Management reference points for the Thompson and Chilcotin late summer-run steelhead (*Onchorhynchus mykiss*) stock aggregates

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Abstract

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The abundances of spawners returning to the populations of late-summer run steelhead that comprise the Thompson and Chilcotin stock aggregates have declined by 65% and 84% respectively over the last three generations. Spawner abundances for the smaller individual populations are now in the range of one hundred to several hundred fish. The management of the fisheries that impact these stock aggregates requires clearly-defined and technically-defensible abundance thresholds for implementing management actions to reduce the potential impacts of fisheries on the dynamics of these stock aggregates and to ensure that possible actions to re-build declining stocks are initiated in a timely manner. I have applied conventional stock-recruit analyses to the time series of observed spawner and reconstructed recruit data for the two stock aggregates to derive an upper conservation concern threshold, below which abundance management actions progressively reduce the mortality imposed by directed and non-directed fisheries on these stocks, and a lower limit reference point below which no fishery would be permitted and management actions are directed solely to stock re-building. The analyses indicated that the adult stock productivity of the aggregates has declined over the last twenty years, and strongly suggested that decreased marine smolt-to-adult survival was an important factor in the declines. I derived possible management thresholds from both time-varying and time-invariant stock-recruit relations and estimated the effects of observational error on the distributions of these thresholds. Because of the uncertainties in the estimates of spawner abundance and in the estimated mortality imposed by non-directed net fisheries, and because of the uncertain dynamics of such small populations, I advocate the use of the upper 80th percentile of the distributions of possible management thresholds to ensure that there is a high likelihood that the true value will not be exceeded before management actions are undertaken. For the two aggregates combined, I recommend a limit reference point of 727 adults and a conservation concern threshold of 1,950 adults. The values for the Thompson stock aggregate are 431 and 1,187 fish while those for the Chilcotin stock aggregate are 296 and 763 fish respectively. These values may change in the future as additional information on the dynamics of the fisheries is acquired.

Introduction

This report provides a brief rationale for and the computation of abundance-based management reference points for the Thompson and Chilcotin late-summer run steelhead (*Oncorhynchus mykiss*) stock aggregates. I do not consider Mid-Fraser steelhead stocks (Nahatlach, Stein, Seton, Bridge), which lack the necessary spawner-recruit data. The biology of the Interior summer-run steelhead stocks and the fisheries that exploit them are described in Anonymous (1998).

Factors in declines in steelhead abundances

The estimated abundances of spawners for the major stocks in the Thompson and Chilcotin late-summer run steelhead stock aggregates have declined considerably since the 1980s (Fig. 1). This change is part of a widespread, synchronous decrease in the abundances of several species of anadromous salmon from the South Coast of British Columbia which is attributed, in part, to greatly reduced smolt-to-adult survival in the marine environment (Mueter et al. 2002). Direct measurements of smolt survival for the Keogh River (northern Vancouver Island) steelhead population (Ward 2000), as well as coded-wire tag returns for index stocks of coho (*O. kisutch*) originating in the Lower Fraser River (Simpson et al. 2004) and smolt survival data (Grant et al. 2011) for Chilko Lake sockeye (*O. nerka*), confirm widespread declines in marine survival of approximately 3- to 4-fold in the early 1990s; these conditions have continued to the present. The mechanism(s) that caused this abrupt change in smolt-to-adult survival are not known, but the declines are correlated with regional-scale variation in oceanic conditions such as sea surface temperature distributions (Mueter et al. 2005, Pyper et al. 2005) that may influence the location and intensity of the upwelling events that drive primary production in the northeast Pacific Ocean. The current declines in smolt-to-adult survival may result from reduced productivity in the portions of the northeast Pacific where the particular salmonid stocks rear, and the effects of reduced marine productivity on wild stocks may be exacerbated by very large increases in the abundance of hatchery-origin salmon competitors during this period (Ruggerone et al. 2010). The result is greatly reduced stock productivity for many stocks of salmon and steelhead. Long-term (2,200 year) records of indices of salmon abundance suggest cyclic variation in salmon numbers at several time scales (Finney et al. 2002), the short-term

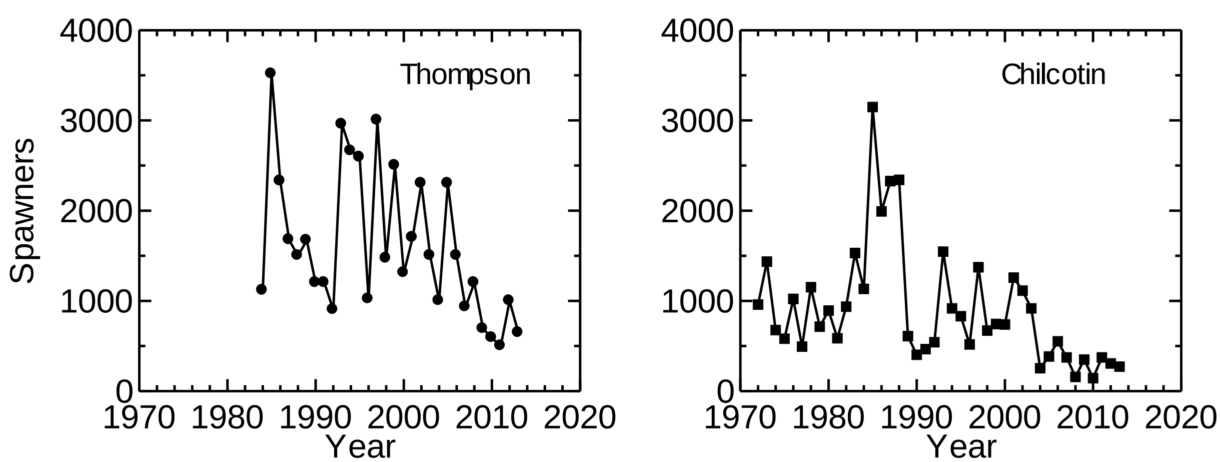

Fig. 1. Estimated number of spawners from the principal stocks of the Thompson and Chilcotin late-summer run steelhead stock aggregates.

variation perhaps driven by periodic climatic phenomena such as El Niño-Southern Oscillation events and/or Pacific Decadal Oscillation events whose effects may be intensified in the current period of climate warming. It is not known whether the current conditions of low stock productivity will persist or whether conditions will eventually revert to a higher production regime.

Fishing mortality is unlikely to be the principal cause of the recent decline in the abundance of the Thompson and Chilcotin steelhead aggregates, although it may have been an important factor in the past. Recreational fisheries for Thompson and Chilcotin summer steelhead have been catch-and-release since 1990, although there is some (generally) low level of catch-and-release mortality (Anonymous 1998, Nelson et al. 2005) which may vary with environmental conditions, gear type, and effort levels, all of which influence the probability of capture and the physiological stress associated with capture and release. Commercial net fisheries for salmon may capture adult steelhead as bycatch during their marine and lower river migrations and impose some level of mortality whose magnitude varies with the same factors as in recreational fisheries. For the Thompson and Chilcotin steelhead aggregates, bycatch mortality is believed to be currently in the range of 10-20% and has generally been below sustainable harvest levels recently (R.G. Bison, BC Fisheries Branch, Kamloops, unpublished). Any level of harvest will, of course, impede stock rebuilding once abundance has declined to a very low level. In-river First Nation fisheries may cause a low mortality in a few cases, either as bycatch or as directed fisheries.

The current regime of greatly decreased marine survival poses significant challenges for the management of low-abundance species such as steelhead. Maintaining a viable fishery while ensuring the persistence of the fished stock becomes an acute problem for steelhead at the currently-observed low values of salmonid smolt-to-adult survival (e.g., about 4% ×÷ 1.7 for Keogh River steelhead; about 3% ×÷ 1.9 for Lower Fraser coho index stocks) because recruitment is near replacement values and the abundances of individual steelhead populations are low, roughly 100 spawners for several populations within the Thompson and Chilcotin management aggregates currently. At such low adult abundances and recruitment levels, stochastic events or uncertain depensatory effects could induce further declines that result in local extirpations.

Management framework

The conventional management of fisheries for steelhead and salmon uses information on stock productivity and adult abundance to define control rules that establish permissible fishing mortality rates and abundance-based trigger points for reductions in (or the elimination of) controllable mortality such as bycatch or harvest. Typically, the management system will incorporate several abundance benchmarks (Fig. 2): a desired abundance state (a “target reference point”, TRP) with an associated range of permissible mortality rates; an upper abundance threshold below which mandatory reductions in controllable mortality rates are applied to assist in rebuilding the population towards the desired state, (a “conservation concern threshold” in our terminology, CCT), and a lower abundance threshold (a “limit reference point”, LRP) below which fishing is not permitted and management activities attempt to rebuild the population above the CCT within a defined time trajectory under normal environmental conditions. The LRP is also intended to provide a substantial buffer against a “high” probability of extirpation (Holt et al. 2009). Between the CCT and the LRP, management actions attempt increasingly to reduce fishing mortality as abundances decline. As abundance declines below the LRP, the viability of a small stock is increasingly at risk. The stock productivity and habitat capacity information that are required to define the

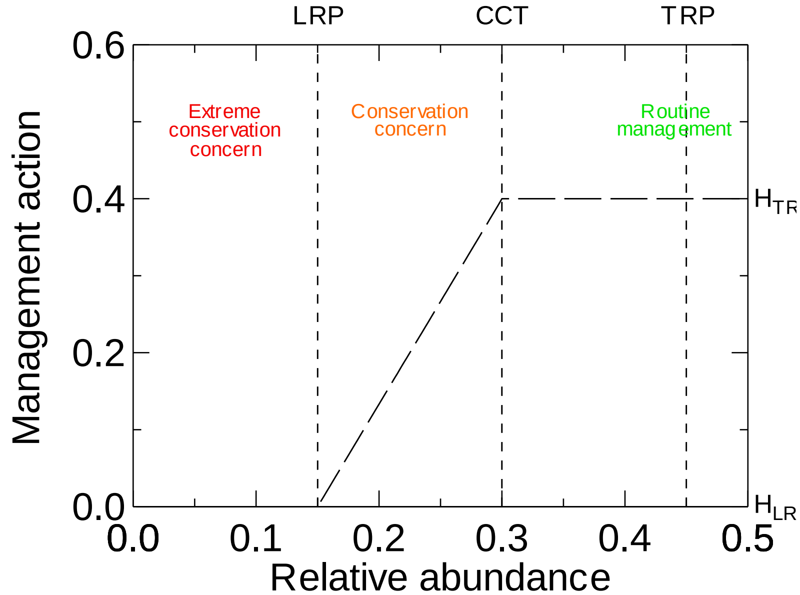

Fig. 2. The elements of an abundance-based precautionary management framework. Three abundance thresholds (the limit reference point, LRP; the conservation concern threshold, CCT; and the target reference point, TRP) force mandatory changes in management actions that are intended to maintain a population within the routine management zone, where sustainable societal benefits can be optimized. Within the conservation concern and extreme conservation concern zones of abundance, management actions are increasingly directed towards promoting population recovery (for example, by reducing fishing mortality rates from H_{TR} to H_{LR}), and potential societal benefits are correspondingly reduced. Population abundance is measured relative to the asymptotic maximum abundance here (see below). The thresholds and mortality rates shown are for illustration only.

thresholds and mortality rates are usually obtained from retrospective analyses of adult stock and recruitment that assume time-invariant relationships and parameter values ("stationarity"). Given information on stock productivity, capacity and environmental variation, the expected performance of alternative management systems can be compared by simulation modelling to identify the system of thresholds and fishery controls that will, on average, best satisfy sets of performance criteria that may include conservation criteria such as fish population size and persistence as well as economic and social criteria related to the fishery (Johnston et al. 2000, Holt et al. 2009, Holt and Bradford 2011).

Target reference points for steelhead stocks are rarely defined explicitly, but are implicitly some very high proportion of the maximal recruitment to maximize recreational fishing opportunities, fishing "quality", and socio-economic benefits, as well as to allow traditional First Nations cultural practices. The conservation concern threshold that is used to signal an unacceptably low abundance and to force mandatory reductions in fishing mortality (or other controllable mortality) has been defined as the spawner abundance at maximum sustainable yield, S_{MSY} (Johnston et al. 2000). Generally there would be no reason to permit recruitment overfishing in a recreational catch-and-release fishery for steelhead because population viability and measures of socio-economic benefits increase with increasing adult abundance. The limit reference point abundance is defined as the spawner abundance from which a stock could recover to the CCT (i.e., to S_{MSY}) within one generation under average environmental conditions in the absence of fishing (Johnston et al. 2000, 2002). Similar definitions have recently been proposed for Pacific salmon (Holt et al. 2009). The intent of this framework is to force management actions that will, on average, permit a stock to rebuild to abundance levels at which it is capable of providing societal benefits within an acceptable time period and before the stock has declined to such low abundance that extirpation becomes a significant risk (Johnston et al. 2000, Holt and Bradford 2011).

Time-varying stock productivity

Time-variation in marine survival alters the productivity of a stock. Effective management benchmarks and control rules become more difficult to define when average production parameters vary through time and future values are uncertain. Where the adult stock productivity varies considerably through time because of large changes in the average marine survival (e.g., regime shifts), there will not be a unique spawner-adult stock-recruitment relationship (SRR). There are several possible approaches to estimate a SRR under such conditions, e.g., Kalman filtering or the use of environmental correlates to account for time-varying stock productivity. In situations where productivity is unknown (common for steelhead populations) or where future variation cannot be predicted, maximal values for S_{LRP} and S_{MSY} can be obtained analytically for certain types of SRR (Johnston et al. 2002; see below) and used as management benchmarks. Alternatively, benchmarks can be defined under long-term average conditions if appropriate data are available (e.g., Holt et al. 2009). Simulation can then examine the performance of various management benchmarks under "realistic" variation in productivity (Johnston et al. 2000, Holt and Bradford 2011). Nevertheless, defining effective management reference point for small, fished stocks under conditions of varying adult stock productivity remains a challenge.

Time-variation in marine survival can still be accommodated within the conventional framework of management benchmarks. Variation in marine survival simply re-scales the density-dependent freshwater spawner-smolt production relationship for steelhead. Data for the Keogh River steelhead population indicate clearly that spawner-to-smolt production is density-dependent while smolt-to-adult survival is density-independent under regimes of both high and low marine survival (Ward 2000). In situations where (1) the adult stock-recruitment relationship is separable into a density-dependent spawner-smolt relationship and a density-independent smolt-to-adult survival and (2) variations in adult stock productivity are largely driven by fluctuations in smolt-to-adult survival rather than by changes in the capacity of the freshwater spawning/rearing environment to produce juveniles, then the spawner-adult SRR is simply a linear re-scaling of the (assumed invariant) spawner-smolt SRR for common functional forms of the SRR (e.g., Ricker or Beverton-Holt SRRs):

Ricker: $$R_t = s \cdot \alpha \cdot S_{t-1} \cdot e^{\frac{-\alpha \cdot S_{t-1} \cdot e^{-1}}{\beta}} = s \cdot \alpha \cdot S_{t-1} \cdot e^{\frac{-s \cdot \alpha \cdot S_{t-1} \cdot e^{-1}}{s \cdot \beta}} = a \cdot S_{t-1} \cdot e^{\frac{-a \cdot S_{t-1} \cdot e^{-1}}{B}}$$

Beverton-Holt: $$R_t = s \cdot \frac{\alpha \cdot S_{t-1}}{\frac{1 + \alpha \cdot S_{t-1}}{\beta}} = \frac{s \cdot \alpha \cdot S_{t-1}}{\frac{1 + s \cdot \alpha \cdot S_{t-1}}{s \cdot \beta}} = \frac{a \cdot S_{t-1}}{\frac{1 + a \cdot S_{t-1}}{B}}$$

where R_t is the adult recruits in generation t, S_{t-1} is the number of spawners in the preceding generation, s is the current smolt-adult survival, α is the average number of smolts produced per spawner as spawner abundance approaches zero, β is the maximal smolt recruitment (i.e., habitat capacity), *a* is the adults produced per spawner as spawner numbers approach zero, and *B* is the current maximal adult recruitment. The effect of variation in marine smolt-adult survival is to alter the realized adult stock productivity, *a* (= $s \cdot \alpha$) and maximal adult recruitment, *B* (= $s \cdot \beta$). Density-dependent variation in recruitment is, however, determined wholly by the ratio of the smolt productivity α to the maximal smolt production (habitat capacity) β. Because of the linear relationship between smolt and adult SRRs, management reference points like S_{MSY} and S_{LRP} can be expressed and monitored either in terms of spawner numbers or the smolts needed to produce the required spawner numbers. The SRR is usually obtained from a time series of spawner counts and estimates of adult recruits (= spawners plus fishing mortalities) but a spawner-smolt SRR will also permit management benchmarks to be determined. It is possible in principle to recover the freshwater (spawner-smolt) SRR from the spawner-adult SRR if smolt survival is known.

Several invariant relationships exist which aid in defining reference points under time-varying stock productivity. The spawner abundances at MSY (S_{MSY}) and at the LRP (S_{LRP}) vary with marine survival and the structural form of the SRR (Fig. 3). S_{LRP} for a given smolt productivity α and a fixed smolt capacity β always has an upper bound as a function of smolt-adult survival *s* but S_{MSY} is a monotonic increasing function of marine survival (Fig. 3). Re-scaling S_{LRP} and S_{MSY} by the smolt capacity β (or the current adult capacity *B*) and re-writing them as functions of adult productivity produces functions that are bounded, with maximal values occurring in the range from 1.5 to 4.0 adult recruits per spawner in the absence of process error (Fig. 4). For a Ricker SRR, the maximum value of S_{MSY} is 0.438 of the (invariant) smolt capacity, β, and occurs at *a* = 2.37 adult recruits per spawner. The maximal S_{LRP} is $0.276 \cdot \beta$ at *a* = 1.65 adult recruits per spawner. These values can be expressed equivalently in spawner numbers as the same proportions of the current (time-varying) adult capacity, *B*. For a Beverton-Holt SRR, the maximal S_{MSY} is $0.25 \cdot \beta$ at *a* = 4.0 adult recruits per spawner and the maximal S_{LRP} is $0.131 \cdot \beta$

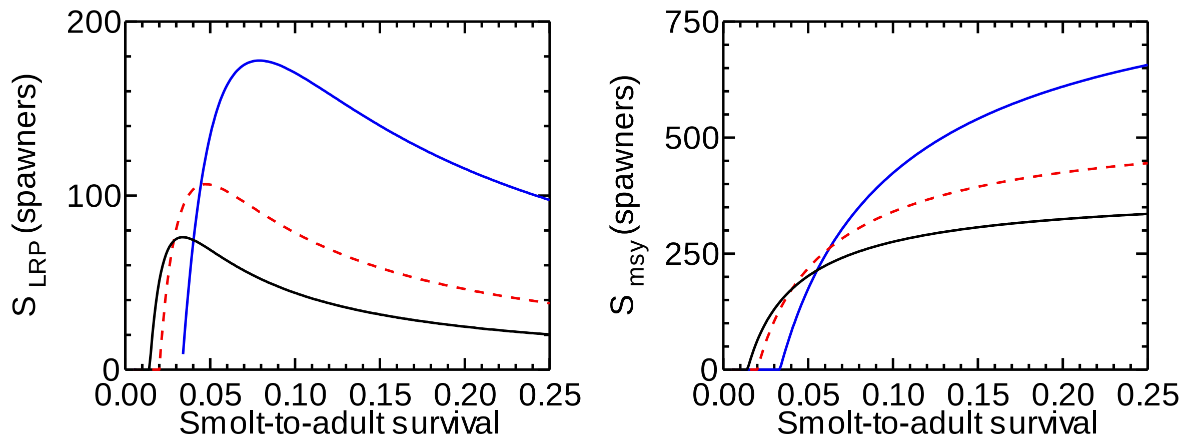

Fig. 3. The LRP and MSY spawner abundances vary with smolt productivity (α) and with smolt-to-adult survival, which together determine the adult productivity. S_{LRP} for a given smolt productivity has a maximum at some value of smolt-adult survival but S_{MSY} is a monotonic increasing function of smolt-adult survival. Data shown are for a Ricker stock-recruit relationship with a smolt capacity (β) of 10 000 and smolt productivities (α) of 30 (solid blue line), 50 (dotted red line) and 70 (solid black line) smolts per spawner. Similar relationships exist for a Beverton-Holt stock-recruit relationship.

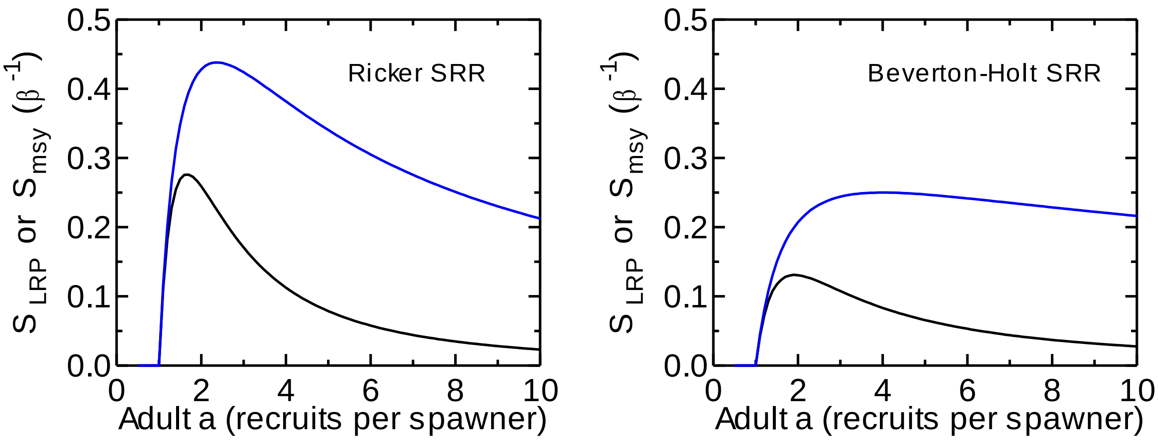

Fig. 4. The number of smolts, as a proportion of the invariant maximal smolt recruitment β, that are required to produce the LRP (black line) and MSY (blue line) spawner numbers at different values of the adult stock productivity a for Ricker (left panel) and Beverton-Holt (right panel) stock-recruitment relations. These functional relationships are identical when expressed in spawner numbers as a proportion of the maximal adult recruitment at the current smolt-adult survival. Note that both the adult productivity a and the current maximal adult recruitment B vary linearly with smolt-adult survival so variation in adult productivity directly reflects variation in smolt-adult survival.

at a = 1.92 adult recruits per spawner (Fig. 4). When adult stock productivity is unknown or varies through time, these maximal LRP and CCT values can be used to set reference points that are approximately correct under conditions of low marine survival such as occur at present, provided the spawner-smolt SRR is known. I use these invariants below to estimate maximal S_{LRP} spawner values from empirical spawner-smolt SRRs for the Thompson and Chilcotin stock aggregates.

While S_{LRP} is always a bounded function of s, S_{MSY} is conditional on the marine survival value at which it is estimated (Fig. 3). Although the proportion of the smolt capacity ($0.438 \cdot \beta$) that is required to produce the maximal S_{MSY} spawners is invariant, the number of spawners that this represents varies with the current smolt-to-adult survival s and increases monotonically with s. The global maximum S_{MSY} value will therefore be determined by the highest survival that has been encountered in the data time-series, or equivalently, by the highest a_t value in the set of Kalman-filtered adult productivity estimates. As s declines, S_{MSY} declines proportionately, which reduces the utility of the current S_{MSY} as a precautionary threshold for small populations. In situations where S_{MSY} varies through time because of variation in marine survival, the appropriate precautionary threshold depends on the intent of the management policy. The general intent of all fishery management frameworks that employ a precautionary abundance threshold like the CCT is to keep the stock at abundances that maintain an acceptable level of socio-economic benefit with little risk of long-term abundance declines. In principle, the performance of various potential CCTs with respect to socio-economic and conservation criteria could be modelled to identify a best performing combination of CCT and fishing mortalities (e.g., Johnston et al. 2000, Holt 2009). In practice, however, the CCT has been often defined as the S_{MSY} under long-term average conditions (e.g., Holt et al. 2009, Holt and Bradford 2011) to maintain the average productivity of the stock. Using the S_{MSY} that is estimated under average conditions as a CCT is risk-adverse under conditions of low marine survival, i.e., it will overestimate the current S_{MSY}, but the strategy performs well in limiting the extirpation risk of small populations in variable environments while producing reasonable socio-economic benefits (Johnston et al. 2000, Holt and Bradford 2011). I use both time-varying and time-invariant SRRs below to estimate S_{LRP} and S_{MYS} from spawner-adult recruit time-series for the Thompson and Chilcotin steelhead aggregates.

Methods

Analytical approach

If the density-dependent term *b* of the SRR can be estimated from a time series of empirical observations on spawners and recruits for a population or stock aggregate, management reference points (S_{LRP}, S_{MSY}) can be computed for any value of adult stock productivity *a*. I determined S_{MSY} numerically in Visual Basic for Applications (Microsoft Corp. 2007) by using SOLVER to maximize the difference between spawners and their average recruits for given values of *a* and *b*, and I similarly used SOLVER to find the S_{LRP} that would then produce S_{MSY} recruits. Average recruitment was corrected for estimation bias (Hilborn 1985, Peterman et al. 2003) although Walters and Korman (2001) argue against this.

I used two methods to determine the density-dependence of recruitment for the Thompson and Chilcotin steelhead stock aggregates. The first method assumed that fish which enter the marine environment at the same time and place experience common effects in a shared marine environment which result in correlations in their survivals. This assumption is supported by the strong spatio-temporal correlations in adult stock productivity seen among different stocks of steelhead (Smith and Ward 2000) and species of Pacific salmon (Mueter et al. 2002, 2005). Furthermore, the smolt-adult survival of Keogh River steelhead is significantly correlated with the mean survival of Lower Fraser River coho index stocks (Inch Creek, Chilliwack River, Salmon River; Simpson et al. 2004, DFO 2009) for the same smolt year (r = 0.59, N = 26, P = 0.001). These observations suggest that it is reasonable to use the average smolt-adult survival measured for Lower Fraser River coho index stocks as a time-varying environmental correlate in a standard SRR analysis for co-migrating Thompson or Chilcotin steelhead smolts. Note that this does not assume that the coho and steelhead smolts experience the same marine survival, only that their survivals co-vary. I assume a Ricker SRR. This makes conservative assumptions about the resilience of the stock aggregate and also allows the parameter estimation to be done by standard linear regression methods on natural-log-transformed data. Although a BH SRR is a plausible model and Ricker and BH SRR generally have roughly equal support in empirical comparisons of SRR structure, the BH model has less tractable statistical and biological properties. The resulting Ricker model is:

$$\ln(R\ /S) = \ln(s) +\ \ln\left(\alpha\right) + b\cdot S +\ \nu$$

where *R* is the estimated number of adult recruits produced by the spawner abundance *S*, *s* is the average survival of coho smolts from the Lower Fraser River index systems for the smolt year of the dominant steelhead smolt age group, α is the average smolt productivity, *b* describes density-dependence and ν is a normally-distributed error term with a mean of zero and a standard deviation of σ_ν. Because steelhead smolts produced by the spawners *S* emigrate over several years at ages 2-4, using the coho survival for the emigration year of the dominant steelhead smolt age-class is only an approximation to the mean survival experienced by all steelhead from the given brood year. The majority of the steelhead smolts are age 2 for

Thompson steelhead and age 3 for Chilcotin steelhead. I used the small-sample version of the Akaike Information Criterion (AIC_c) to compare the performance of models with and without coho marine survival as an environmental covariate. The purposes of this analysis were to assess the influence of variations in marine survival as a factor in the declines of the Thompson and Chilcotin steelhead stock aggregates and to estimate the density-dependent component of the SRR.

The second method used Kalman filtering to fit a Ricker SRR with time-varying stock productivity values to the time series of spawner-recruit data. The method and its rationale follow Peterman et al. (2000, 2003): the observation equation is the natural-log-transformed Ricker model with time-varying stock productivity a_t and the system equation is a random walk for the stock productivity a_t with a normally-distributed error term ω with a mean of zero:

$$\ln(R\ /S) = \ln\ (a_t) + b \cdot S +\ \upsilon$$

$$\ln\ (a_t)\ =\ \ln\ (a_{t-1})\ +\ \ \omega$$

I used the R-package MARSS version 3.2 (Holmes et al. 2012) rather than the equations published by Peterman et al. (2003) to do the estimation. The two methods make slightly different assumptions about initial priors and apply different estimation algorithms. Results from the two methods differ very slightly, but both show identical trends in a_t values and estimate similar b values. Both apply a smoother to the posterior estimates. I did separate analyses for the Thompson and Chilcotin stock aggregates (i.e., I did not assume common process errors) because differences in the mean age at smolting will result in recruits from the same brood year experiencing different ocean conditions. I used stock aggregates rather than the individual populations in the analyses because the stocks are managed as aggregates, because the reconstruction of the recruitment time series relies on fishing mortality estimates that can only be obtained for the aggregates, because the in-season estimates of abundance which are compared to abundance targets to make decisions on fishery openings/closures refer to the aggregates, and because estimates of the abundances of the aggregates are likely to be more accurate than those of the individual stocks.

I used the estimates of the density-dependent term b from the Kalman filter analysis to determine S_{LRP} values as a function of current adult productivity a, similar to Fig. 3; this procedure is self-consistent because adult productivity can vary independently of the (constant) density-dependence term through variation in marine survival. I used the maximal value of S_{LRP} over the range of a values encompassed in the data as the best estimate of a LRP value for a stock aggregate. I assessed the uncertainty in the maximal S_{LRP} estimates by using simulation to generate the frequency distribution of maximal S_{LRP} values. I re-estimated the S_{LRP} values 2,500 times, drawing the annual estimates of spawner numbers and fishing mortality estimates (from which recruitment is estimated, see below) from normal distributions with a coefficient of variation (CV) of 0.3. This is a very simple approach to determining the variation in maximal S_{LRP} values, but it is reasonable given the nature of the data. Because the abundances of the stock aggregates are low (and those of the component populations even lower), I wanted to have a high likelihood that the S_{LRP} used for decision-making is at or above the true S_{LRP}. I propose to use the upper 80^{th} percentile of the distribution of maximal S_{LRP} values as the estimate of the LRP threshold for a stock aggregate.

I estimated the CCT as the S_{MSY} calculated under “average” conditions, i.e., from a log-transformed time-invariant Ricker SRR that pooled all data over periods of both low and high productivity (1972-2007 brood years for the Chilcotin aggregate and 1984 to 2008 for the Thompson aggregate). The underlying assumptions are that the available data series represents long-term average condition and that the current period of low adult productivity is not permanent. Data for the Thompson aggregate are lacking for the high-productivity period in the 1970s so estimates of “average” conditions may be biased low. Because the spawner and recruit data that are used in the analysis are estimated with error, I determined the frequency distribution of S_{MSY} values by randomly drawing 2,500 annual estimates of spawner numbers and fishing mortality from normal distributions with CVs of 0.3, re-estimating the SRR and re-computing the S_{MSY}. I propose to use the upper 80th percentile of the resulting distribution of S_{MSY} estimates as the CCT for a stock aggregate to ensure that the precautionary threshold has a high likelihood of being at or above the true value for these small populations.

Spawner and recruitment data

Steelhead spawner estimates are available for the major populations in the Thompson stock aggregate (Nicola, Deadman, Bonaparte) from 1984 onward. Spawner estimates are available for the Chilcotin stock from 1972 onward. Currently, spawners are enumerated by resistivity counters for the Bonaparte (2002 onward) and Deadman (2009 onward) populations; resistivity counts have been calibrated to video records and are adjusted for counter efficiency (about 94% for the Deadman and 100% for the Bonaparte fishway). Prior to the installation of the resistivity counters, spawner estimates for the Deadman and Bonaparte populations were obtained by operating counting fences in the lower rivers and applying an expansion factor that was derived from telemetry studies (e.g., for areas below the fences). Since 1999 spawners in the Nicola watershed have been estimated in late spring using a maximum-likelihood area-under-the-curve (AUC) method from periodic visual counts in survey areas, combined with both visual and radio-tags application at the Thompson River confluence to measure observer efficiency and stream residence within the survey areas. The method is described in detail by Bison (2006) and generally follows those proposed by Hilborn et al. (1999) and Korman et al. (2002). Prior to 1999, spawners in the Nicola watershed were estimated as the sum of visual counts of non-kelted fish from periodic (usually weekly) helicopter surveys and/or stream walks; counts were not adjusted for observer efficiency, which was thought to be high because of favourable flows and water clarity throughout the period. Spawner estimates for the Chilcotin stock are estimated from the peak count from aerial surveys of the upper Chilko spawning area in late May (Spence 1978). The precisions of the various estimates are poorly known. Only the AUC method provides a statistically-sound estimate of uncertainty in the escapement, but the likelihood function is asymmetric, with an indeterminate upper bound. The standard deviation (SD) estimated from the lower 95% bound (i.e., about 2 SD) is roughly 30% to 40% of the estimate (from Bison 2006, Table 4), although this can vary considerably among years.

Steelhead returns in a given year were estimated by adjusting the sum of (escapements plus recreational fishery mortality) by an estimate of the instantaneous mortality for the preceding interception fisheries (i.e., bycatch mortality in ocean and river-mouth salmon fisheries and First Nation set-net fisheries downstream of the recreational fishery area) for the year of return *t*:

$$R_t = (S_t + M_t) \cdot e^{F_t}$$

where S_t is the spawning escapement, M_t is the estimated recreational fishery kill, and F_t is the instantaneous mortality rate from the fisheries that intercept late-summer steelhead. Recruits from a given brood year were obtained by summing the proportions of subsequent annual returns that originated from that year. Returns were allocated to their brood year using either the smolt age distribution from scale samples collected in the sport fishery and/or counting fences or by using the long-term average age composition for years in which scale samples were not available. The proportions by age differ between Thompson and Chilcotin steelhead. Age data for Chilcotin fish (5 years) are much more limited than those for Thompson fish (31 years).

The steelhead catch in the recreational fishery was estimated from creel surveys or from annual Steelhead Harvest Analysis (SHA) questionnaire data calibrated to creel data. SHA data were adjusted for reporting biases by applying the correction factors estimated by DeGisi (1999). Steelhead mortalities were estimated as the adjusted reported harvest plus catch-and-release mortality which was estimated using a post-release mortality rate of 3.2% (Anonymous 1998). F_t was estimated from a simple stochastic "boxcar"-type model of the net fisheries that are known to catch steelhead in the approach waters. Bison (2007) provides a detailed description of the model structure and its parameterization. The model computes expected steelhead catches using the algorithm of Cave and Gazey (1994) for the Fraser River stock sockeye fisheries. It uses observed fishing patterns, fishery-specific harvest rate estimates for other salmon stocks in the same fisheries, steelhead migration patterns estimated from the Albion test-fishery catches, and (hypothesized) combinations of gear-specific mortality rates to estimate F_t from 1992 onward. For years prior to 1992, F_t was fixed at 0.40, which is the average estimated rate for the period immediately following (1993-97) and before fishing patterns were changed to conserve Interior coho. The CV of the F_t estimates is roughly 20% to 30% (computed from Bison 2007).

The spawner estimates, F_t estimates, and age data are given in the Appendices.

Results and Discussion

Much of the inter-annual variation in the number of adult recruits per spawner for the Thompson and Chilcotin late-summer steelhead stock aggregates was explained by the measured survival of co-migrating coho smolts (Table 1), which strongly suggests that regional marine conditions have been an important factor in decreases in steelhead abundance over the last two decades. Models that included coho survival as a covariate were strongly supported by the data: ΔAIC values were 17.01 for the Chilcotin data and 4.56 for the Thompson data while the increases in adjusted R^2 were 0.37 and 0.14 respectively (Table 1). These results accord well with those of other studies, which have found recent trends of declining adult stock productivity at regional spatial scales for numerous stocks of several salmon species (Mueter et al. 2005, Pyper et al. 2005). Unlike these other studies, our analysis specifically identifies the changes as common effects from marine survival variation, and allows us to infer that possible decreases in freshwater productivity have likely been a less important factor in the observed declines of Thompson-Chilcotin steelhead (Fig. 1). The coefficients of the survival terms in Table 1 suggest that steelhead smolts survive at rates between 1.4× and 1.9× those of marked coho smolts.

Table 1. Coefficients and statistics for the natural-log-transformed Ricker stock-recruit relation:

$$\ln(R/S) = \ln(s) + \ln(\alpha) + b\cdot S$$

that includes or excludes the observed survival of co-migrating coho smolts as a covariate for the Chilcotin and Thompson steelhead stock aggregates. R = adult recruits from a given brood year, S = spawner abundance in the same brood year, s = the smolt-to-adult survival of co-migrating coho smolts from the same smolt year, S = spawner abundance in the same brood year, α = smolt productivity per spawner, and b = the density-dependence effect on smolt producivity per spawner. Note that the comparisons used the same set of years for models with and without the coho survival covariate, although additional data were available for years without survival data. Note also the ln(α) term for the models without survival covariates is the adult productivity, ln(a), since a = s·α.

term	**Chilcotin aggregate**		**Thompson aggregate**	
ln(s)	0.6651	N/A	0.3645	N/A
ln(α)	2.9962	0.7655	2.4321	1.1608
b	-0.0009141	-0.0006861	-0.0006507	-0.0005855
adjusted R^2	0.610	0.244	0.472	0.332
AIC	51.45	68.46	41.93	46.49
P	< 0.0001	0.0050	0.0006	0.0024
N	27	27	23	23
σ_v	0.574	0.799	0.543	0.611

The estimates in Table 1 for the models that include coho survival allow the SRR for freshwater smolt production to be recovered (although "smolt" here refers to those steelhead that share common mortality factors with co-migrating coho that originate in the lower Fraser River, i.e., those steelhead smolts that have survived the in-river migration from their freshwater production areas to the lower Fraser River in the vicinity of Chilliwack). The freshwater "smolt" productivity of the Chilcotin steelhead aggregate during the last two decades is estimated as 20.0 smolts per spawner and the maximum smolt recruitment (to the lower Fraser River) as 9,494 fish. The 95% confidence limits on the ln(α) term give a range of smolt productivity between 7.2 and 55.5 smolts per spawner. The estimated smolt productivity of the Thompson aggregate is lower, at 11.4 smolts per spawner, with a range of 3.4 to 38.0 smolts per spawner. The estimated current maximum recruitment of Thompson-origin smolts to the lower Fraser River is about 7,456 fish. The measured in-river survival of acoustically-tagged steelhead smolts from the Coldwater and Deadman rivers (Thompson watershed) has varied between about 18% to 55% among years and populations (Welch et al. 2011) so actual smolt production may be several times higher than our estimates. Although the uncertainty in the estimates of the maximum smolt recruitment is very high, it is clear that the stock aggregates are currently much less productive than simple habitat models (Riley et al. 1998) would suggest. Decreases in the average size of adult spawners have been inferred from the Albion test-fishery data (R.G. Bison, unpublished) and may partly account for the apparent low productivity of the steelhead aggregates.

The estimates of smolt productivity and maximum smolt recruitment for the models that use coho survival as a covariate can be combined with the relationships shown in Fig. 3 to estimate the S_{LRP} spawner numbers for the two stock aggregates. The smolt production required to return the maximal S_{MSY} spawners is $0.438 \cdot \beta$, thus 4,158 smolts for the Chilcotin aggregate and 3,266 smolts for the Thompson aggregate. The point estimates of the spawner numbers that will produce these smolt outputs (i.e., the S_{LRP} spawner abundances) are 214 adults for the Chilcotin and 301 adults for the Thompson aggregates.

Kalman-filtered estimates of adult stock productivity confirm that the declining trends in adult returns (Fig. 1) are a consequence of continuous declines in stock productivity for both steelhead stock aggregates from the late 1980s onward (Fig. 5). Current values are very low, only slightly above replacement for the Chilcotin aggregate and about 1.7 recruits per spawner for the Thompson aggregate. Parameter estimates for the Kalman-filtered Ricker SRR (Table 2) result in point estimates of the maximal S_{LRP} spawner numbers of 226 for the Chilcotin aggregate and 285 for the Thompson aggregate. The 80th percentile S_{LRP} values from 2,500 replicate estimates (Fig. 6) are 256 spawners for the Chilcotin aggregate and 355 for the Thompson. The 90th percentile values are 274 and 380 fish, respectively.

Parameter estimates for the time-invariant Ricker SRR (Table 2) represent our current understanding of the long-term average productivity of the two stock aggregates. Because marine survival and adult stock productivity vary through time (Fig. 5), the "long-term average" conditions that provide our estimates of SRR parameters and the management benchmarks that are derived from these SRRs depend upon the time intervals for which we have data. The estimation intervals differ for the two stock aggregates, and it seems likely that the average adult productivity estimate for the Thompson stock aggregate will be biased low by the absence of data from the 1970s and early 1980s (Fig. 5). Over the (rather short) period of record, the average stock productivity of the Chilcotin aggregate is 2.32 adult recruits per spawner while that of the Thompson aggregate is 2.83 adult recruits per spawner. The average SRR results in point estimates of S_{LRP} spawner abundances of 270 fish for the Chilcotin and 329 for the Thompson aggregates, and MSY spawner abundances of 689 and 939 spawners respectively. The expected long-term average unfished equilibrium stock sizes are about 1,624 for the Chilcotin aggregate and 2,255 spawners for the Thompson aggregate. The 80th percentile S_{LRP} values from 2,500 replicate simulations (Fig. 7) are 296 and 431 spawners for the Chilcotin and Thompson aggregates respectively while the 80th percentile S_{MSY} values are 763 and 1,187 spawners. The 90th percentile S_{LRP} values are 323 and 495 fish respectively, while the S_{MSY} values are 798 and 1,253 spawners.

The choice of appropriate management reference points depends in part on the quality of the information available to estimate the thresholds and in part on the intent of the policy. The policies that are stated in the Fisheries Program Plan for the BC Ministry of Environment are to "conserve wild fish and their habitats" and to "optimize recreational opportunities based on the fishery resource" (Anonymous 2007, p. 17). The available information indicates that the Thompson and Chilcotin aggregates of late-summer run steelhead are, on average, moderately productive stocks with low equilibrium stock sizes. Under current conditions of ocean survival, however, stock productivities are very low, as are current spawner escapements. This suggests that current management policy should be weighted towards conservation concerns. Over the last three generations (15 years for the Thompson aggregate, 18 years for the Chilcotin) escapements have declined 65% for the Thompson aggregate and 84% for the Chilcotin

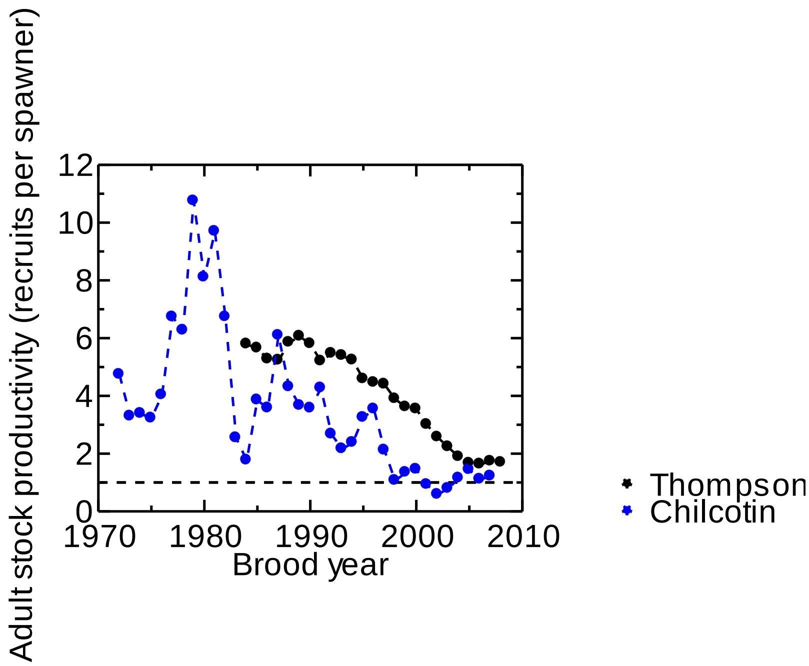

Fig. 5. Stock productivity in adult recruits per spawner has declined from the late 1980s onward for both the Chilcotin and Thompson steelhead stock aggregates. Current values are near replacement (a = 1, shown as a horizontal dashed line) for the Chilcotin aggregate and less than 2 for the Thompson stocks.

Table 2. Coefficients for Kalman-filtered and standard (time-invariant) Ricker stock-recruit relations:

$$\ln(R/S) = \ln(a) + b \cdot S$$

for the Chilcotin and Thompson steelhead stock aggregates. The Chilcotin data are for the 1972-2007 brood years while the Thompson data are for the 1984-2008 brood years. R = adult recruits from the spawners S, a = adult stock productivity per spawner, and b = the density-dependence effect on stock productivity per spawner. σ_ω and σ_v are the standard deviations of normally-distributed process and observation error respectively. S_{MSY} and S_{LRP} are estimates of the spawner numbers at maximum sustainable yield and at one-generation recovery to S_{MSY} respectively.

	Chilcotin aggregate		**Thompson aggregate**	
term	Kalman-filtered	time-invariant	Kalman-filtered	time-invariant
ln(a)	N/A	0.8433	N/A	1.0415
b	-0.0008662	-0.0006922	-0.0006887	-0.0005446
σ_ω	0.406	N/A	0.188	N/A
σ_v	0.079	0.749	0.338	0.610
adjusted R^2		0.238		0.286
N	36	36	25	25
S_{LRP}	226	270	285	329
S_{MSY}	N/A	689	N/A	939

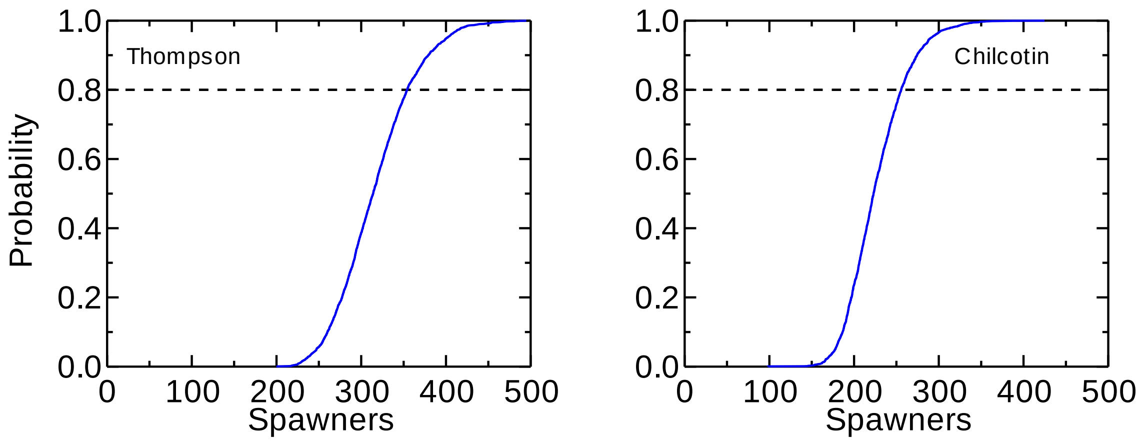

Fig. 6. The cumulative distribution of maximal LRP spawner abundances for the Thompson and Chilcotin steelhead stock aggregates from 2,500 replicate estimates of the parameters of a time-varying (Kalman-filtered) Ricker stock-recruit relation with spawner numbers and fishing mortality rates drawn from random normal distributions centered on the observed values with a coefficient of variation of 0.3. The dotted line marks the 80^{th} percentile of the distributions.

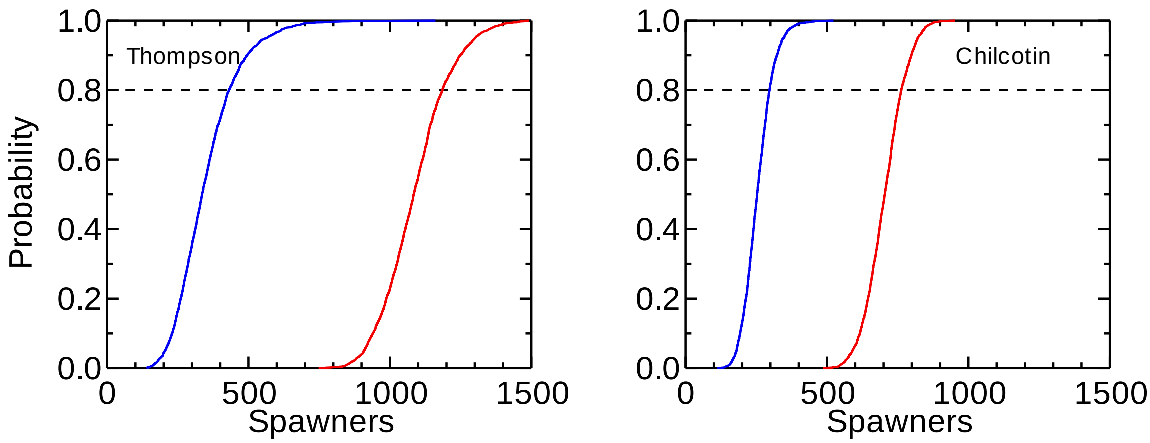

Fig. 7. The cumulative distribution of LRP (red) and MSY (blue) spawner abundances for the Thompson and Chilcotin steelhead stock aggregates from 2,500 replicate estimates of the parameters of a time-invariant Ricker stock-recruit relation with spawner numbers and fishing mortality rates drawn from random normal distributions centered on the observed values with a coefficient of variation of 0.3. The dotted line marks the 80^{th} percentile of the distributions.

aggregate. These aggregates have been identified as a “conservation unit” based on their phylogenetic relationships and life history characteristics (Parkinson et al. 2005) and may qualify as “designatable units” under Committee on the Status of Endangered Wildlife in Canada (COSEWIC) criteria. Under COSEWIC status assessment criteria (COSEWIC 2001), these stock aggregates could be classed as “threatened” and “endangered”, respectively.

Establishing clearly-defined and technically-defensible criteria for management actions such as reductions in or the elimination of human-induced mortality must be a management priority. Although the impacts of the recreational fishery on these aggregates is likely to be low, with expected catch-and-release mortality rates in the range from 1.6-3.6% (Anonymous 1998, Nelson et al. 2005), bycatch mortality from other fisheries could be much higher (Bison 2007, Baker and Schindler 2009). In the absence of such criteria and agreed-upon management actions by all users of the resource, both the fish and the fisheries that impact them may be at risk.

I recommend a conventional management framework consisting of a conservation concern threshold (CCT) at the estimated S_{MSY} escapement and a limit reference point (LRP) at the S_{LRP} escapement. The available data support several possible values for these management thresholds. Because of the uncertainties in the data and in the viability of very small individual populations, I propose that the upper 80^{th} percentiles of the estimated distribution of S_{MSY} and S_{LRP} values be used to ensure that there is a high likelihood that the true values of these thresholds are not exceeded before management acts to reduce or eliminate mortality. Because the implied “use” objective is to maintain the average productivity of the aggregates, I propose that the values derived from the time-invariant Ricker model be used. Thus the limit reference point would be 727 adult spawners for both aggregates combined and the conservation concern threshold would be 1,950 adult spawners. At in-season escapement estimates at or below 727 fish, all fisheries that impact these steelhead stocks would cease. At estimated escapements between 727 and 1,950 adults, measures to restrict mortality would be implemented progressively. At estimated abundances above 1,950 spawners, management objectives could be negotiated with users to achieve jointly-defined goals, which would likely include rebuilding the aggregates to higher abundances.

Several predictors of expected adult returns could be used for pre-season planning purposes. The Kalman-filtered estimates of time-varying stock productivities for the two aggregates can be combined with the average age-at-return estimates and the observed spawner numbers to project returns in future years (Fig. 8). Alternatively, the strong autocorrelation between returns in successive years could be used to provide an initial estimate of the expected run size (Table 3), although the predictive power may be lower. Such predictions could be used in conjunction with abundance-dependent harvest control rules to adjust the management of the various fisheries to expected returns; in-season estimates of abundance (e.g., from the Albion test-fishery) could then refine the pre-season estimates to make in-season adjustments.

A primary goal of the current management framework is to ensure that there is little risk that these stock aggregates or the individual populations within them would be extirpated. Although I have identified potential thresholds for management actions, I have not evaluated their ability to achieve our conservation goal, nor have I assessed the use benefits that might be obtained. However, previous simulations of steelhead population dynamics suggest that stocks with characteristics similar to the Thompson and Chilcotin aggregates would generally persist under low levels of exploitation (Johnston et al. 2000). In

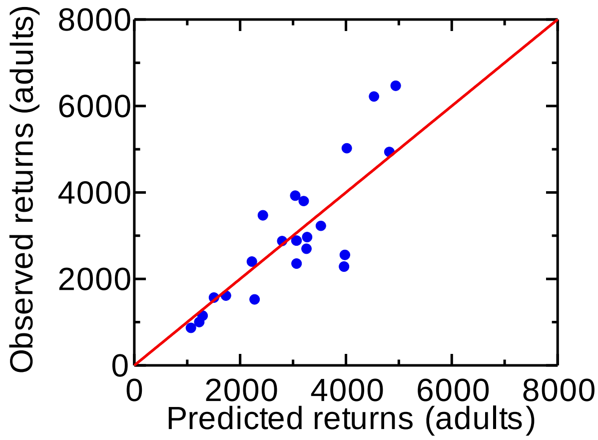

Fig. 8. Observed returns of adult spawners to the combined Thompson and Chilcotin steelhead stock aggregates are predictable from the time series of time-varying adult stock productivity values from Kalman filtered stock-recruit relationships and the average proportions-returning-at-age data. The median absolute error of the predicted returns is 21%. The one-to-one line is shown in red.

Table 3. Prediction of the adult steelhead returns in the next year from that of the current year:

$$\ln(\text{Returns}_{t+1}) = \text{constant} + \ln(\text{Returns}_t)$$

where Returns_t = pre-fishery adult numbers in the return brood year t.

term	Chilcotin	Thompson	Both pooled
constant	1.7455	3.2456	2.3025
$\ln(\text{Returns}_{t-1})$	0.7427	0.5713	0.7093
adjusted R^2	0.516	0.292	0.469
P	< 0.0001	0.0017	< 0.0001
N	35	28	28
$\sigma_{estimate}$	0.561	0.479	0.457

principle, alternatives similar to that illustrated in Fig. 9 can be compared by modelling the expected dynamics of the fishery management system, including the management control rules. While such simulations embody many tenuous assumptions, they do allow comparisons between alternative management regimes and should be done for these aggregates. The management thresholds that are proposed here may change as additional data on the dynamics of the stocks accumulate or as a result of evaluations of the expected performance of alternative management systems.

The application of the proposed thresholds to management decisions requires estimates of fish abundance that, in general, have low precision and which may result in "incorrect" decisions in a certain proportion of cases. For example, most maximum likelihood (ML) estimators of fish abundance approximate the 50th

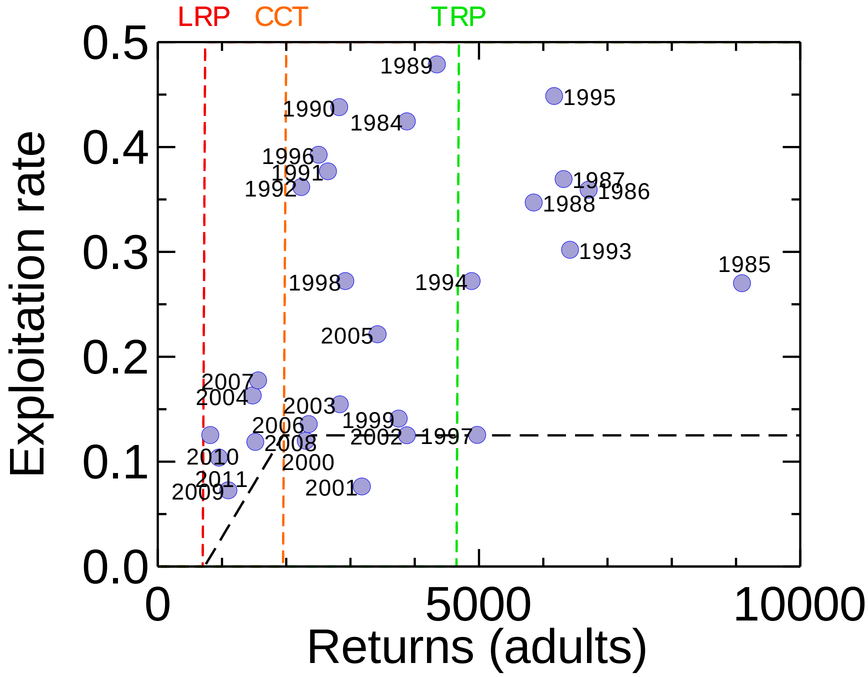

Fig. 9. An example of a possible harvest control rule (dashed black line) for fisheries that impact the Thompson-Chilcotin steelhead stock aggregates, incorporating a limit reference point (LRP) at an expected return of 727 adults, a conservation concern threshold (CCT) at 1,950 adults, and a target reference point (TRP) at 4,682 adults. Within the routine management zone, a maximum combined exploitation rate of 0.125 would be permitted for all fisheries that kill steelhead; this value is the median estimated exploitation rate of sport and net fisheries over the period since 2000. Below an expected return of 1,950 adults the fisheries would be adjusted to reduce the maximum exploitation rate, with a zero permitted impact (complete fisheries closures) at an expected return below 727 adults. Long-term societal benefits from the stock aggregates could (in principle) be maximized at adult returns near the target reference point. Estimated exploitation rates for the period from 1984 to 2011 are shown in blue, labelled by the spawning year of returning adults.

percentile of the distribution of possible abundance values given the observed data. If the uncertainty in the ML estimate is large, the true number may be quite different than the ML estimated number simply by random chance. If the consequences of an incorrect decision are likely to be grave, then the operational use of the proposed thresholds may require that the risk of an incorrect decision be reduced. Because the dynamics of very small populations of steelhead are very uncertain (e.g., possible depensatory effects at low fish abundance may result in irreversible declines), the operational use of the proposed LRP as a criterion for closing or opening fisheries may require a high degree of certainty that the estimated number of fish exceeds the threshold. For example, fishery managers may decide as a matter of policy that there must be less than a 20 percent chance that the number of fish is less than the LRP. In general, this policy will require that the ML estimate of fish abundance that is used to open or close the fishery be greater than the nominal LRP (Fig. 10); the actual ML estimate of abundance that corresponds to a 20 percent likelihood of the true estimate being less than the LRP will depend on the precision of the estimate. The in-season estimates of steelhead abundance that are derived from catches of steelhead in the Albion test-fishery have large uncertainty (R.G. Bison, unpublished), so the operational criterion that is used to assure

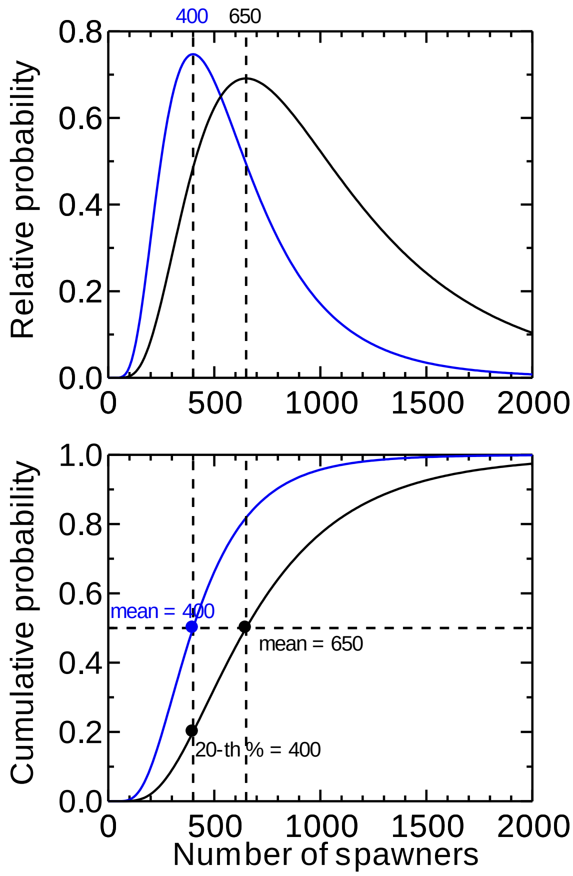

Fig. 10. Uncertainty in the estimates of steelhead abundance from observational data such as catches in the Albion test-fishery may require an operational decision criterion that is considerably greater than the nominal decision threshold in order to reduce the likelihood that fish numbers are below the threshold. The need for such a policy will depend on the risk that is associated with an incorrect decision, such as potential depensatory effects if steelhead number are allowed to decline to low abundance by not removing fishing mortality. In this example, the nominal threshold for management action is 400 fish. In the upper panel of the example, the distributions of the relative likelihoods of possible fish abundances are shown for maximum likelihood (ML) estimates of 400 and 650 fish. The lognormal distributions are assumed to have a constant coefficient of variation in this example. The lower panel shows the cumulative probability distributions, i.e., the likelihood that the estimate is less than a particular value. In this example, it is clear that a ML estimate of 400 fish has a 50 percent likelihood of the abundance being less than the threshold. In the example, the ML estimate of 650 has its 20^{th} percentile of the cumulative likelihood distribution at the nominal threshold of 400 fish. If the management policy is to assure that there is less than a 20 percent chance that abundance is below the nominal threshold of 400 fish, the operational criterion for taking action at the threshold would be a ML estimate ≥ 650 fish, based on the observed data. Other risk policies would result in different operational criteria for taking action.

that fish numbers are above the proposed LRP may be substantially higher than the nominal LRP value proposed here (see Fig. 10). The risk associated with different operational decision criteria can be assessed approximately by simulation of the management system.

Acknowledgements

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Appendix 1[a]. Spawner and adult recruit data for the Thompson steelhead stock aggregate, 1984-2008. F_t is the estimated harvest rate of net fisheries that catch steelhead.

					Proportion by age-at-return					
Brood year (t)	$Spawners_t$	$Angler\ kill_t$	F_t	$Returns_t$	P4	P5	P6	P7	P8	$Recruits_t$
1984	1115	1031		2146	0.154	0.769	0.077	0.000	0.000	3254
1985	3514	838		4352	0.027	0.946	0.027	0.000	0.000	2049
1986	2326	1349		3675	0.020	0.660	0.320	0.000	0.000	1943
1987	1675	1141		2816	0.000	0.895	0.105	0.000	0.000	1765
1988	1500	80	0.40	2357	0.061	0.898	0.020	0.020	0.000	4099
1989	1671	558	0.40	3326	0.000	0.840	0.160	0.000	0.000	4251
1990	1200	267	0.40	2189	0.019	0.837	0.144	0.000	0.000	3844
1991	1200	77	0.40	1905	0.000	0.886	0.114	0.000	0.000	1516
1992	900	36	0.40	1396	0.000	0.892	0.108	0.000	0.000	3323
1993	2955	36	0.34	4207	0.000	0.894	0.091	0.015	0.000	2165
1994	2660	128	0.28	3688	0.000	0.889	0.074	0.037	0.000	3234
1995	2591	129	0.55	4734	0.000	0.781	0.205	0.014	0.000	1326
1996	1019	133	0.41	1738	0.043	0.872	0.085	0.000	0.000	2138
1997	3000	69	0.12	3447	0.064	0.936	0.000	0.000	0.000	2448
1998	1470	114	0.26	2061	0.056	0.933	0.011	0.000	0.000	1736
1999	2500	97	0.12	2930	0.033	0.967	0.000	0.000	0.000	1303
2000	1310	54	0.10	1506	0.043	0.754	0.188	0.014	0.000	3173
2001	1700	37	0.06	1851	0.000	0.958	0.042	0.000	0.000	1521
2002	2300	75	0.11	2646	0.036	0.849	0.109	0.006	0.001	1139
2003	1500	34	0.15	1781	0.036	0.849	0.109	0.006	0.001	1289
2004	1000	14	0.16	1186	0.036	0.849	0.109	0.006	0.001	824
2005	2300	26	0.24	2952	0.000	0.922	0.078	0.000	0.000	512
2006	1500	34	0.13	1744	0.000	0.765	0.235	0.000	0.000	668
2007	930	19	0.18	1134	0.024	0.833	0.143	0.000	0.000	1175
2008	1200	19	0.11	1362	0.038	0.811	0.132	0.019	0.000	795
2009	690	0	0.07	743	0.019	0.750	0.212	0.019	0.000	
2010	590	20	0.11	678	0.063	0.625	0.313	0.000	0.000	
2011	500	0	0.11	556						
2012	1000	28	0.20	1256						

[a] data provided by R.G. Bison, BC Fish and Wildlife Branch, Kamloops, BC

Appendix 2[a]. Spawner and adult recruit data for the Chilcotin steelhead stock aggregate, 1972-2007. F_t is the estimated harvest rate of net fisheries that catch steelhead.

					Proportion by age-at-return					
Brood year (t)	$Spawners_t$	$Angler\ kill_t$	F_t	$Returns_t$	P4	P5	P6	P7	P8	$Recruits_t$
1972	960	202								2012
1973	1435	159								1328
1974	677	533								1275
1975	581	278								1112
1976	1022	179								1666
1977	494	487	0.40	1464						2232
1978	1152	365	0.40	2263						2566
1979	715	142	0.40	1278						4313
1980	893	21	0.40	1364	0.000	0.063	0.688	0.313	0.000	3247
1981	586	49	0.40	946	0.000	0.000	0.800	0.200	0.000	3509
1982	936	20	0.40	1426	0.000	0.000	0.545	0.364	0.091	2884
1983	1531	23	0.40	2318						996
1984	1133	41	0.40	1751	0.000	0.000	0.714	0.286	0.000	703
1985	3149	43	0.40	4762						822
1986	1992	53	0.40	3051						1220
1987	2328	31	0.40	3520						1964
1988	2342	14	0.40	3516						1304
1989	610	87	0.40	1039						1303
1990	403	37	0.40	657						1002
1991	466	46	0.40	763						1370
1992	542	33	0.40	859						890
1993	1546	40	0.34	2230						856
1994	917	5	0.28	1219						970
1995	830	6	0.55	1456						1325
1996	518	4	0.41	787						1206
1997	1373	4	0.12	1546						887
1998	672	2	0.26	877						379

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Appendix 2. (continued)

Brood year (t)	$Spawners_t$	Angler $kill_t$	F_t	$Returns_t$	Proportion by age-at-return P4	P5	P6	P7	P8	$Recruits_t$
1999	744	2	0.12	841						527
2000	739	2	0.10	819						580
2001	1258	6	0.06	1347						388
2002	1114	8	0.11	1251						234
2003	917	8	0.15	1074						324
2004	254	11	0.16	310						235
2005	384	2	0.24	490						406
2006	552	0	0.13	627						369
2007	374	2	0.18	449						331
2008	158	0	0.11	177						
2009	350	0	0.07	377						
2010	144	0	0.11	160						
2011	374	0	0.11	418	0.000	0.028	0.873	0.099	0.000	
2012	307	0	0.20	375						

[a] data provided by R.G. Bison, BC Fish and Wildlife Branch, Kamloops, BC

Appendix 3. Smolt-to-adult survival rates[a] for coho index stocks from the Lower Fraser River and for the Keogh River steelhead stock.

	Lower Fraser River coho stock				steelhead
Smolt year[b]	Chilliwack River	Inch Creek	Salmon River	Mean	Keogh River
1982	0.120			0.120	0.261
1983	0.144			0.144	0.155
1984	0.188			0.188	0.183
1985	0.131	0.067		0.099	0.253
1986	0.174	0.089	0.124	0.129	0.100
1987	0.181	0.203	0.229	0.204	0.133
1988	0.126	0.109	0.136	0.124	0.067
1989	0.106	0.080	0.136	0.107	0.154
1990	0.090	0.071	0.081	0.081	0.063
1991	0.057	0.097	0.098	0.084	0.036
1992	0.059	0.083	0.088	0.077	0.030
1993	0.064	0.060	0.100	0.075	0.033
1994	0.037	0.055	0.071	0.054	0.026
1995	0.040	0.039	0.082	0.054	0.040
1996	0.025	0.011	0.045	0.027	0.024
1997	0.013	0.005	0.028	0.015	0.081
1998	0.013	0.019	0.028	0.020	0.146
1999	0.034	0.011	0.062	0.036	0.045
2000	0.047	0.058	0.073	0.059	0.079
2001	0.032	0.018	0.071	0.040	0.032
2002	0.025	0.010	0.036	0.024	0.018
2003					0.026
2004		0.015		0.015	0.076
2005		0.008	0.014	0.011	0.023

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Appendix 3. continued[a]

	Lower Fraser River coho stock				steelhead
Smolt year[b]	Chilliwack River	Inch Creek	Salmon River	Mean	Keogh River
2006		0.013		0.013	0.031
2007		0.074	0.012	0.043	0.031
2008		0.018		0.018	0.048
2009		0.025		0.025	0.047

[a] Coho survival data are from Simpson et al. (2004) and DFO (2006, 2009, 2010, 2012)

[b] Coho survivals are reported by brood year or return year but have been adjusted to the smolt release year.