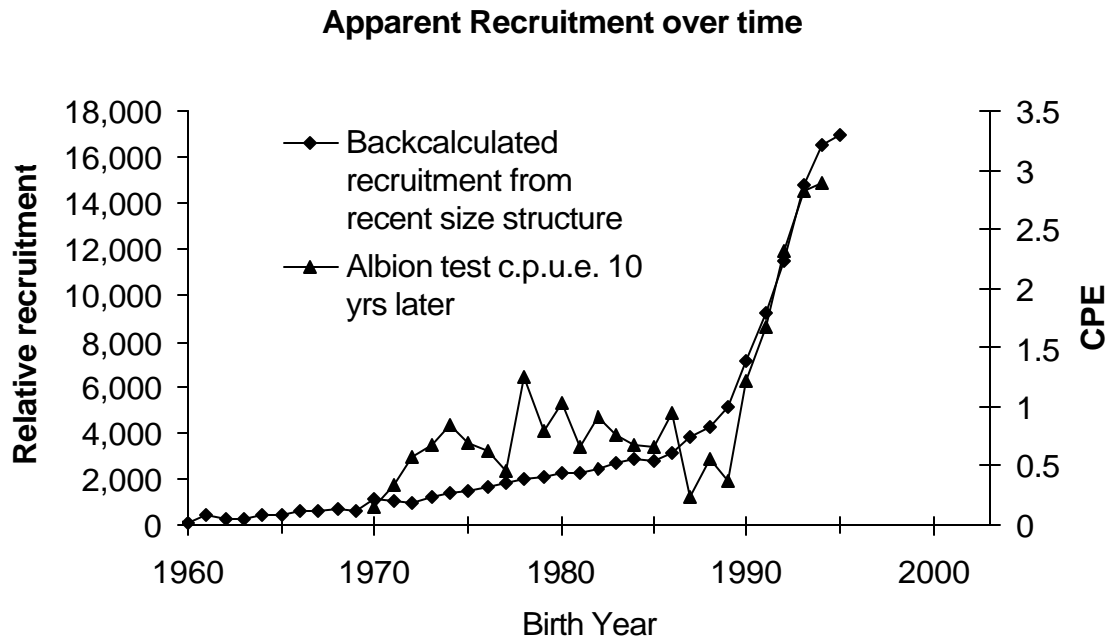


Figure 15. Two indices that suggest rapid increase in recruitment of fish from early 1990s spawning. Albion Chinook + chum gill net test catch per effort (and/or total annual catch) increased greatly in the late 1990s, and there are disproportionately high numbers of younger fish in recent size composition data gathered in conjunction with PIT tagging.

While such a recruitment increase could of course be due to some favorable change in environmental/habitat conditions, we strongly suspect that it is the result of a strong surge in spawning around 1990-1995. We suspect in turn that this surge was due to closure of the upriver gill net fishery. If that fishery was generating very high exploitation rates on 7-10 yr old fish up until 1980, as indicated by the SRA model, then after the closure a surge of such juveniles would have moved through the population to reach sexual maturity at around age 20, in the late 1980s and early 1990s. Very likely it is those young adult fish that have produced the recruitment surge we are seeing today.

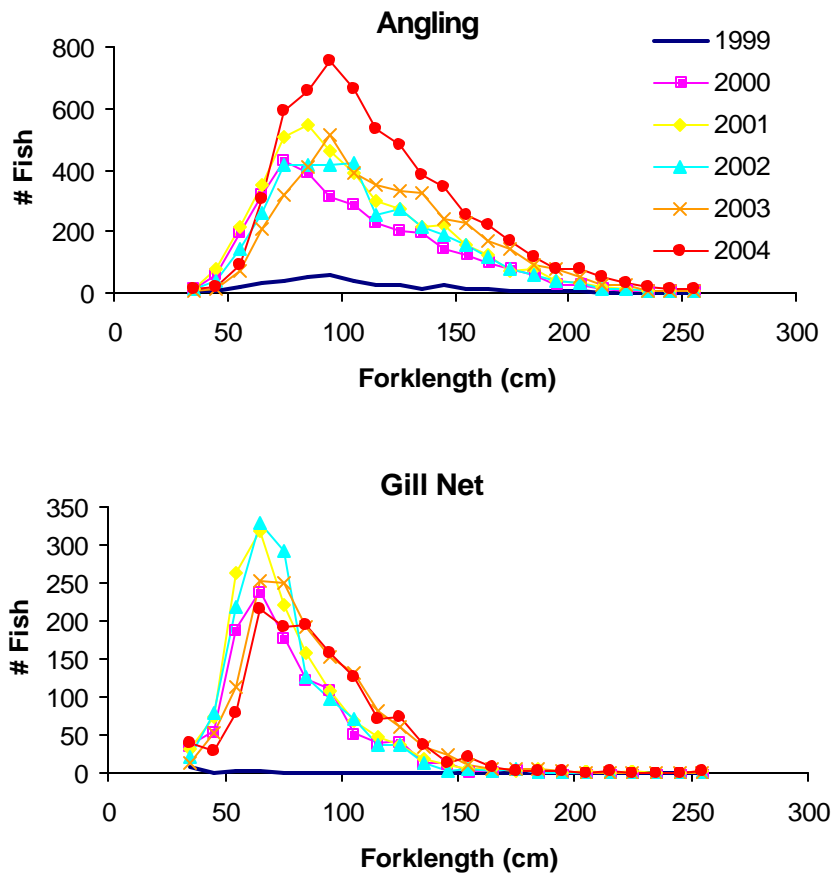


Figure 16. Length frequencies of fish captured as part of the FRSCS tagging effort by angling and gillnet.

However, this argument about the importance of young adult fish in producing the recruitment surge depends heavily on a hidden assumption about relative fecundity (or spawning frequency) of fish just reaching the age at first maturity. Typically we find that recently mature fish have radically lower fecundity than older, much heavier fish, so that their contribution to total egg production is relatively minor. Under that assumption, the SRA model indicates that the surge of maturing fish would have had relatively little impact on total egg production. But if the SRA model is modified to make such young fish relatively more fecund, then the model does predict the observed recruitment surge.

Another difficulty with the argument about maturing fish producing a recruitment surge is that the SRA model assumptions, which lead to a pulse of such maturing fish (very high gill net exploitation rates in the 1970-80 period; see Fig. 14), in turn lead to predicted age structures in the late 1990's and early 2000's that disagree sharply with available data (Fig. 17). These models predict a “hole” in the population age structure representing loss of those cohorts that were 7-11 years old while exploitation rates were high, i.e. missing fish in ages 25-40 sampled from 1995 to 2000. Such fish are obviously not missing from the sampled age structure.

Predicted vs Observed Age Distributions

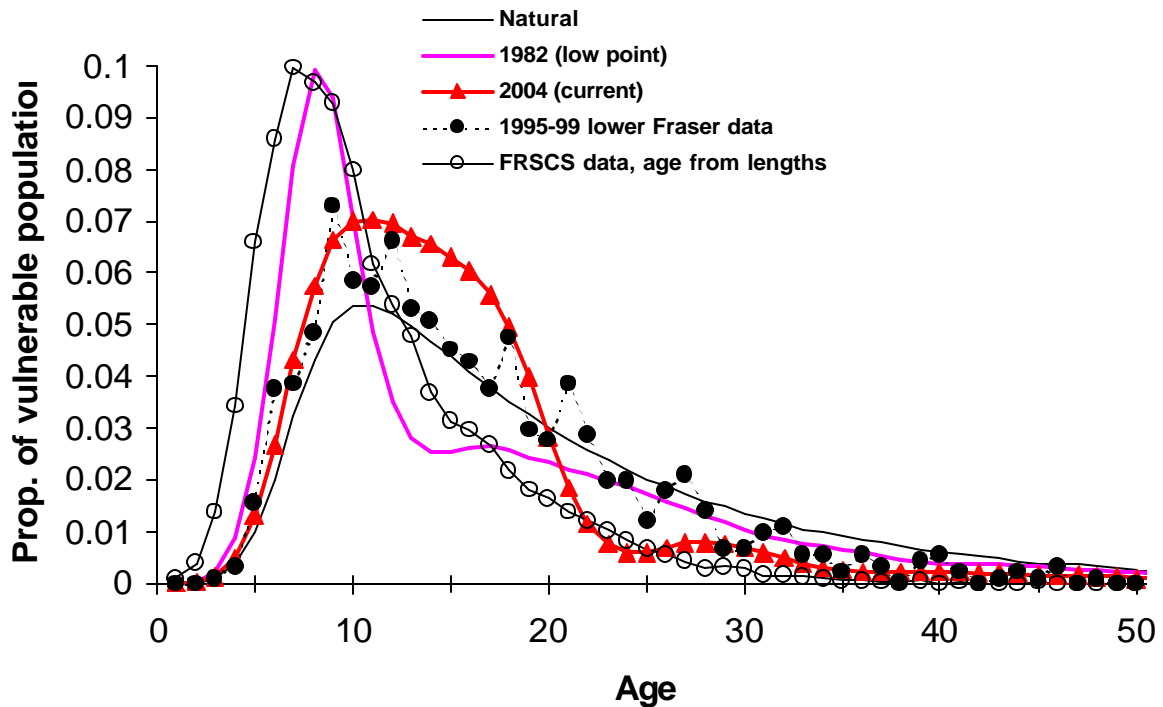


Figure 17. Predicted age structures from the SRA model for 1982 and 2004 compared to sampled age structure in 1995-99 and calculated 2000 age structure from 1999-2004 length distribution of PIT tagged fish from FRSCS data.

Ageing errors have likely smoothed the sample patterns in Fig. 17 to some degree, but almost certainly not enough to have masked the severe hole in age structure predicted by SRA models that estimate very high gill net exploitation rates. The presence of good numbers of fish that must have survived the gill net fishery indicates that the stock is very likely even healthier than indicated by the SRA models, and also suggests that there could well be many more fish in the population than indicated by the mark-recapture estimates (more fish need to be present today in order to explain the number of fish that needed to have been present in 1970-80 to have produced the gill net catches without those catches having caused a deep hole in the age structure).

4.3.4 Is it possible that the population is considerably larger than indicated by mark-recapture data, and is stable rather than growing?

Based on age composition data for 1995-1999 reported in RL&L (2000), Korman and Walters (2001) estimated that the lower river population has an apparent total mortality rate of around $Z^*=0.1$ and might be stable or growing slowly. These data show no severe depression of age 20-40 fish as might be expected if the fish had been subject to high gill net exploitation rates in the 1970-1990 period. Over that period, reported gill net catches averaged around 5300 fish/year (Echols 1995). But the gill net size data indicate that only about 25% of the total 60cm+ (roughly age 7 and older) fish are vulnerable to gill nets. So during the 1970-90 period, assuming $(\text{Catch})=U \times (\text{proportion vulnerable}) \times (\text{total } N \text{ } 60\text{cm}+)$, we can estimate the total $N \text{ } 60\text{cm}+$ numbers as $(\text{Catch})/(U \times 0.25)$ for different assumptions about the annual exploitation rate U of gill net vulnerable fish, while constraining the U to values low enough to not have severely impacted the age structure. So if U averaged 0.2 during the 1970-90 period, the average $N \text{ } 60\text{cm}+$ population must have averaged at least $5300/(.2 \times 0.25)=106,000$ fish. U values higher than 0.2 predict noticeable distortion of the recent (1995ff) age structure (a hole or missing older fish).

It is of course possible that older fish were over-represented in the 1995-99 samples for some reason and that a higher proportion of fish than 25% were in fact vulnerable to gill

nets during 1970-90. RL&L sampling was concentrated upstream from areas in the lower river where juveniles are concentrated and they used large hooks on long lines that likely selected for larger fish. But taking the direct age composition sampling and catch data at face value, we certainly cannot reject the hypothesis that the population was/is at least double that estimated by mark-recapture methods and by fitting the SRA model to the marking data, i.e. has essentially recovered from pre-1920 fishery impacts. Under this hypothesis, we would interpret the size composition data from PIT tagging as being a distorted sample with smaller fish overrepresented and/or older fish being underrepresented, an obvious possibility for recreational fishing gear and possible wider dispersal or emigration of larger fish from areas where the recreational tagging has concentrated.

4.3.5 How severe are current fishery impacts, especially on stock rebuilding rates?

We estimated the incidental take of sturgeon from gill nets used in First Nation (FN) weekend fisheries. To do this we computed the monthly average of the number of set nets in the water during the weekend upstream of Mission from DFO monitoring records (Eamon Miyagi, Department of Fisheries and Oceans). To index CPE we used the average catch of sturgeon per set net per month from first nation fishers participating in the FRSCS Lower Fraser River First Nations Sturgeon Stewardship Program (Lakahahamen/Laq'aimel, and Hatzic) located a few kilometers upstream of Mission. The number of fish caught over the month (based on weekend fishing) and the number of nets are known at these locations (T. Nelson, FRSCS, unpublished data). CPE was calculated for each month from April through August from 2000 through 2004 and multiplied by the corresponding number of set nets upstream of Mission (to Sawmill Ck. ca. 120 km upstream) to estimate the total catch. Incidental mortality was estimated by multiplying the total catch by mortality rates of 0.07 and 0.13 representing the low and high range of values obtained from the FRSCS gill net mortality assessment (T. Nelson, FRSCS, unpublished data). We caution that estimates of gill net catches based on a very limited set of CPE estimates are inherently uncertain. We are assuming that CPE estimates from a handful of nets in one small region of the lower Fraser River are

representative of the catch rates for the entire lower river. While the estimates we provide are highly uncertain they provide a starting point for this analysis. Future monitoring efforts should focus on providing more representative CPE estimates.

The majority of FN fishing effort upstream of Mission occurs between April and August (Fig. 18). The average effort over these months from 2000-2004 has been quite consistent at 185 nets per weekend (CV= 13%). Peak effort occurs in August with an average of 314 nets between 2000 and 2004. Based on the FRSCS data, catch rates of sturgeon increased steadily from April to August reaching a peak of over 30 fish per net per month in August. The total number of fish estimated to have been caught between April and August averaged approximately 17,000 between 2000-2004 with the majority of catch occurring in July and August (Table 1). Data provided by the FRSCS support an assumed 7% and 13% mortalities associated with gillnet capture, that results in incidental harvest estimates of approximately 1,000 to 2,000 fish per year. Estimates of total catch of sturgeon (harvest + release) and harvest through DFO catch monitoring from 2000-2003 averaged 1,237 and 11 fish, respectively. Assuming our estimates are more representative, DFO catch monitoring underestimates the total catch by about 14 fold and the total incidental mortality by 100-200 fold.

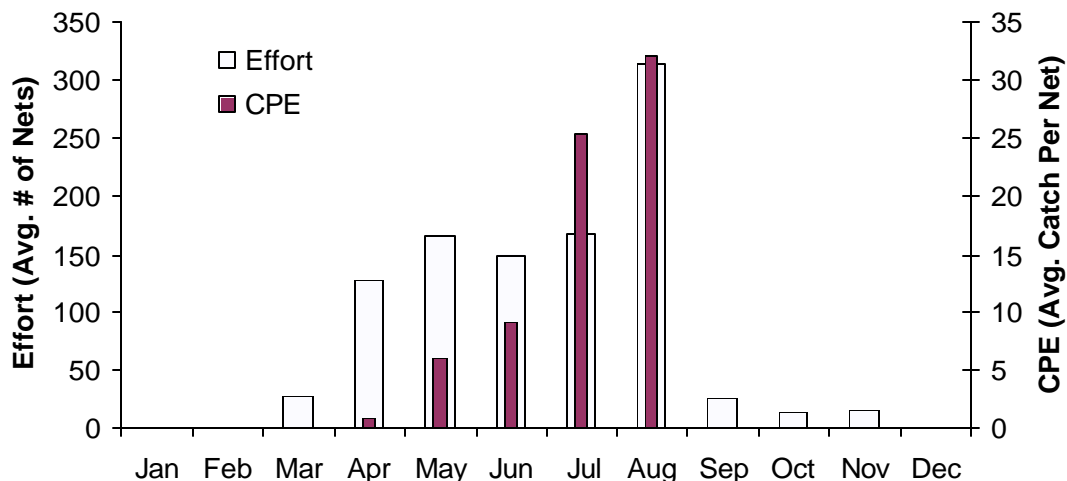


Figure 18. Seasonal trends in effort for the First Nation’s gillnet fishery upstream of Mission and the incidental catch rates from 3 locations near mission from 2000 through 2004.

Table 1. Estimates of the number of sturgeon caught and incidental mortality associated with the First Nations gillnet fishery during the weekend upstream of Mission.

	2000	2001	2002	2003	2004	Avg
April	131	102	94	99	99	105
May	1,107	1,546	858	659	867	1,007
June	1,230	5,138	371	116	1,088	1,589
July	12,995	925	5,112	1,078	4,809	4,984
August	8,294	8,144	13,275	13,541	5,738	9,798
Total Caught	23,757	15,854	19,710	15,493	12,600	17,483
Low Kill (7%)	1,663	1,110	1,380	1,085	882	1,224
High Kill (13%)	3,088	2,061	2,562	2,014	1,638	2,273

While acknowledging data uncertainties it is important identify that our estimation procedures for incidental FN harvest yield minimal values. First, estimates only include fishing upstream of Mission and do not include capture and incidental mortality of fish caught below Mission in the drift net fishery. Second, FN effort estimates from DFO only include fishing during the weekends and not other openings or illegal non-weekend fishing. Monthly catch at FRSCS cage monitoring sites also only include fish captured during weekend fishery. Our calculations therefore do not account for fish captured outside of the weekend period. Finally, incidental mortality rates estimated from FRSCS data are minimal values as some fish released from the recovery cage were in very poor condition and could well have died. As well, during the early part of the assessment some fish that were dead when the nets were picked were not added to the cages.

We were unable to estimate the current incidental catch of sturgeon in the commercial salmon gillnet fishery. Estimates of the by-catch of sturgeon in the commercial fishery is very likely underestimated through existing DFO monitoring. Capture of sturgeon in the

Albion chinook and chum test fisheries indicate that by-catch in drift nets can be substantial. This potential source of mortality should be evaluated through modifications to the current commercial catch monitoring program coupled with an assessment of latent mortality using the FRSCS assessment methods.

Under even the minimum estimates of abundance of 60 cm+ sturgeon in the lower Fraser, i.e. 40,000-60,000 fish, current by-catch mortalities by First Nations gill net fishing (1700 fish/yr) and recreational fishing (100 fish/yr) most likely do not exceed 4% of the population. There is some indication from PIT tag mark rates (proportion of marked fish in sample examined for marks) that the gill net fishery may see a different portion of the population than the angling fishery. Mark rates by gear and year were as follows:

	Angler	Gill Net
1999	0.02	0.00
2000	0.07	0.04
2001	0.17	0.09
2002	0.21	0.12
2003	0.21	0.14
2004	0.24	0.20

Gill net catches (a high proportion of this sample is from the Albion test fishery) had considerably lower mark rates until 2004, indicating that gill nets saw unmarked fish that were not being seen by anglers. Probably mixing of fish from areas of intensive tagging has gradually evened out the mark rate over space in the last year or so. But we can use the 2004 mark rates to obtain a direct estimate of the exploitation rate caused by gill nets. If the total gill net catch in 2004 was around 12,600 fish (Table 1), and if these fish had a mark rate of 0.2 as indicated by the data, then the gill net fishery must have captured a total of around $12,600 \times 0.2 = 2,520$ PIT tagged fish. Averaged over 2004, we estimate from the numbers of fish tagged and corrected for survival to 2004 that there were between 9,000 and 11,000 tagged 60 cm+ fish at risk to gill net capture (low value assumes 85% annual survival, high value assumes 92% survival). Thus the gill net exploitation rate must have been on order $2,520/10,000 = 0.25$ of the 60cm+ population,

but only about 7-13% of this rate represents mortality if all fish were released.

Incidentally, a simple population estimate would be the catch divided by the exploitation rate, i.e. $12,600/0.25=50,400$ fish, which agrees well with more complex mark-recapture estimators.

However, note that if we are correct in concluding from age selectivity assessments that only about 25-40% of the 60cm+ population is vulnerable to gill net capture, then the 25% exploitation rate indicated by mark rate and total catch estimates must represent a rate of 2-4 times greater per year on the subpopulation actually vulnerable to gill nets. This is higher than the rate estimated for the 1980s from the SRA model (Fig. 14), and is high enough to have caused more distortion in the population age-size structure than is apparent from size and age composition data (see Figure 17), for that recent historical period when fish captured in gill nets were still being retained.

The apparent inconsistency between gill net exploitation rate estimates from SRA and tagging data versus available size-age data (exploitation rates predict a hole that is not observed in the age distribution) suggests that the population may be larger than indicated by mark-recapture data, and that the age structure may be smoothed by movement of fish into the river from some pool of fish that has not been fully subject to gill net exploitation. Jim Echols (DFO, pers. comm.) suggests that white sturgeon may move into estuarine and/or ocean areas, perhaps traveling long distances while in that spatial refuge from most fishing impacts. Limited Fraser River tagging data support this hypothesis; a juvenile sturgeon tagged and released from a commercial gill net vessel 5 km west of the mouth of the Fraser River was recaptured twice upstream of Mission, and three white sturgeon caught in the Lower Fraser during the FRSCS tagging study were confirmed recaptures from Columbia River tagging projects (T. Nelson, FRSCS, pers. comm.). There are also numerous records of individual sturgeon movements between the extreme lower estuary and the upper Fraser Valley (Nelson et al. 2004). Age patterns of strontium concentration in fin rays (Veinott et al. 1999) suggest that as much as half of juvenile fish (<15yr) may have an estuarine rearing period. To explain inconsistencies in the data, and also why the natural mortality rate M appears to be higher than expected

from sturgeon growth parameters, roughly 5-10% of the age 5 and older juveniles would have to move out of the river each year; this would result in a “refuge population” of around half the total juvenile population vulnerable to gill netting, if the refuge fish have about the same migration rates back into the river. Note that it is important to this hypothesis that much of the emigration movement consists of fish too young to be fully vulnerable to gill nets, otherwise the fish representing this ‘hole’ in the age structure would simply have moved out to sea and then back into the river resident population.

There is a unique opportunity at this time to actually estimate movement rates out of and back into the Fraser, and to measure large-scale movements along the Pacific coast. The POST (Pacific Ocean Shelf Tracking) program is deploying long-lived (6-yr battery) VR3 acoustic receivers on transects across the continental shelf from California to Alaska, to listen for acoustic tags from a variety of research programs. Receivers will also be deployed within the Fraser River to measure outmigration survival of salmon. Acoustic tags are now available with battery lives of at least 6 years. We very strongly recommend that such tags be applied to a sample of white sturgeon from the Lower Fraser River.

4.4 A Stochastic Approach to Stock Reduction Analysis

In deterministic SRA models we calculate a single trajectory of stock sizes over time, essentially asking the question “If the stock size had unfished recruitment R_0 and historical recruitments exactly equal to the predicted values from a stock-recruitment curve after fishing started, how large would R_0 have to have been in order that the stock have ended up at its present estimated size?”. Despite knowing that no stock ever exhibits recruitments over time that are close to a recruitment curve, we hope that answering this question will at least give the expected or most likely trajectory for the stock to have followed. Unfortunately this is not really a wise hope, considering nonlinearities of population dynamics responses to harvest.

A less presumptuous but computationally more difficult question is to ask “If the stock had average unfished recruitment R_o and exhibited recruitments over time that varied around a mean recruitment relationship (around a stock-recruitment curve) in a pattern similar to that observed for other populations, how probable is it that stock size would end up at or near the present estimated size?”. The essential idea in asking the stock reduction analysis question this way is to recognize that we cannot, and should not, and need not, pretend to know how a given stock behaved over time in any precise or even average way. More precisely, we can ask what the probability is of having obtained the observed population size estimate if the stock (1) had mean historical recruitment rate R_o , (2) followed a mean stock-recruitment relationship that passed through R_o , (3) exhibited realistic variation around that relationship, and (4) was subject to observed historical harvest removals. This probability is an integral of the form

$$P(\hat{N} | R_o) = \int_N P(N | R_o) P(\hat{N} | N) dN$$

Here $P(N | R_o)$ is the probability of the stock ending up at size N given R_o (and a mean recruitment curve with variation, and the historical catches), and $P(\hat{N} | N)$ is the probability of obtaining the estimate \hat{N} if the population size were actually N . The second of these distributions $P(\hat{N} | N)$ is essentially just a likelihood function, representing the odds of obtaining the observed estimate given the true parameter value; we can approximate it simply by assuming normality for the estimator, with some standard deviation $\sigma_{\hat{N}}$. The first, $P(N | R_o)$ is much more complex to calculate.

If we were dealing with a very simple population model of the form $N_{t+1} = f(N_t, C_t)$ without complexities in the recruitment prediction due to changes over time in size-age composition and fecundity, a good computational approach to evaluation of $P(N | R_o)$ would be to convert the model into a discrete Markov process with transition matrix $P(N_{t+1} | N_t, C_t)$ as suggested by Hammond (2004). Unfortunately that approach does not appear to be practical for models with large numbers of state variables (numbers by age). Instead, we must resort to some scheme that involves sampling the probability

distributions of past states, as suggested by Schnute and Richards (2001) and Richards and Schnute (1998).

In the sampling approach to estimation of $P(N | R_o)$, we parameterize recruitment variation in some simple way, e.g. by annual log-normal multipliers $R_t = e^{w_t}$ on the mean recruitment relationship so that $N_{1,t} = f(E_{t-1})R_t$ where E_{t-1} = total egg production the previous year and $f(E)$ is the stock-recruitment curve. We then choose a large sample of the recruitment “anomalies” w_t , assuming these to have been sampled from a normal distribution with mean zero and standard deviation s_R (typical values of s_R are in the range 0.3-0.6). Each sequence of T+A such values (T=number of years, A=number of ages) defines a possible population history, where the first A w_t values in the sequence are applied to the initial age distribution $N_{a,1}$ and the rest to recruitments over time. This approach recognizes that the stock likely had not only variation over the historical fishing period, but also likely varied in its initial abundance from its very long term average abundance. When a very large number (e.g. 10000) of such sequences and simulations have been done, we can approximate $P(N | R_o)$ simply by examining the frequency of N values (binned into suitable sample intervals) over the simulation trials. Note that at least for low R_o “hypotheses”, this frequency distribution is not expected to be just smooth and continuous; instead it will have at least some proportion $P(N=0|R_o)$ of trials where the historical catches are predicted to have driven the stock to extinction (i.e. where stock size is predicted to have been less than the observed catches).

We have developed a simple Visual Basic application to do the sampling approach to estimation of $P(N | R_o)$ and $P(\hat{N} | R_o)$ for age-structured population models.

Surprisingly, for the sturgeon case where T=125 yrs and A=100 ages, it only takes about 25 sec to generate 10000 sample trajectories (for one R_o hypothesis); this time drops to 19 sec per R_o hypothesis when we use the same sample w_t sequences in comparisons of calculated probability distributions across R_o hypotheses. An example of the time probability distributions for stock size generated by this application for the Fraser

sturgeon example are shown in Fig. 19, for an R_0 case (22000 annual age 1 recruits) close to that estimated by deterministic SRA.

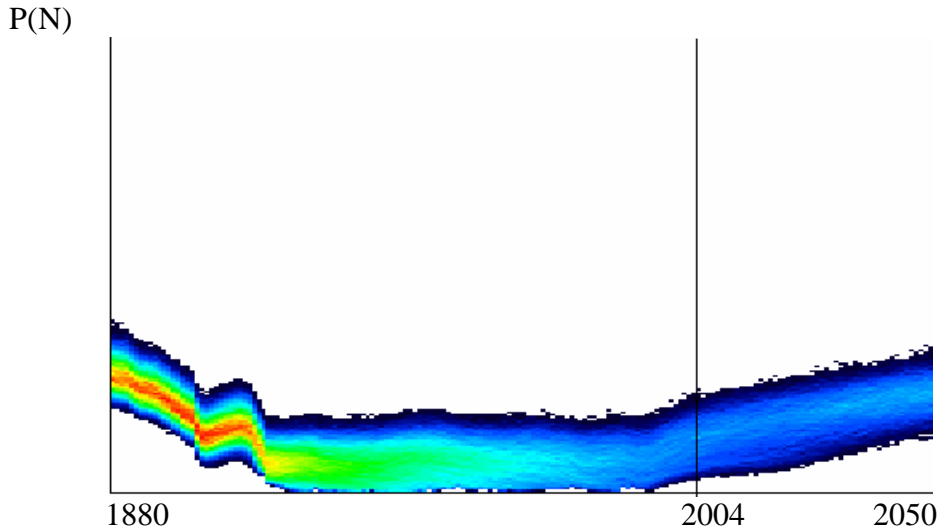


Figure 19. Probability distributions for vulnerable stock size (to line fishing) for lower Fraser River white sturgeon, using stochastic SRA. Rainbow color scale shows high probability values in red, low values in blue. Model run with mean historical recruitment $R_0=22000$ age 1 recruits, recruitment compensation $K=5$. Forward projections to 2050 are assuming total bycatch mortality of 10000kg/yr (around 1000 fish/yr) by First Nations gill net and recreational fisheries. Note that for this R_0 hypothesis, a high proportion of the simulated recruitment histories resulted in stock extinction (see Fig. 20).

This approach offers two quite distinct measures for judging the credibility of alternative mean recruitment R_0 hypotheses. One is based only on historical catches, and is simply the probability that that stock would have been able to produce those catches without going extinct, $1-P(N=0|R_0)$. The other is the overall probability, integrated over possible current population sizes, of producing the observed population estimate $P(\hat{N} | R_0)$. The second measure will necessarily be low for R_0 hypotheses implying high $P(N=0|R_0)$, but using it implies a possibly unwarranted trust in our ability to assess uncertainty about the current population estimate (i.e. to specify $P(\hat{N} | N)$ correctly, which is obviously problematic in the sturgeon case where the estimate is from mark-recapture methods that may be biased downward due to some proportion of the stock being inaccessible to

capture). Fig. 20 shows how these measures vary for the lower Fraser River sturgeon case, assuming recruitment compensation $K=5$; note that the results for 1000 simulation trials are virtually identical to those for 10,000 trials.

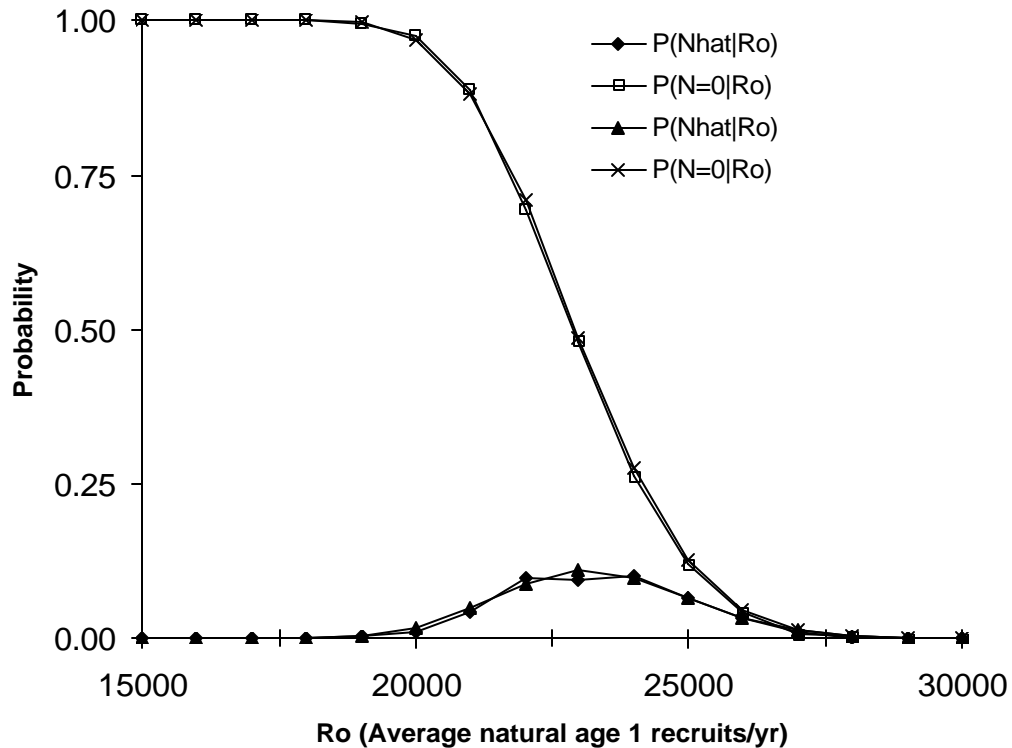


Figure 20. Calculated probabilities of extinction before 2004 ($P(N=0|Ro)$ lines) and of having produced recent population estimates of around 50,000 fish (with standard deviation 10,000) vulnerable to angler sampling ($P(Nhat|Ro)$ lines) for lower Fraser River sturgeon. Alternative estimates for each relationship are for using 1000 versus 10000 simulation trials to estimate probabilities of alternative population trajectories.

For the sturgeon case, it is encouraging that the two credibility measures are in good agreement, and that the unfished average recruitment rate for lower Fraser white sturgeon was very probably between 20,000 and 25,000 age-1 fish per yr. The deterministic SRA gave a most probable value (for explaining mark-recapture results) of around 21,700 fish, about 1,000 less than appears most probable from the stochastic analysis which accounts

for how a stock with a natural average of less than 22,000 fish would have had a high probability of being driven extinct by the historical catches. Further, all of the R_0 hypotheses that represent a stock even remotely likely to have survived the historical fishery lead to high probabilities that the stock will grow over the next 50 years, so as to approach the unfished level if there is no retention fishery and if bycatch mortalities do not increase.

There are at least two important caveats about results such as those in Fig.'s 19 and 20. First, considerably higher variances in population size (and uncertainty about average R_0), are obtained if autocorrelated variation in recruitment residuals w_t is admitted, i.e. there may have been recruitment “regime shifts” that caused w_t to behave as $w_t = \rho w_{t-1} + v_t$ with v_t independent but $\rho > 0.4$. Such high ρ values are relatively uncommon in stock-recruitment data sets, but they certainly do occur. Second, “small” populations such as Fraser sturgeon may have considerable variation in abundance of older fish due to “demographic stochasticity”, i.e. variation in numbers of fish surviving each year just due to vagaries in the binomial process of individual survival. Our Visual Basic application allows individual-based sampling of survival for individuals whenever cohort size ($N_{a,t}$) is less than some critical value N_c . When N_c is set to 1000 for sturgeon (so that we simulate individual survival events for most cohorts by the time they reach mature ages; with annual $S=0.92$, cohort numbers are essentially deterministic for $N_{a,t} > 1000$), we fortunately do not see noticeably higher variation in vulnerable abundances than shown in Fig. 19.

5.0 DISCUSSION

We have examined a wide variety of data sources in this attempt to estimate abundance and trends in lower Fraser white sturgeon. In the end it is discouraging to see that these data sources do not imply a single, unambiguous interpretation or hypothesis about what is happening. At one apparent extreme, the mark recapture analyses, Albion gill net index data, and SRA population models indicate that the population is about 60,000 fish and is growing rapidly. At the other, age composition data in conjunction with gill net catch estimates imply that the population could be much larger and stable.

But in fact we cannot even say conclusively that the population is either growing or stable. The most troubling inconsistency in the data is between the estimates of exploitation rates for gill net fishing from SRA models and mark rates in gill net catches, versus direct evidence of survival from age and size composition samples. The exploitation rate estimates indicate that there should be a hole in the population age structure corresponding to periods of high gill net retention in the 1970-90 period, but no such hole is evident in the size-age data. If we take the age structure data at face value, and assume that reported commercial gill net catches did not in fact cause high exploitation rates, then we must admit that the population being harvested by commercial fishers must have been much larger than we would estimate is present today based on mark-recapture methods. So if we then also take the mark-recapture estimates at face value, we are forced to conclude that there may even have been a considerable population decline since 1970. In fact, the Albion chinook test fishing data even lend support to this possibility; despite increase in the combined chinook and chum test fishery sturgeon catch and c.p.u.e., the chinook data do show a decline in catch per effort from 1984 to 1999, before joining the chum data in indicating a rapid population increase since then. Also, the SRA model shows decline for the 1970-1992 period, but followed by rapid increase after retention of sturgeon in recreational and net fisheries was stopped in 1994.

It is important to emphasize that every data source available for this assessment is subject to widely different interpretation depending on assumptions that cannot be directly tested. Our reconstructions of age structure from length composition could be severely distorted by errors in estimating age from length (uncertainty about growth curve) and from size selectivity in sampling. The direct age composition data could also be distorted by age-size changes in vulnerability to sampling gear. The mark-recapture data could be leading to underestimates of abundance due to overestimates of capture probabilities, due in turn to differentially high recapture probabilities for tagged fish due to concentration of fishing in the areas where fish were tagged. Historical fishery catches could easily be underestimates, and we do not have accurate information on the mean size of fish caught (only catches in weight are available). Abundance indices for sturgeon based on gill net test fishing for Chinook and chum salmon, in just one sampling location, could be changing due to changes in distributions rather than abundance and could be contaminated by reactions of sturgeon to the presence of other fish (both sturgeon and salmon) in the nets.

Looking at all of the data at once does not prevent anyone who has some particular belief from explaining away whatever data do not agree with that belief by referring to one or more of these possible (even likely) causes of misinterpretation. This observation has two key implications: (1) there is plenty of opportunity for sturgeon experts to waste a great deal of time debating about alternative hypotheses and interpretations of the data, without ever coming to any useful conclusion or wise consensus; (2) it would be grossly irresponsible to pretend certainty about any one belief or assessment, and to base future policy on that belief.

This wide range of uncertainty about abundance and trend leads to considerable uncertainty about current conservation status of the population. SRA results indicate that even if the population is still small and growing, the current spawning population is likely producing at least 10% of the number of eggs each year that a natural population would produce (this is a better index of reproductive stock status than the number of spawning fish, since it weights fish by the size-fecundity relationship and emphasizes the

importance of older, more fecund fish). In contrast, the stable population hypothesis based on age composition and gill net catch data would imply that current egg production is more like 40% of the natural production, with further increases to be gained mainly through increasing mean fecundity as older fish accumulate in the population. However, both the 10% and 40% extreme calculations may be considerable underestimates of recovery status, since they make the most conservative possible hypothesis about the importance of age for fecundity (namely that fecundity is proportional to body weight, which results in much higher fecundity estimates for older females). In fact, it is entirely possible that older females do not have such high importance for egg production, because they may spawn less frequently than younger females.

Should a retention fishery be allowed, by any interest group (First Nations, commercial, recreational)? This is a much more difficult question, to which we cannot provide an unambiguous recommendation. Most likely the stock can safely sustain an exploitation of 5%-10%/yr on 60cm+ fish, considering that it apparently survived such rates for most of the last century from gill net fishing alone. The real question is not whether the stock can withstand some reasonable exploitation rate, but rather whether retention should be prevented for at least some years in order to allow the stock to grow to a more productive level; there is a very considerable risk that allowing a sustainable exploitation rate at this time would lead to “sustained overfishing” (harvest that is sustainable but keeps the stock at a less productive level than would be possible to achieve). This is exactly the same question that has plagued debates about whether to allow harvest of some maritime cod stocks, with the added uncertainty that we are not really sure whether the sturgeon stock was ever badly depleted in the first place. Theoretically, the stock should move to an optimum level on its own, if an optimum constant exploitation rate is applied (which would be about 3-4%/yr for an MSY criterion of “optimum”). But if the initial stock level is much below optimum, as indicated by some SRA scenarios, it is easy to show with simulations of age structure recovery that even an optimum exploitation rate can radically slow the stock recovery, increasing the time required for recovery from an order of a few decades to an order of centuries. In simulations, increasing the fishing mortality rate from 0 to 2% increased the time for the stock to recover to 50% of our estimate of

the unfished population size from 20 to 40 yrs. The simulations also showed that fishing mortality rates as low as 5-8% would completely stop population growth. The risk of greatly delaying recovery might be justified if there were some pressing social or economic need to provide consumptive harvest (as has been argued in the cod situation), but no such need has been demonstrated in the sturgeon case. Given that there is considerable potential for stock increase, especially in view of recent indices showing recruitment increase, we just do not see a good case for risking that potential by allowing retention fishing by any interest group until more information is available about stock size trends.

The existing PIT tagging program in conjunction with the Albion gill net test fishery should allow clear determination over the next decade whether recent signs of recruitment increase are real and whether the stock is still increasing. The PIT tagging program might be improved somewhat through measures to spread tagging more evenly and/or adding a program component to track dispersal and seasonal migration rates using acoustic tags (and regular mobile tracking to determine distribution of tagged fish over time). To test whether the Albion fishery is measuring abundance changes or just shifts in distribution of sturgeon, it would be useful if practical to operate the same basic test fishing methods in several other locations along the river, for at least a few years. To improve estimates of stock structure and growth, it would be helpful to do as many age determinations as possible on fish from both the PIT tag program and the Albion test fishery, using the same methods as for the 1995-99 age composition samples. Beyond these improvements in gathering more of the same types of data as in the past (with attendant continuing risks of misinterpretation as noted above), some consideration should be given to somewhat radical and potentially very costly methods for moving to direct population counts, e.g. with a Didson acoustic camera. Besides providing minimum estimates of absolute abundance, such direct counting methods could provide useful seasonal-spatial data for distinguishing among hypotheses about changes in tag recovery rates due to changes in distributions versus changes in vulnerability to gear.

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The comprehensive analysis contained in this report focussed on estimation of population parameters and model development for population assessment. While management options are considered at various points in the text this appendix brings a number of those statements together to address specific questions of management interest. Further calculations were also performed using the deterministic SRA model in order to examine possible population effects of a broader set of management options. Development of this model was initiated in order to support a comprehensive allowable harm assessment. Information presented here indicates the utility of this model when undertaking such an assessment.

In presenting this analysis it is important to note that habitat alterations are not considered as part of this analysis. Links between the availability of different habitat types and population dynamics were not examined as part of this study, however, habitat alterations that caused recruitment declines could threaten the recovery of this population. Further habitat declines could also affect recovery.

1. What is the present / recent species trajectory?

Recent analysis by Nelson et al. (2004) indicates that the population was approximately 62,611 for 40-220 cm fish as of February, 2004, and the population had grown relative to their estimate of 50,564 fish as of January 2002. The present results indicate that a large number of fish may not be vulnerable to sampling in the Fraser River due to their residence in estuary or marine waters, and that the present total population may be as high as 100,000.

While there is continued uncertainty about mortality rates the most reasonable estimate appears to be $M=0.08$, and at this level the population should continue to increase in the foreseeable future. If recent recruitment increases are due to release from the effects of the historic commercial gill net fishery, then elevated juvenile numbers should be maintained.

2. What is the expected order of magnitude / target for recovery?

The present project was not intended to identify specific recovery targets, however, the SRA model can be used to evaluate the ability to achieve various targets. Potential recovery targets might be based on total population, number of mature females, or egg production, with the latter being most influential to population dynamics.

There is some indication from the present analysis that total population may be near maximum sustainable yield, however, there are also concerns that the optimum levels are higher. A recovery target of 10,000 mature adults (5,000 females) has been suggested as part of conservation planning for the Lower Fraser River, while 2,500 has been used in the Nechako and Columbia Recovery plans (NRWSRI 2004, UCWSRI 2004). No target

has been set for annual egg production as a measure of recovery. Present annual egg biomass may range from 10-40% of historic levels, particularly due to the low abundance of larger more fecund females. Targets of 25% and 50% of historic abundance are considered here for comparative purposes.

3. What is the expected time frame for recovery to the target?

Definitive answers to this question are challenging due to the uncertainty in a variety of parameter estimates. However, reasonable estimates lead to the following general guidelines. All of these statements include only natural mortality of $M=0.08$ with the modeled policy of no anthropogenic impact initiated in 2005. It is important to note that values are derived from the deterministic model and may vary from actual populations. Also uncertainty regarding parameter values (e.g. age specific fecundity and inter-spawning interval) can have large effects on outcomes of these scenarios. However, the relative comparison of scenarios provides an indication of the time scale of recovery to various thresholds.

- the current population may be near its MSY
- a target of 10,000 reproducing adults could be achieved in 8 years (2013)
- a target of 2,500 reproducing adults has already been achieved
- a target of 50% of historic egg production could be achieved in 61 years (2066)
- a target of 25% of historic egg production could be achieved in 27 years (2032)

These scenarios indicate that all recovery targets except egg production might be achieved within 10 years. Relative to the estimated generation time of 40 years, all targets might be reached within one generation, except 50% of historic egg production.

4. What is the maximum human-induced mortality which the species can sustain and not jeopardize survival or recovery of the species?

Modeling results indicate that an anthropogenic mortality rate of $M=0.05-0.08$ would stop population growth.

5. What is the aggregate total mortality/harm attributable to all human causes.

Estimated natural mortality rates are about 0.08. Estimates of the number of mortalities due to gill net by catch (1700) and using an assumed mortality of 100 fish due to the catch and release fishery leads to a total mortality rate due to anthropogenic sources of about 0.03 (assumed population=62,000). Quantitative methods employed for the estimation of natural mortality rates also suggest that the mortality rate due to anthropogenic sources is likely less than 4%.

6. What would the projected population be under various harvest scenarios?

Modeling the effects of gill net harvest and angling suggest that recovery scenarios might be expected to achieve a threshold of 25% of historic egg production as outlined below.

- current angling and gill net mortality achieves this threshold in 57 years (2062)
- current angling mortality and no gill net mortality achieves this threshold in 37 years (2042)
- no angling mortality and current gill net mortality achieves this threshold in 46 years (2051)
- doubled angling and gill net mortality would prevent recovery and is unsustainable.
- current angling mortality and doubled gill net mortality leads to a low egg production and does not meet the recovery target in 2 generations.
- double angling mortality and current gill net mortality achieves this threshold in 75 years (2080)

These scenarios indicate that recovery is delayed if activities leading to anthropogenic mortality are allowed, and that the gill net by-catch has a greater effect than the catch and release fishery. Relative to white sturgeon generation time of 40 years scenarios that allow current levels of mortality due to angling and gill net by catch still allow recovery within 1.5 generations. Since recovery appears to be most sensitive to mortality due to gill net by-catch, this fishery should be a priority for further monitoring and efforts to decrease mortalities due to this source.