Assessment of the Contribution of Two Constructed Side-Channels to Coho Salmon Smolt Production in the Englishman River

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ASSESSMENT OF THE CONTRIBUTION OF TWO CONSTRUCTED SIDE-CHANNELS TO COHO SALMON SMOLT PRODUCTION IN THE ENGLISHMAN RIVER

by

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ABSTRACT

M. Wright and J.A. Taylor. 2018. The contribution of two constructed side-channels to coho salmon smolt production in the Englishman River. Can. Tech. Rep. Fish. Aquat. Sci. 3261: 21 p.

Additional off-channel rearing habitat for juvenile coho salmon in the Englishman River watershed was provided through the construction of two side-channels, between 1989 and 1998. A series of annual mark-recapture programs was then conducted between 1998 and 2005 to estimate the contribution coho smolts from these channels to overall outmigration from the Englishman River. While methods varied among the various programs, a simple Petersen estimator was the calculation of choice in a majority of years, including some incorporating temporal stratification (PPE). The unadjusted (for unsampled stream length) estimates of smolt population size ranged from 29,238 (2001) to 47,591 (1999). Comparisons among these years indicate that significant departure from the modal value (41,890 in 2003) occurred only in the case of the lowest population levels, in 1998 (32,481) and 2001.

The utility of the dataset in providing a baseline for evaluating the subsequent channel enhancement strategy for the Nature Trust channel in 2007 was examined using a normal approximation to the binomial distribution. The method was found to be relatively insensitive to adjustment in catch levels and size of mark releases, over the range of population sizes encountered in the historical mark-recapture series. A change in population level of less than 11% from the modal value could be detected at the conventional level of $\alpha=0.05$.

Consequently, the program to quantify smolt production was reinstated between 2009 and 2011 to provide estimates for comparison with historical production. In two of these years (2009 and 2011) smolt production was significantly greater than in any year of the previous series: particularly in 2009 when coho smolts increased by 111% over the migration in 2003 . However, in 2010 the output was not significantly greater than that of the modal year (40,391 versus 41,890)

Historically, the largest contribution made by the side channels to overall production in the Englishman system occurred in 1998 (25%). In contrast the recently enhanced Clay Young channel contributed an average of 40% to smolt outmigration between 2009 and 2011. The relative contribution in these years varied only slightly (s.d. 3.8), suggesting that channel production as well as that in the wider watershed was influenced by similar exogenous factors.

Entre 1989 et 1998, on a aménagé deux chenaux latéraux dans le chenal de la rivière Englishman. Une série de programmes annuels de marquage-recapture ont ensuite été effectués entre 1998 et 2005 pour estimer la contribution saumon coho de ces chenaux à l'émigration globale de la rivière Englishman. Alors que les méthodes variaient d'un programme à l'autre, un simple estimateur de Petersen constituait le calcul du choix dans la majorité des années, y compris certaines incorporant la stratification temporelle (PPE). Les estimations non ajustées (pour les longueurs de cours non échantillonnées) de la taille de la population de saumoneaux se situaient entre 29 238 (2001) et 47 591 (1999). Les comparaisons entre ces années indiquent que l'écart significatif par rapport à la valeur modale (41 890 en 2003) ne s'est produit que dans le cas des niveaux de population les plus bas, en 1998 (32 481) et en 2001.

L'utilité de l'ensemble de données pour fournir une base de référence pour l'évaluation de la stratégie subséquente d'amélioration du canal pour le canal Nature Trust en 2007 a été examinée en utilisant une approximation normale de la distribution binomiale. La méthode s'est révélée relativement insensible à l'ajustement des niveaux de capture et de la taille des rejets de marques, par rapport à la gamme de tailles de population rencontrées dans la série historique de marquage-recapture. Un changement du niveau de population de moins de 11% par rapport à la valeur modale a pu être détecté au niveau conventionnel de $\alpha = 0,05$.

Par conséquent, le programme de quantification de la production de smolts a été rétabli entre 2009 et 2011 afin de fournir des estimations à des fins de comparaison avec la production historique. Au cours de deux de ces années (2009 et 2011), la production de smolts était significativement plus élevée que dans n'importe quelle autre année de la série précédente, particulièrement en 2009, lorsque les saumoneaux cohos ont augmenté de 111% par rapport à la migration de 2003. Cependant, en 2010, la production n'était pas significativement supérieure à celle de l'année modale (40 391 contre 41 890)

Historiquement, la contribution la plus importante des canaux latéraux à la production globale dans le système anglais a eu lieu en 1998 (25%). En revanche, le canal Clay Young récemment amélioré a contribué en moyenne 40% à l'émigration des smolts entre 2009 et 2011. La contribution relative de ces années n'a varié que légèrement (écart-type 3,8), suggérant que la production des chenaux ainsi que celle du bassin versant par des facteurs exogènes similaires.

1.0 INTRODUCTION

Declining escapements of coho and other anadromous species in the 1980's stimulated development of the Englishman River Salmon Maintenance Plan (Hurst 1988) to address limiting factors on productivity, such as extreme fluctuations in seasonal flows that resulted in lack of summer off-channel rearing areas and a paucity of winter low velocity refuge areas for pre-smolts (Miller 1997). This, in turn, led to the construction of the Weyerhaeuser Channel in 1989 (initially named the MacMillan Bloedel Ltd. Channel) to increase the amount of side-channel habitat in the Englishman system. A second channel, the Nature Trust Channel (then Fletcher Challenge Ltd. Channel and subsequently Timber West Channel), was constructed in 1992.

The functionality of these channels was examined in a number of population estimates of juvenile coho and other species, produced in the 1990's. However, these studies employed different methodologies and were difficult to compare directly (Miller 1997). As well, there was no way to compare the contribution of the channels to overall production in the river. The first directed efforts to quantify the contribution of channel reared coho smolts to the Englishman system were made in series of projects initiated in 1998, using mark-recapture.

In 2001, the Englishman River was selected by the Pacific Salmon Endowment Fund Society (PSEFS) as one of the watersheds to be the focus of strategic recovery planning. An essential part of recovery evaluation is development of annual baseline data on coho and steelhead smolt abundances to permit assessment of trends in stock dynamics. The Englishman River Watershed Recovery Plan (ERWRP; Bocking and Gaboury 2001) initiated a series of programs to address these issues through the Community Fisheries Development Centre and local fisheries stream stewards. From 2002, these studies were ratified by ERWRP and funded by PSEFS. These programs have been similar in design and have produced a series of population estimates for juvenile coho migration that form a baseline dataset to identify trends in stock dynamics. As part of the planning for the extension of the Nature Trust side-channel, it became necessary to assess the utility of the existing data to detect and quantify resultant changes in the productivity of the Englishman system.

2.0 STUDY AREA

The Englishman River flows from Mount Arrowsmith north-east for 28 km to enter the Strait of Georgia just south of Parksville, on Vancouver Island (Fig 1). It drains a watershed of approximately 324 km^2. The Englishman River primarily supports runs of coho *(O. kisutch)* and chum *(Oncorhynchus keta)*, with less numerous escapements of chinook *(O. tshawytscha)*, pink *(O. gorbuscha)*, sockeye *(O. nerka)* steelhead *(O. mykiss)*, and anadromous cutthroat trout *(O. clarki)* (Brown et al. 1977). Anadromous fish can access 15.7 km of mainstem, up to the natural barrier of the Englishman River

Falls. Additional anadromous fish habitat is provided by tributaries that increase the accessible length to 31 km (Decker et al. 2003). Among these, Centre Creek is a major contributor at 5.2 km long, representing approximately 17% of the total linear habitat.

Two constructed side-channels initially provided 950 m (Weyerhaeuser) and 1,380 m (Nature Trust) of low gradient habitat in the lower 7 km of river. The Weyerhaeuser Channel is located approximately 6 km upstream from the estuary, on the south bank of the mainstem. It was constructed in 1989, primarily to create summer and winter rearing habitat for juvenile coho. The initial constructed length was 600 m: overall length was extended in 1998 and 2 spur channels were added for an overall wetted area of 6,000 m^2. The Nature Trust channel flows into the mainstem from the north bank, 1 km further upstream. It provided 17,709 m^2 of low gradient (0.5%) habitat. Both channels derive flows from groundwater upwelling as well as controlled intake of river water. In combination, these channels represented a substantial contribution to coho production in the Englishman River system, with estimates ranging from 10% (2003, Schick and Decker 2004) to 25% (1998, Decker et al. 2003).

In 2007 the Nature Trust channel was extended by 2.9 km, bringing the total available rearing habitat to 7.44 ha. This channel was re-named the Clay Young channel in 2009.

3.0 METHODS

3.1 Historical studies 1998 to 2005

The basic elements of programs initiated in 1998 enabled an estimate of total coho smolt population size from a simple Petersen mark-recapture estimator, using catch data from two rotary screw traps (RSTs) in the lower Englishman River (Decker at al. 2003). Marks were released in conjunction with enumeration of a substantial portion of the smolt outmigration from the Nature Trust and Weyerhaeuser side-channels and, from 2001 to 2004, from Centre Creek, a natural tributary. Permutations of the design have included stratification of mark releases by release site only (1999) and with the inclusion of temporal (release period) stratification, analyzed with a pooled Petersen estimator (PPE) and the use of a maximum likelihood estimator after Plante (1990) and as used by Arnason et al. (1996) in their Stratified Population Analysis System software package (SPAS). Generally, a series of estimates of population size were obtained from geographical stratification (release and recovery combinations), and, in a majority of years, the population estimates have been obtained by pooling the temporal strata (release periods). Details of the various programs and the resultant estimates of population size are provided in the following material.

3.1.1 The Simple Petersen Estimator

Decker et al. (2003) provide estimates of population size for the years 1998, 1999 and 2001. Estimates for the years 2002 and 2003 were provided by Schick and Decker (unpubl. data) and for 2004 and 2005 by Taylor (unpubl. data). All estimates were derived from the Chapman (1951) modification of the simple Petersen estimate. Estimation for population size is:

$$N_1 = (M+1)(C+1) / (R+1) \quad (1.1)$$

$$Var(N_1) = (M+1)(C+1)(M-R)(C-R) / (R+1)^2(R+2) \quad (1.2)$$

$$95\% \ CI\ (N_1) = \pm 1.96 \times Var\ (N_1) \quad (1.3)$$

where:

M = number of marked smolts released from two side-channels

C = number of marked and unmarked smolts recovered at the RST(s)

R = number of marked side-channel smolts recovered at the RST(s)

In years that incorporated temporal stratification, stratum totals were summed to estimate N_1.

3.2 Contemporary methods

In contrast to the historical series, although first adopted in 2005, the simple stratified M-R technique of Carlson et al. (1998) formed the sole method used in studies conducted between 2009 and 2011. The design simplified both the marking protocol and the resultant count of recoveries as well as lowering personnel costs. For the first time since 1998 only one RST was deployed, since, for a given trap efficiency, calculation of the number of marks required for the program could be calculated in advance, as described below.

The stratified estimator requires the application of unique mark types within designated marking periods to provide an estimate of capture probability (trap efficiency) over time, so that variation in efficiency can be addressed within the assumption of reasonable consistency in strata. This approach requires temporal stratification such that each trap efficiency trial is discretely paired with one capture period. An important element in planning was to determine the number of marks that must be released in order to achieve an appropriate level of accuracy for desired precision. Data from historical programs was initially used to generate the necessary parameters to calculate the required sample size for mark releases per stratum. After 2009, each previous year's migration pattern was used for this purpose.

3.2.1 PPE Estimation method

Strata estimates are from:

$$\hat{N}_h = \frac{(n_h + 1)(M_h + 1)}{m_h + 1} - 1 \qquad (1.4)$$

where

$\hat{N}_h$ = estimate of population size for stratum h

M_h = number of marked smolts in stratum h

n_h = number of smolts in the RST catch in stratum h

m_h = number of recaptured marks in stratum h

Total smolt abundance is given by:

$$\hat{N} = \sum_{h=1}^{L} \hat{N}_h \qquad (1.5)$$

Given that predicted release of marks plus total catches in any RST was expected to be less than the anticipated population of smolts, the result is an approximately unbiased estimate.

The tally of marked smolts from RST catches represents sampling without replacement and, hence, the distribution of m_h for ranges of M_h and n_h, is hypergeometric. However, for populations greater than 100, simpler distributions, such as the binomial and normal, are satisfactory approximations (Robson and Regier 1964). Given the very large smolt population size, the normal approximation to the variance for $\hat{N}_h$ is adequate, in the form:

$$\text{v}(\hat{N}_h) = \frac{(M_h + 1)(n_h + 1)(M_h - m_h)(n_h - m_h)}{(m_h + 1)^2 (m_h + 2)} \qquad (1.6)$$

and the overall variance is:

$$v(\hat{N}) = \sum_{h=1}^{L} v(\hat{N}_h) \qquad (1.7)$$

(see Seber 1982:p60 for conditions to satisfy an approximately unbiased estimate of variance).

Approximate 95% confidence limits for $\hat{N}$ are:

$$\pm 1.96 \sqrt{v(\hat{N})} \qquad (1.8)$$

Consistency in the capture efficiency of the RSTs through time was examined using a χ^2 contingency test. Randomness of the marking sample was tested by comparing the frequency distributions of marked and unmarked coho in size classes of 10mm (65 –

105mm), using a $\chi 2$ goodness of fit test after Seber (1982: p74). Similarly, size selective catchability was tested by comparing the distributions for recaptured and not recaptured smolts ($\chi 2$ Seber 1982: p71).
The precision of the estimate was assessed using the parametric method described by Carlson et al. (1998). The number of recaptures in each stratum (m_h) was treated as hypergeometrically distributed with parameters { $\hat{N}_h$, M_h and n_h }. One thousand random variates m_{jh} were drawn from the hypergeometric distribution using Systat© and used to calculate $\hat{N}_{jh}$ from equation 3. The precision of the estimate of population size was calculated as bias-corrected percentile confidence intervals (Efron and Tibshirani 1993), where:

$P_{UPPER/LOWER} = \Phi(2Z_O \pm 1.96)$ following calculation of the constant Z_o (p185).

3.2.2 Calculation of mark releases

An appropriate goal for the level accuracy and precision was based on the recommendation of Robson and Regier (1964) for fairly accurate management work: an acceptable level of error is ±25% to be exceeded not greater than 5% of the time (α=0.05). A large number of smolts were expected to be available in each year of the study, smolt numbers were not anticipated to be a limiting factor in any but the initial and final strata. The total relative error (r_h) was set at ±15% for 95% precision, and the calculated number of marks required to achieve this target was considered to be a minimum for the program.

Strata totals from the previous migration were used to estimate the proportion of the population encountered in each time period (ϕ_h) : a total of 5 strata were anticipated for a provisional program duration of early April 17 to early June. A conservative capture efficiency of 7.5% was assumed for the RST. Assuming a constant relative error (i.e. $r_1 = r_2 = ... = r_L$) then the expected stratum relative error (r_t) was estimated from:

$$r_h = \frac{r_t}{\sqrt{\sum_{h=1}^{L} \phi_h^2}} \qquad (1.9)$$

and the number of marks required for release per stratum was calculated from:

$$M_h = \frac{K}{e_h(100)} \qquad (1.10)$$

where K is a constant described by the power function y=3E+6x$^{-1.8893}$ constructed for α=0.05 from data given in Carlson et al. (1998).

3.3 Comparing the historical studies 1998 to 2005

Since the basic data from the various years are easily accessible (Table 1), the estimates of population size can be compared using the goodness-of-fit methods of Chapman (1951).

Designating the populations to be tested as N_a and N_b, and from the notation in Section 3.2.1, then (after Seber 1973 p.121) let:

$$\tilde{N} = \frac{\lambda_a^3 m_{2b} u_{1b} u_{2b} + \lambda_b^3 m_{2a} u_{1a} u_{2a}}{m_{2a} m_{2b} [\lambda_a^2 u_{1b} u_{2b} + \lambda_b^2 u_{1a} u_{2a}]} \tag{1.11}$$

where $\lambda_a = n_{1a} n_{2a}$, $u_{1a} = n_{1a} - m_{2a}$, $u_{2a} = n_{2a} - m_{2a}$, etc.
Defining:

$$T_1 = \sum \frac{(m_{2c} - \lambda_c / \tilde{N})^2}{\frac{\lambda_c}{\tilde{N}} (1 - \frac{n_{1c}}{\tilde{N}})(1 - \frac{n_{2c}}{\tilde{N}})} \tag{1.12}$$

and

$$T_3 = \sum \frac{2(m_{2c} - \lambda_c / \tilde{N})^2}{\frac{\lambda_c}{\tilde{N}} (1 - \frac{n_{1c}}{\tilde{N}})(1 - \frac{n_{2c}}{\tilde{N}}) + \frac{m_{2c} u_{1c} u_{2c}}{\lambda_c}} \tag{1.13}$$

in both cases ∑ denotes summation for both population values, c = a, b.

For large values of N, T_1 and T_3 are approximately distributed as chi-squared with 1 degree of freedom. Then, testing the null hypothesis H_o: $N_a = N_b$ we reject H_o at 100(α)% for values > $\chi^2_{(0.05,1)}$. T_1 and T_3 were used for cases where the values of λ_a and λ_b varied widely, and moderately, respectively (Appendix 1). None of the comparisons generated values of λ that were equal.

A further set of comparisons was run using the normal approximation to the binomial distribution. This provided a check on the goodness-of-fit tests and enabled a relatively straightforward estimate of the boundary along which a significant difference in population size could be just detected, for various levels of n_1 and n_2.

$$z = \frac{|m_{2a} - m_2 p_o| - 1/2}{\sqrt{(m_2 p_o q_o)}} \tag{1.14}$$

where $m_2 = m_{2a} + m_{2b}$ and $p_o = \lambda_a / (\lambda_a + \lambda_b)$, $p = 1 - q$
For the null hypothesis H_o: $N_a = N_b$ the rejection region for the appropriate tail of the distribution is $Z \geq z_a$ for α=0.05 (95% confidence level).

4.0 RESULTS AND DISCUSSION

The various estimates of population size, prior to 2005, were published as adjusted estimates, to account for the unsampled portion of the Englishman River below the sampling/recapture point. However, the legitimacy of this transformation is doubtful, given that only Shelley Creek contributes a limited amount of rearing habitat below the RST locations. Consequently, all analyses were performed on the original, unadjusted estimates; these numbers are provided along with the adjusted estimates in Table 1. In a majority of years, it was possible to construct estimates from combinations of mark releases, as well as total marks. In each case the authors selected the most appropriate final estimate for the year, and no attempt was made to incorporate the additional estimates into the current analyses.

The initial program in the series, conducted in 1998, utilized a single rotary screw trap (RST) in the lower Englishman River, positioned in the location of the LRST(lower trap) in subsequent programs (Decker et al. 2003). Full stratification, temporally by week and geographically by marking location, was used. However, the final maximum likelihood estimate (MLE) was accompanied by extremely wide 95% confidence limits (33,531 ± 31.6%). Despite potential bias from variability in the consistently low capture probabilities (range 0% to 3.3%), the similar PPE (34,578 ± 14.9%) was adopted for the comparative analyses discussed below; the magnitude of the population estimate is of less importance than the associated error in this case. Both estimates were constructed using the total marks released.

In the following year, two RSTs were employed at the 1998 location and catches in the two RSTs were combined to construct the estimate. Consequently, the final estimate for 1999 was derived from the PPE (50,622 ± 11.6%) using all mark releases. The 2001 program combined the mark type used for the constructed channels and added a unique mark for the release of smolts from Centre Creek, used for the first time in the program. In this year an RST was positioned at an upstream (URST) location and marked smolts were released from this site. The final estimate (31,005 ± 3.6%) achieved a high degree of accuracy, due to the large number of marks released (10,559) in conjunction with increased trap efficiency at the LRST (17.6%) substantially greater than in any other year. This estimate was generated from total marks applied and catches at the LRST.

In 2002, discrete marks were applied to smolts from the side channels, Centre Creek and the URST. A PPE of 44,303 ± 9.7% was calculated from releases at the URST with recaptures at the LRST (Schick and Decker 2003). This estimate was smaller and had poorer accuracy than that using all marks (49,215 ± 3.9%) but was chosen on the basis of possibly variable capture probabilities for marked and unmarked smolts. This was not assessed in the study, but was thought to have resulted from differences in migration timing between marked and unmarked smolts.

Stratification was again used in 2003, with three release periods identified through different marks applied to smolts from the channels (Schick and Decker 2004). Marks released from the URST were not differentially marked by time period and the authors state that these releases were used to form the best estimate (PPE 44,417 ± 8.2%.), although this figure actually corresponds to the use of all marks (8,210) with the catch at the LRST (2,203) for recovery of 431 marks (Table 2); the calculation from the URST mark releases gives a similar estimate of 41,085 with wider CI of ± 14%. The initial, more accurate, estimate has been retained for the current analyses.

Both temporal and geographical stratification was attempted in 2004, with the channels and Centre Creek, as well as the URST catch receiving a distinct mark that changed in each of 6 application strata (Taylor 2004). Unfortunately the design was overly complicated from a field perspective, and the final analyses again provided only a PPE. The total number of marks released was larger than in any previous program (15,426), and the best estimate was calculated from all releases and the catch data from the LRST (41,331 ± 8.9). However, some accuracy was sacrificed by basing the estimate on the LRST catch total (1,624) rather than the more efficient URST catch of 4,138. The former directly included migrants from the lower river in the LRST catches, avoiding adjustment for the unsampled mainstem smolt contribution.

The primary mark releases in the 2005 program originated only in the Nature Trust channel to simplify the study design. Two RSTs were again used in this program, with channel and URST releases identified by distinct marks that alternated through 6 release strata (Taylor 2005). Capture probabilities for channel marks were consistent among strata at the LRST. Therefore, the PPE was unbiased and similar to the stratified estimate. The latter was adopted as the best estimate of migration (42,701 ± 12.5) but the PPE has been used in the present analysis (42,904 ± 12.1). Both of the estimates, above, are unadjusted for mainstem length.

4.1 Comparisons among historical years

There are two objectives to making a series of comparisons among the years of the program. First, it is necessary to determine the degree of temporal variation to ensure that the data series encompasses a reasonable sub-set of the natural fluctuation of the Englishman River coho stock. Secondly, the utility of future comparisons will depend on the degree to which the study design can discriminate population variation at a desired level of precision: set here at a conventional 95%.

In general, while there have been a number of change to the program design since 1998, the final estimates are equivalent in terms of the methods used and can be directly compared. However, some alterations and practical challenges within the program have had a direct but unknown effect on specific estimates. Changes to the definition of a smolt adds an unknown degree of bias: smolt size was set at > 75mm in 1998 and 1999, reduced to > 70mm in 2001 (Decker et al. 2003), all coho were considered to be smolts in 2002 (Schick and Decker 2003), size was unspecified in 2003 (Schick and Decker 2004) and re-set to > 65 mm in the final 2 years (Taylor 2004 and 2005). The degree to which inclusion of all juveniles increased the estimate in 2002 relative to the most restrictive definition in 1998 and 1999 cannot be assessed. However, it may not represent the largest source of error. Flooding in 2005 may have resulted in an underestimate of the migration, but the extent of bias in the estimate, if any, is unknown. Other years, in which sampling ended prior to the cessation of smolt movement (2002 – 2004), are anticipated to have suffered minor error from this source, since remnant population levels were already low by the conclusion of sampling. In the following analyses, these effects have been assumed to have been small enough to be accommodated within the confidence bounds of the estimates.

The various estimates and their 95% confidence intervals are illustrated in Fig. 2. In general, Englishman River smolt migrations were fairly consistent over the years, with the exception of lower numbers in 1998 and 2001. The CV over the unadjusted series was an acceptable 15.9%, Consequently, the first objective appears to have been met by the range of migration sizes assessed (29,238 to 47,591, mean 38,857). For the second objective, it is necessary to set a suitable parameter for comparison over future years.

While the entire matrix of estimates could be compared over the time series, it is only necessary to choose a representative value against which the range of population sizes can be assessed. An appropriate basis for comparison is the modal value (2003), representing the most frequently encountered migration size. The, alternative, median value (2002) was so similar to the mode (41,783 and 41,890) that all comparisons are equally valid for this parameter. Both methods of calculation (goodness-of –fit and normal approximation) found the 1998 and 2001 population levels to be the only significant departures form the modal estimate (Table 2), although the 1999 migration was almost large enough to qualify (z=-1.65, p=0.05). Parameters for the calculation of the goodness-of-fit tests are provided in Appendix 1.

4.2 Precision of the historical data series

In order to assess the effect of proposed habitat enhancement on the Englishman River, the sensitivity of the method used to provide migration estimates is of importance i.e. the level of precision achieved will influence the population size at which departure from the baseline becomes significant. For example, at a basic level, the inherent bias in the simple Petersen estimator is approximately $100e^{-n1n2/m2}$ so that the estimate is unbiased only when n_1 (marks) times n_2 (catch) exceeds the population size by a factor of 3 or 4 (Robson and Regier 1964). This does not affect the present series of estimates, all of which are < 1/100 of $n_1 * n_2$. However, accuracy for a given level of precision is also a function of the number of marks and the subsequent catch. Appropriate levels of accuracy can be determined for combinations of n_1 and n_2 so that the required degree of effort to improve the amount of error in the estimate can be assessed and designed for (Robson and Regier 1964).

The boundary for detecting a significant difference with 95% confidence was assessed for the range of parameters encountered in the annual studies. Numbers of marks released (n_1) ranged from 15,426 in 2004, to 3,015 in 2002 (Table 1). Catches (n_2) were also variable, from only 545 in 1998 to 5,135 in 2001. The population sizes that were just significantly less than the modal year ($\Phi(z) = P(Z \leq z = 0.0495)$ were calculated using catches fixed at 2,500 for variable mark releases (Appendix 2a) and with marks fixed at 6,000 and varying catches between 2,500 and 5,000 (Appendix 2b). Similarly, population levels that just exceeded the mode by a significant amount are shown in Appendix 3a and b).

These calculations indicate that the current program can detect a significant difference in population size within the range - 9% to - 11% of the estimate for the modal year, for reductions in migration size, and from +11% to +13% when larger migrations occur. This equates to limits of 37,951 to 46,406 for the values of marks released and catches examined (Appendices 2 and 3). In order to increase the resolution of the method, larger numbers of marks, combined with greater catch efficiency would be required. However, the method is relatively insensitive to such increases, with release of 15,000 marks (achieved in 2004) resulting in only 0.5% improvement in resolution (+10.8% in Appendix 3 to +10.3%). The same result would be achieved by increasing the marks released to 10,000 and catches to 3,750, which may be a more realistic target. However, it is not necessary to deviate from the normal mark and catch rates other than to improve the precision of the annual estimate. Comparisons with the existing data series are sufficiently accurate to discriminate relatively small changes from the mode. For example, the total output of smolts from Nature Trust channel has exceeded 11% of the system wide coho population in 5 out of 7 years of the historical dataset. Consequently, if an increase of this magnitude (approximately, 4,600 smolts) is attributable to a new enhancement strategy, we can prescribe a significant level of enhancement with 95% confidence. Any larger increase would be detectable at a higher level of precision e.g. a 15% increase (to 48,171 smolts) would be significant at the 99% level.

4.3 The contemporary data series

In each of the 2009 to 2011 years there was significant temporal variation in capture probability as indicated by a chi-square test of homogeneity (df range 3 to 5 $p<0.001$ in all cases). Consequently, coho smolt abundances were reported as a stratified mark-recapture estimate. However, the equivalent PPE abundances were similar in each case and bias, determined through the parametric bootstrap procedure of Carlson et al. (1998), was also small in all cases (Table 3).

In 2009, although the program start was delayed by unusually cold weather, smolt densities in the Clay Young channel substantially exceeded historical levels. The total count of juvenile coho from the Clay Young Channel was 35,160 individuals, of which 7,539 (21.4%) were marked for population estimation.

The estimate of total smolt numbers, including the channel population, was 85,467 (95% CI 78,241- 92,692); the associated error for this estimate (coefficient of variation CV 4.3%) far lower than that targeted in pre-study planning. Increased precision stemmed from the reasonably high capture probabilities from the RST (mean 6.7%) in conjunction with a larger than required release of marks from the channel in most periods: releases totaled 7,539 smolts versus the design minimum of 5,295. The PPE used for comparison with the historical data was potentially an overestimate, but this is of little consequence since the 2009 outmigration, by either estimate, was more than double that of 2003.

In 2010, sampling of the mainstem migration was delayed beyond the start of movement of smolts from the channel. This resulted in an underestimate of smolt abundance, population size being reported as 42,038 467 (95% CI 33,688 – 50,387) by Taylor and Wright (2010). Interpolation of early, missed, data based on the 2009 outmigration, resulted in a total of 44,083 (95% CI 35,672 – 52,493) smolts (CV 9.7%). Bootstrapping produced close agreement with the higher total (44,312 smolts 95% CI 36,073 – 62,994) and indicated that bias was low although the precision of this estimate was lower (bias corrected CV 15.5%). Capture probabilities were very high (range 11.3% to 18.5%) with the exception of the final stratum (1.2%) which may have reflected re-adjustment of the RST to fish in lower water levels in June. This value was largely responsible for the wide confidence interval around the estimate. The PPE estimate from the interpolated data was 40,391 with the expected tighter confidence interval (± 3,443 individuals, ± 8.5%) and this figure has been used in the following analysis.

Of the 18,044 smolts collected from Clay Young Channel in 2010, 18.1% (3,270) were marked. The contribution of Clay Young channel smolts to overall production in the system was 45.9%, based on the PPE estimate of abundance and 42.0% of the stratified estimate.

The 2011 program achieved good precision (CV 5.2%) from the release of 4,788 marked smolts with an overall capture efficiency of 9.7% (range 4.0% to 10.4%). Overall emigration from the Englishman system, during the study, was estimated to be 57,498 ± 5,851 smolts (± 10.2%), of which 36% were contributed by the channel. However, temporal stratification was incomplete, with recaptures of marked smolts from earlier release periods encountered on two occasions. This required the combination of two strata on the first occasion and pooling of strata 4 to 6 on the second. Consequently, the equivalent PPE was very similar to the stratified estimate, 57,548 (95% CI 52,073 – 63,023).

4.4 Comparison of contemporary and historical years

In two of the contemporary years the estimates indicate a highly significant increase in outmigration from the Englishman River (Table 4). In 2009 the estimated outmigration of 85,467 was an increase of 79.6% over the largest historical year (1999 47,591smolts) and was 111.4% larger than the modal year. This year was very highly significantly larger than the mode (z=-20.3 p<0.001 Table 4). Smolt density in the Clay Young Channel was approximately 8,580 km^{-1}, far exceeded the previous highest recorded density of 5,451 smolts.km^{-1} in the Nature Trust Channel in 1998 (Decker et al. 2003). In contrast, the 2004 and 2005 smolt densities in the Nature Trust Channel were 4,270 km^{-1} and 2,865 km^{-1}, respectively (Taylor 2005).

While not as great an increase over the modal year (37.4%) the 2011 outmigration (57,548 smolts) was also very highly significantly larger (Z=-4.50 p<0.001). It was also significantly larger than the 1999 outmigration (z=-1.65 p=0.01 Table 4). The total count of juvenile coho from the Clay Young Channel in 2011 was 19,960 individuals. Adjusted for unsampled length, the estimate from the Clay Young channel is 20,499 smolts, or 5,000 smolts.km^{-1}. Due to the very large numbers of chinook juveniles originating from an upstream impoundment, sampling had to be halted before the movement of coho was complete. On the last day of sampling 142 coho smolts were captured, consequently the calculated density is an underestimate of channel production, although likely a minor one. The 2011 production by the channel was comparable to the 1999 estimate, although the overall system production was greater.

In 2010 the PPE of 40,391 was slightly smaller than that of the modal year (41,890) but greater than the lower limit at which we should detect a significant difference (~38,000 smolts Section 4.2). The 2010 and 2003 estimates of abundance are not significantly different. However, the 1999 estimate was 15.1% greater and was significantly larger (z= 2.96 p <0.01). The total count of juvenile coho from the Clay Young Channel was 18,044 individuals: on the last day of sampling 188 were captured, indicating that the outmigration was incomplete and that this total is an underestimate of channel production. Adjusted for unsampled length, the estimate from the Clay Young channel is 18,531 smolts, or 4,520 smolts.km^{-1}. While this density of smolts is high and exceeds the range of estimates provided by Marshall and Britton (1990) for coastal streams (363 – 3,018 km^{-1}) it is only 53% of the 2009 total. It also falls below the 5,451 smolts.km^{-1} recorded from the Nature Trust Channel in 1998 (Decker et al. 2003) but exceeds the 2004 density of 4,270 km^{-1} (Taylor 2005).

5.0 CONCLUSIONS

The extremes of coho smolt migration from the Englishman River, recorded between 1998 and 2005, varied by ~±22% of the modal value, although significant departure from the mode (2003) occurred in only 2 years (1998 and 2001). Estimates of population size that vary from the 2003 estimate by + 11% (~ 4,500 smolts) or by – 9% (~ 3,900 smolts) can be assessed as significant with 95% confidence. The series of estimates conducted between 2009 and 2011 exceeded the above degree of change in two years, respectively increasing by 111.4% in 2009 and by 37.4% in 2011. These changes were sufficient to reflect highly significant increases in coho productivity. In 2010, the PPE estimate was slightly lower than the modal year (-3.6%), although the potentially more precise stratified estimate was greater than the mode. However, population size was not significantly different in either case.

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Table 1. Summary of annual population estimates of coho smolt outmigration and population parameters and estimation variables, 1998 – 2005.

Year	Adj. $\hat{N}$	$\hat{N}$	95% CI	n_1	n_2	m_2
1998	34578	32481	4831	7792	545	130
1999	50622	47591	5513	6862	1691	243
2001	31005	29238	1063	10559	5135	1854
2002	44303	41783	4051	3015	4654	335
2003	44417	41890	3444	3485	2203	186
2004	41331	38627	3436	15426	1624	648
2005	42904	40390	5661	3694	2600	223

Table 2. Comparisons of historical estimates of smolt abundance with the modal (2003) year.

Year	$\hat{N}$	T_3	z	P $(T \geq \chi^2_{.05,1})$	P $(Z \geq z_a)$
1998	32481	7.61	2.46	0.01	0.01
1999	47591	3.31	-1.65	0.07	0.05
2001[1]	29238	20.90	6.77	<0.01	<0.01
2002	41783	0.001	-0.01	0.98	0.50
2004	38627	2.57	1.29	0.38	0.11
2005	42904	0.12	0.27	0.73	0.39

[1] Calculated using T_1 due to the very large difference in λ.

Table 3. Summary of annual population estimates of coho smolt outmigration and population parameters and estimation variables, 2009 – 2011.

Year	$\hat{N}$	95% CI	n_1	n_2	m_2
2009	88536	7225	7539	5964	507
2010	40391	8410	3270	4830	421
2011	57548	5851	4788	4313	358

Table 4. Comparison of the contemporary estimates of smolt abundance ($\hat{N}_C$) with the modal (2003) year and with selected years from the historical data series ($\hat{N}_H$).

Year	$\hat{N}_C$	$\hat{N}_H$	T_3	z	P ($T \geq \chi^2_{.05,1}$)	P ($Z \geq z_a$)
2009 vs. 2003	88536	41890	31.74[1]	-20.28	<0.001	<0.001
2010 vs. 2003	40391	41890	3.42	1.60	0.06	0.05
2010 vs. 1999	40391	47591	3.91[1]	2.96	<0.01	<0.01
2010 vs. 2001	40391	29238	14.44	-4.65	<0.01	<0.01
2011 vs. 2003	57548	41890	7.52[1]	-4.50	<0.001	<0.001
2011 vs. 1999	57548	47591	6.05	-2.32	0.01	0.01

[1] Calculated using T_1 due to the very large difference in λ.

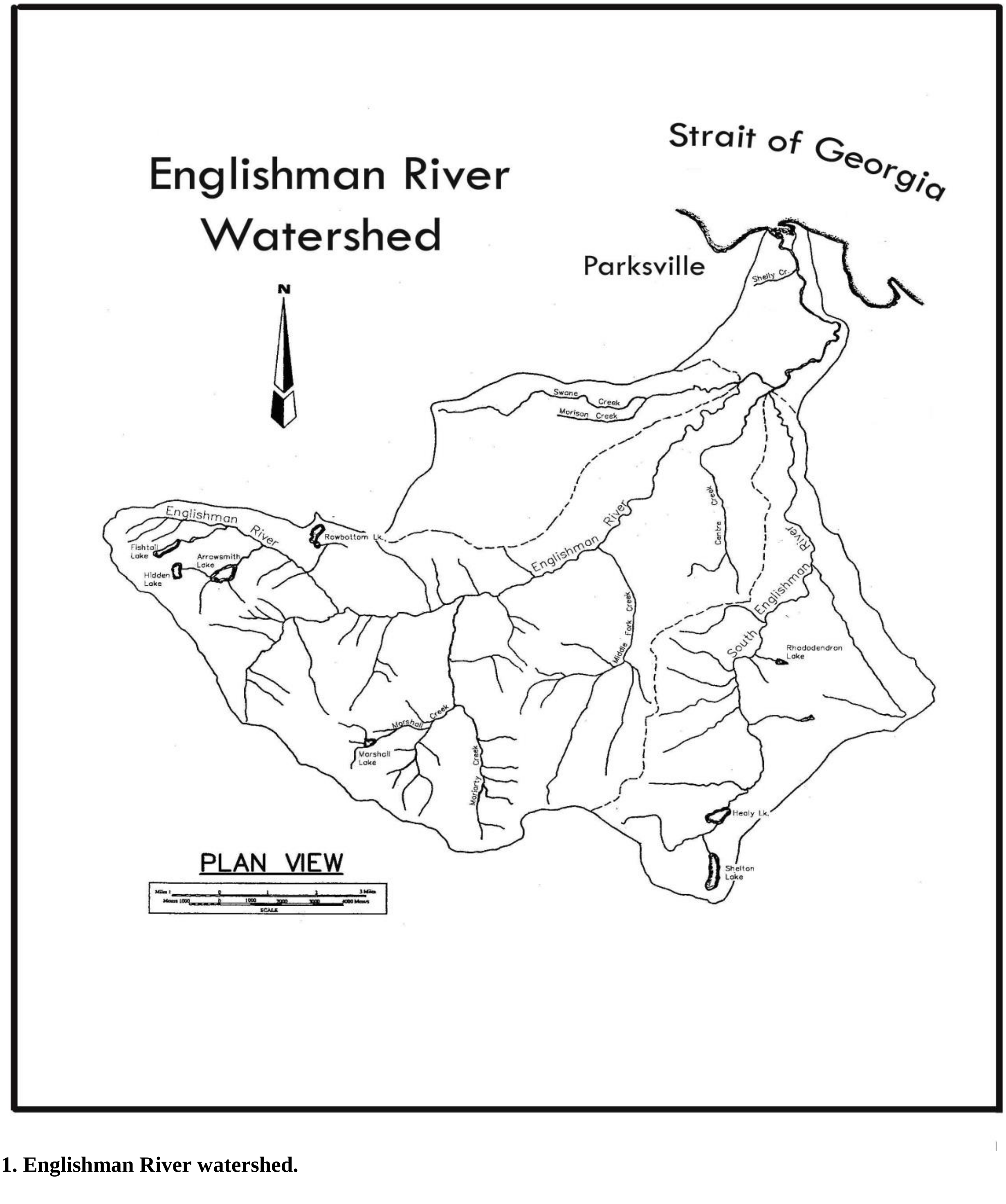

Figure 1. Englishman River watershed.

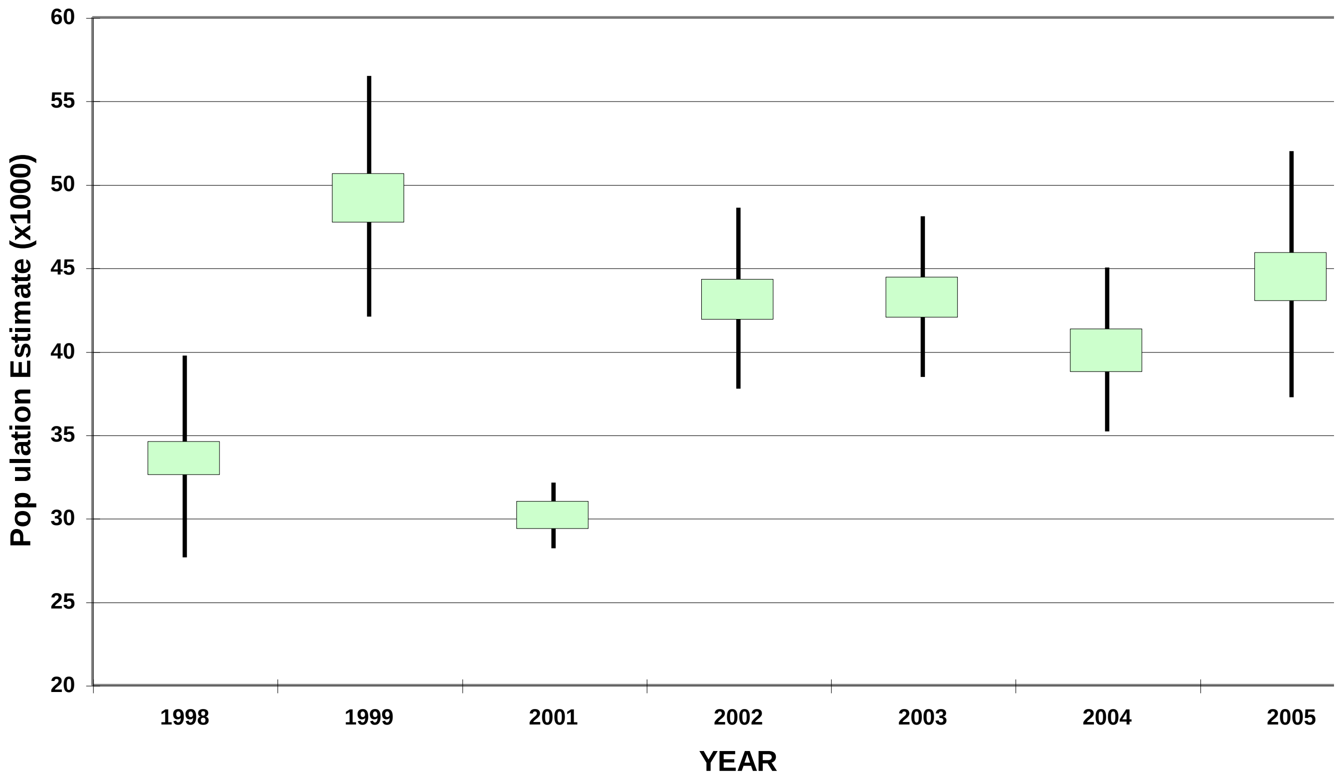

Figure 2. Coho smolt population estimates and their 95% confidence intervals for historic years of 1998 - 2005

APPENDIX 1. PARAMETERS AND SIGNIFICANCE LEVELS FOR COMPARISONS AMONG POPULATION ESTIMATES BY GOODNESS OF FIT AND NORMAL APPROXIMATION TO THE BINOMIAL DISTRIBUTION.

Year	λ_a	u_{1a}	u_{2a}	λ_b	u_{1b}	u_{2b}	$\tilde{N}$	T_3	z	$P(T \geq \chi^2_{0.05,1})$	$P(Z \geq z_{0.05})$
1998 vs 2003	4246640	7662	415	18086630	7779	1772	40171	7.61	2.46	0.01	0.01
1999 vs 2003	11603642	6619	1448	18086630	7779	1772	44116	3.31	-1.65	0.07	0.05
2001 vs 2003[1]	54220465	8705	3281	18086630	7779	1772	31628	20.90	6.77	<0.01	<0.01
2002 vs 2003	14031810	2680	4319	18086630	7779	1772	41932	0.001	-0.01	0.98	0.50
2003 vs 2004	18086630	7779	1772	25051824	14778	976	39826	2.57	1.29	0.38	0.11
2003 vs 2005	18086630	7779	1772	9604400	3471	2377	42318	0.12	-0.36	0.73	0.36
2004 vs 2005	9604400	3471	2377	25051824	14778	976	39562	2.55	1.35	0.11	0.09

[1] Calculated using T_1 due to the very large difference in λ.

APPENDIX 1. VALUES OF $\hat{N} \leq N$ SATISFYING $\Phi(z)=0.0495$ IN COMPARISON WITH THE 2003 (MODAL) ESTIMATE OF 41,890, FOR a) VARYING NUMBERS OF MARKS RELEASED n_1 AND b) VARYING CATCH SIZES n_2. ALSO PROVIDED ARE MARKS RECOVERED m_2 FOR EACH POPULATION LEVEL.

a)	$\hat{N}$	n_1	n_2	m_2	% change
	37630	8000	2500	531	10.2
	37685	8500	2500	564	10.0
	37734	9000	2500	596	9.9
	37779	9500	2500	629	9.8
	37819	10000	2500	661	9.7
	37856	10500	2500	693	9.6
	37951	12000	2500	790	9.4

b)	$\hat{N}$	n_1	n_2	m_2	% change
	37339	6000	2500	402	10.9
	37531	6000	3000	480	10.4
	37675	6000	3500	557	10.1
	37787	6000	4000	635	9.8
	37877	6000	4500	713	9.6
	37951	6000	5000	790	9.4

APPENDIX 2. VALUES OF $\hat{N} \geq N$ SATISFYING $\Phi(z)$=0.0495 IN COMPARISON WITH THE 2003 (MODAL) ESTIMATE OF 41,890 FOR a) VARYING NUMBERS OF MARKS RELEASED n_1 AND b) VARYING CATCH SIZES n_2. ALSO PROVIDED ARE MARKS RECOVERED m_2 FOR EACH POPULATION LEVEL.

a)	$\hat{N}$	n_1	n_2	m_2	% change
	46857	8000	2500	427	11.9
	46780	8500	2500	454	11.7
	46710	9000	2500	482	11.5
	46647	9500	2500	509	11.4
	46590	10000	2500	537	11.2
	46538	10500	2500	564	11.1
	46406	12000	2500	646	10.8

b)	$\hat{N}$	n_1	n_2	m_2	% change
	47280	6000	2500	317	12.9
	47001	6000	3000	383	12.2
	46794	6000	3500	449	11.7
	46636	6000	4000	515	11.3
	46509	6000	4500	580	11.0
	46406	6000	5000	646	10.8