Implications of Demographic Uncertainty for Harvest Management of North American Sea Ducks

Guidance for Prioritizing Sea Duck Joint Venture Research and Monitoring Investments

Credit: M. Hoffman

Mark Koneff
Chris Dwyer
Guthrie Zimmerman
Kathy Fleming
Paul Padding
Patrick Devers

March 2016

EXECUTIVE SUMMARY

In 2010, the Sea Duck Joint Venture (SDJV) identified the need for improved science support for harvest and habitat management of North American sea ducks. In order to prioritize monitoring and research needs in support of harvest management, we applied a Prescribed Take Level (PTL) framework to assess the influence of uncertainty about sea duck demographic parameters on comparisons of observed and allowable harvest estimates. We focused on 7 populations of North American sea ducks: the American subspecies of common eider (*Somateria mollissima dresseri*), the continental populations of long-tailed duck (*Clangula hyemalis*) and white-winged scoter (*Melanitta fusca*), and eastern and western populations of black (*M. americana*) and surf scoter (*M. perspicillata*).

Prescribed Take Level (PTL) is an estimate of the allowable harvest of a population. Formulated as total harvest, calculation of PTL requires estimates of population size (N_t) and maximum growth rate ($r_{\max}$), while formulation of PTL as a harvest rate requires only an estimate of $r_{\max}$. We used a total harvest formulation of PTL for all populations, except common eider where banding data were sufficient to formulate PTL based on harvest rate. We defined $r_{\max}$ as the maximum growth rate achievable by a population in the absence of harvest under average environmental conditions. We derived $r_{\max}$ from the maximum finite growth rate ($\lambda_{\max}$) using an age-structured population projection matrix. In implementing the PTL framework we: (1) combined information from empirical studies and the opinions of experts to create probability distributions reflecting uncertainty in the individual demographic parameters needed to conduct the PTL; (2) used simulation to propagate that uncertainty into probability distributions of allowable harvest for each species; (3) compared estimates of allowable harvest to observed harvest; and (4) evaluated the sensitivity of the comparison of allowable to observed harvest estimates to uncertainty in the parameters used to derive those estimates.

We relied on a combination of published and unpublished data and estimates as well as the results of a formal expert elicitation to specify probability distributions for the parameters used in this assessment: age-specific survival, fecundity (calculated from reproductive rates such as nest success, clutch survival, and breeding propensity, as well as harvest age ratios), fall population size, observed harvest (sport and subsistence), and for common eiders, observed harvest rate. The probability distributions reflected uncertainty about the true mean value of each demographic parameter for each population. We used Monte Carlo simulations to estimate $r_{\max}$, allowable harvest, and observed harvest for each population. We then used linear regression to assess the sensitivity of the difference between allowable and observed harvest estimates to uncertainty in the component parameters of $r_{\max}$, fall population size, and observed harvest. We identified populations at risk of overharvest by the proportion of simulations where observed harvest exceeded allowable, and categorized demographic information needs into three levels of priority based on their uncertainty and their influence on the comparison of allowable and observed harvest.

Our literature search revealed a dearth of empirical data for most of the populations, and our effort to augment the empirical data by eliciting opinions from subject-matter experts met with limited success. Accurate quantification of uncertainty was a crucial component of the assessment, and our results and conclusions below are conditional on adequate descriptions of

uncertainty for each parameter. In general, our allowable harvest (or harvest rate) estimates were very uncertain, much more so than the estimates of observed harvest.

American Common Eider. The median allowable harvest rate for American common eiders was –0.0009 (95% credible interval -0.0812; 0.0692). The percent of simulations where observed harvest rate was less than allowable harvest rate was 20%. The comparison of observed and allowable harvest rates was most influenced by uncertainty in adult survival, as well as several components of fecundity including duckling survival, the ratio of juvenile to adult female wings in samples submitted by hunters (i.e., harvest age ratio), hatching success, and clutch size. Highest priorities for research and monitoring were estimates of age ratios and duckling survival.

Eastern/Western Black Scoter. For eastern black scoters allowable harvest was 29,940 (807; 93,753), and the percent of simulations where observed harvest was less than allowable harvest was 52%. For western black scoter allowable harvest was 10,854 (-11,058; 37,219), and observed harvest was less than allowable harvest in 30% of the simulations. Adult survival was highly influential for both populations but due to its low uncertainty was only a medium priority for research and monitoring. For eastern black scoters, the highest priority information needs were population size and duckling survival, while moderate priority needs, in addition to adult survival, included age ratio, and the proportion of hens first breeding at age 2. For western black scoters, 3 fecundity parameters were the highest priorities for research or monitoring: nest success, duckling survival, and harvest age ratio. Observed harvest was also categorized as a high priority information need though it was less influential on comparisons between allowable and observed harvest than the fecundity parameters.

Eastern/Western Surf Scoter. For eastern surf scoters, the median allowable harvest of 23,149 (-9,308; 78,894) was less than the median observed harvest by approximately 15,000 birds, The percent of simulations in which observed harvest was less than allowable harvest was 25%. High priority information needs based on the sensitivity analysis were harvest age ratios, nest success, and population size. Adult survival and differential vulnerability were classified as moderate information needs. For western surf scoters, the median allowable harvest was 14,354 (-61,985; 82,110). Observed harvest was less than allowable harvest in 59% of the simulations. Adult survival was most influential on comparisons of observed and allowable harvest and was the highest priority information need. Other high priority information needs included population size, clutch size, juvenile survival, and differential vulnerability.

White-winged Scoter. Median allowable harvest was 13,054 (-68,824; 61,072). The percent of simulations in which observed harvest was less than allowable harvest was 36%. Observed harvest was a high priority information need, although its influence on the harvest comparison (based on absolute slope) was less than the 4 parameters that were ranked as moderate priority information needs (differential vulnerability, nest success, hatching success, and adult survival) as a result of a larger relative uncertainty surrounding observed harvest.

Long-tailed Duck. Median allowable harvest for long-tailed ducks was -48,966 (-202,663; 60,561). The percent of simulations in which observed harvest, 43,044 (32,151; 57,589), was less than allowable harvest was only 5%. Reproductive rate estimates for long-tailed ducks from the literature were very low compared to all populations other than common eiders. Population size was the only high-priority information need identified according to our criteria. Four parameters were categorized as moderate priority information needs: adult survival, nest success, proportion of first time breeders breeding at age 2, and survival of second-year birds.

In general, this assessment highlights the high degree of uncertainty associated with simulated values of allowable harvest for all populations. We have particularly low confidence in the assessment for long-tailed ducks, and the assessment for American common eider may apply only to the segment of this population breeding in Maine and the Maritimes. Comparisons of our simulated median values of intrinsic growth rates were lower than theoretical maximum values indicating that these populations were experiencing sub-optimal environmental conditions, input parameter values were not consistent with growth unconstrained by density or harvest, or input parameter values were representative of only a subpopulation with lower growth potential than the entire population.

Conclusions from this assessment include: (1) reductions in uncertainty in the high and moderate priority parameters could most significantly improve harvest inferences and decision making; (2) uncertainty about overall fecundity had more influence on comparisons of allowable and observed harvest than adult survival or observed harvest, however, individual components of fecundity can be difficult to study at a population scale; (3) adult survival, though characterized by less uncertainty than individual components of fecundity, is a high priority information need given the sensitivity of growth rate and allowable take to this parameter, and (4) uncertainty about population size was a high priority information need for four of the six populations where it factored into the assessment. We recommend that the SDJV (1) prioritize research and monitoring efforts on the long-tailed duck and American common eider; (2) prioritize research and monitoring on high priority parameters identified for each population; (3) continue efforts to integrate the operating procedures and analysis of presently disparate breeding population surveys for sea ducks; and (4) conduct PTL assessments periodically, incorporating new information in order to revise priority information needs.

Implications of Demographic Uncertainty for Harvest Management of North American Sea Ducks

Guidance for Prioritizing Sea Duck Joint Venture Research and Monitoring Investments

Mark Koneff, U.S. Fish and Wildlife Service, Division of Migratory Bird Management, Orono, ME
Chris Dwyer, U.S. Fish and Wildlife Service, Division of Migratory Birds, Northeast Region, Hadley, MA
Guthrie Zimmerman, U.S. Fish and Wildlife Service, Division of Migratory Bird Management, Sacramento, CA
Kathy Fleming, U.S. Fish and Wildlife Service, Division of Migratory Bird Management, Laurel, MD
Paul Padding, U.S. Fish and Wildlife Service, Division of Migratory Bird Management, Laurel, MD
Patrick Devers, U.S. Fish and Wildlife Service, Division of Migratory Bird Management, Laurel, MD

INTRODUCTION

The population dynamics of sea ducks (Tribe *Mergini*) are poorly understood relative to other North American waterfowl (Caithamer et al. 2000, Goudie et al. 1994, Kehoe et al. 1994, Sea Duck Joint Venture Continental Technical Team 2003, Sea Duck Joint Venture Management Board 2008, Boyd et al. 2015). Sea duck life histories are characterized by high adult survival, delayed maturation, and low reproductive capacity suggesting that population abundance of these species may be sensitive to factors influencing adult survival (e.g., harvest). Increased interest in sport harvest of sea ducks in some areas in recent decades may be related to regulatory restrictions on other popular harvested species such as Canada goose (*Branta canadensis*) and American black duck (*Anas rubripes*). Special hunting regulations, established in the 1930s to increase hunting opportunity on sea ducks, reflect past perceptions that these species were lightly harvested and could sustain additional harvest pressure. However, limited population monitoring data for North American sea ducks suggest that 10 of 15 species were declining in the 1980s and 1990s (Goudie et al. 1994, Kehoe et al. 1994, Caithamer et al. 2000, Sea Duck Joint Venture Continental Technical Team 2003, Sea Duck Joint Venture Management Board 2008). Although the extent and causes of declines are largely unknown, this has increased concern over sea duck harvest and the limitations of available population data. Recent analyses indicate that 11 of 22 North American sea duck populations are now stable or increasing, but these do not include the common eider (*Somateria mollissima*) and scoter (*Melanitta* spp.) populations for which special hunting regulations exist in eastern North America (Bowman et al. 2015).

In 2010, the Sea Duck Joint Venture (SDJV) identified the need for improved science support for harvest and habitat management of these species. As a first step toward addressing the needs of harvest management decision-makers, the SDJV established a harvest management subcommittee to engage the harvest management community, determine priority information needed to support decisions, assess the ability of SDJV research and monitoring investments to address those needs, and incorporate priorities into the next SDJV Strategic Plan Revision.

Working with the SDJV harvest management subcommittee, we developed a Prescribed Take Level (PTL; Runge et al. 2009) assessment framework to evaluate how much influence uncertainty in various demographic parameters has on inferences about contemporary harvest levels. The PTL framework estimates allowable take level, in this context allowable harvest level, which is the number of birds (or proportion of total birds) that can be harvested from a population, given a stated harvest management objective. Allowable harvest estimates derived from the PTL framework can be compared to observed levels of harvest to assess the appropriateness of contemporary harvest levels and regulations. By incorporating uncertainty about underlying demographic parameters into the PTL assessment as probability distributions, simulation can be used to propagate that uncertainty into probability distributions reflecting uncertainty in both allowable and observed harvest levels. Comparing the derived probability distributions for allowable versus observed harvest allows harvest managers to assess the risks associated with current harvest policies, while still taking into account the uncertainty associated with estimates of allowable and observed harvest. Sensitivity of the risk associated with contemporary harvest levels to uncertainty in individual demographic parameters can then be assessed to provide guidance on population monitoring and research priorities.

In implementing this framework we: (1) combine information from the literature and expert elicitation to create probability distributions reflecting uncertainty in the individual demographic parameters needed to conduct the PTL; (2) use simulation to propagate that uncertainty into probability distributions of allowable harvest for each species; (3) compare estimates of allowable harvest to observed harvest; (4) evaluate the sensitivity of the comparison of allowable to observed harvest estimates to uncertainty in the parameters used to derive those estimates; and (5) provide recommendations on monitoring and research priorities to the SDJV in support of harvest management.

METHODS

This assessment focused on five species of North American sea ducks: common eider, long-tailed duck (*Clangula hyemalis*), white-winged scoter (*Melanitta fusca*), surf scoter (*M. perspicillata*), and black scoter (*M. americana*). These species were selected by the SDJV for initial focus because of concern over their population status and their importance as game species in some regions. Some of these species consist of two or more populations or subspecies and demographic rates may vary greatly among them. The specific species, subspecies, and populations assessed include the American subspecies of common eider (*S. m. dresseri*), the continental populations of long-tailed duck and white-winged scoter, and eastern and western populations of black and surf scoters. Although the SDJV and the North American Waterfowl Management Plan (NAWMP 2012) recognize four North American subspecies of common eider, we focused on the American subspecies because it is shared between Canada and the U.S. and its status is of growing conservation concern.

Prescribed Take Level Framework

The PTL framework is based on the theory of density-regulated population growth (Runge et al. 2004, Runge et al. 2009, Johnson et al. 2012). PTL is a generalization of the Potential Biological Removal (PBR; Wade 1998) framework and is applicable to a broader class of take applications, including hunting. When annual estimates of population size are available, PTL can be applied to annual harvest management decisions as:

$$PTL_t = F_0 \times \left(\frac{r_{\max}}{2}\right) \times N_t \tag{1}$$

where PTL_t is the prescribed or allowable harvest level (in numbers of individuals) for year *t*, and F_o is a scaling factor representing the management objective, i.e., the desired take level relative to maximum sustained yield (Runge et al. 2009, Johnson et al. 2012). The term $r_{\max}/2$ represents the maximum allowable harvest rate. The functional form of density dependence can influence allowable take (Johnson et al. 2012, Williams 2013). The harvest management subcommittee explored options for assessing the influence of nonlinear density dependence on allowable take level. However, a lack of published information and limited monitoring data for these populations led us to assume linear density dependence throughout this assessment. In general, given the life history strategies of sea ducks, these species would be expected to exhibit stronger density dependent regulation at population sizes nearer to carrying capacity. The assumption of linear density dependence in the PTL formulation can be expected to result in more conservative assessments of allowable take, from a conservation standpoint. Though we elected to assume linear density dependence (i.e., $\theta = 1$ in the theta-logistic model), future assessments could incorporate uncertainty in the functional form of density dependence as a probability distribution on θ similar to other demographic parameters.

If harvest rate can be directly measured for a population, PTL can be expressed as an allowable harvest rate by:

$$PTL = F_0 \times \left(\frac{r_{\max}}{2}\right) \tag{2}$$

A strategy based on harvest rate requires no associated measure of population size to ensure sustainability, while one based on total harvest must include a monitoring program capable of tracking changes in population size to appropriately scale harvest to population size. In either formulation, a harvest strategy with an objective of MSY would set $F_o = 1$, whereas F_o values <1 reflect a more conservative strategy (from a conservation standpoint) with an associated equilibrium population size greater than that under an MSY strategy. No explicit harvest management objectives have been established for sea ducks other than the objective of sustainability implicit in the provisions of the Migratory Bird Treaty Act. We therefore assumed that the harvest objective for sea ducks was MSY and set $F_o = 1$ for all species assessed.

Most of the best available population estimates of the sea duck species of interest are from winter surveys (western black scoter is the exception, see below under *Parameters*). To use winter population estimates in the PTL framework, we had to either: 1) convert the estimates to breeding population (pre-birth pulse) estimates, or 2) convert them to fall population (i.e., post-birth pulse) estimates and modify the PTL framework to accommodate a post-birth pulse estimate. We chose to modify the PTL framework to accommodate fall population estimates because experts believed that there were fewer assumptions converting winter population estimates to a fall flight than a spring breeding population. F. A. Johnson (U.S. Geological Survey, Southeast Ecological Science Center, unpublished data) derived the maximum sustainable harvest rate for a fall population as:

$$h_{\mathrm{MSY}}^{\mathrm{FF}} = \frac{r_{\max} \times (1 - hFF_{\mathrm{MSY}})}{2} \tag{3}$$

Therefore, assuming linear density dependence and an objective of maximum sustained yield (i.e, $F_o = 1$), we calculated allowable harvest rate for the American common eider, for which comparative estimates of observed harvest rates are available, using equation (3), and for all other populations using:

$$H_{t\ \mathrm{MSY}}^{\mathrm{FF}} = \left(\frac{r_{\max} \times (1 - hFF_{\mathrm{MSY}})}{2}\right) \times N_t^{\mathrm{FF}} \tag{4}$$

where N_t^{FF} is a fall population (i.e., fall flight) estimate for year t. While the harvest rate-based PTL for eiders is specific to the female segment of the population, the total harvest-based PTL estimates for the other species of interest include both sexes since we lack information on the sex composition of the fall population.

Estimating $r_{\max}$

We defined $r_{\max}$ as the maximum growth rate achievable by a population when that population is not exposed to the source of mortality of interest (in this case harvest), is not under any resource limitations causing density-dependent regulation, and is experiencing otherwise average environmental conditions. This definition implies that $r_{\max}$ is not a species- or population-specific constant and is determined by a species' life history traits as expressed in a particular environmental setting (Runge et al. 2004). This further implies that $r_{\max}$ for a population can change over time based on change in the mean environmental conditions experienced by the population.

A variety of methods exist to estimate $r_{\max}$ (Runge et al. 2004) and each has inherent assumptions and implications with respect to the evaluation of harvest sustainability. Because sea ducks exhibit varying degrees of age structure with respect to reproductive rates (Bordage and Savard 1995, Brown and Fredrickson 1997, Gilliland et al. 2009, Goudie et al. 2000, Robertson and Savard 2002, Savard et al. 1998) we used the age-structured population projection matrix **A** (shown below for 3 age classes; Caswell 2001) to estimate the maximum finite population growth rate in the absence of harvest ($\lambda_{\max}$):

$$\mathbf{A} = \begin{bmatrix} b_1 \times p_1 & b_2 \times p_2 & b_3 \times p_3 \\ p_1 & 0 & 0 \\ 0 & p_2 & p_{\mathrm{A}} \end{bmatrix} \tag{5}$$

We used a post-birth pulse formulation of the projection matrix with four age classes for eider and three for the other species. We calculated $\lambda_{\max}$ as the dominant eigenvalue from the projection matrix using the *popbio* package in program R (Stubben and Milligan 2007, R Core Team 2014) and estimated $r_{\max}$ as $\lambda_{\max} - 1$ for a discrete growth process.

While the projection matrix formulation requires the estimation of a larger number of input parameters than some other methods of estimating $r_{\max}$, such as the demographic invariant method (DIM) described by Niel and Lebreton (2005), it has the advantage of greater generality and flexibility from a management perspective. An example is the ability to incorporate both fecundity and survival processes affecting $r_{\max}$ into the projection matrix, both of which are related to differences in the environmental setting within which specific management populations occur. While context-, or population-specific $r_{\max}$ estimates are of greatest utility in harvest management, the challenges in estimating demographic rates under conditions applicable to $r_{\max}$, led us to also use the DIM to estimate $r_{\max}$. The DIM requires only estimates of adult survival and age-at-first-breeding, and is based on fundamental relationships between survival, fecundity, and generation time that apply broadly within taxonomies. We consider DIM estimates of $r_{\max}$ as theoretical maximum values that can be contrasted to the population-specific estimates derived from the matrix models. We contrast the DIM and projection matrix estimates of $r_{\max}$ by examining overlap in the computed credible intervals. To apply the DIM, we computed mean body mass of adult hens from the literature (Nelson and Martin 1953, Korschgen 1977, Bellrose 1980, Vermeer and Bourne 1984, Bordage and Savard 1995, Leafloor et al. 1996, Savard et al 1998, Robertson and Savard 2002, Kellert et al. 2005) and used an allometric relationship between body mass and adult survival (Johnson et al. 2012) to compute adult survival in the absence of harvest.

Comparison of Allowable Harvest to Observed Harvest

We compared estimates of allowable harvest, as total harvest for scoters and long-tailed ducks and as harvest rate for eiders, to observed total harvest for scoters and long-tailed ducks and observed harvest rate for eiders. Using Monte Carlo simulation with 10,000 iterations, we sampled from the probability distributions described for the demographic parameters and computed an allowable harvest estimate for each iteration (Runge et al. 2009, Johnson et al. 2012). During each iteration, we also sampled from the probability distribution for total observed harvest (for scoters and long-tailed ducks) and observed harvest rate (for eiders). The final step in each iteration was to compare the simulated allowable and observed harvest. We then computed the proportion of iterations where observed harvest (or harvest rate) was less than allowable harvest (or harvest rate). We interpreted those proportions as measures of the relative risk of harvesting in excess of maximum sustained yield accounting for uncertainty in both allowable and observed harvest. Proportions closer to zero reflected a higher relative risk of overharvest, given a management objective of maximum sustained yield (i.e., $F_o = 1$).

Developing Probability Distributions for Demographic Parameters

We relied on a combination of published and unpublished data and estimates as well as the results of a formal expert elicitation to specify probability distributions for the parameters used in this assessment. We sought to develop probability distributions that reflected uncertainty about the true mean value of each demographic parameter for each population.

Process

Literature search.— Members of the harvest subcommittee searched both published and “gray” literature and compiled information on survival rates, fecundity components (age-specific breeding propensity, clutch size, nest success, hatching success, and duckling survival), overall fecundity, age of first and last breeding, population size, and harvest rates and/or total harvest for the seven populations under consideration. We sought estimates of the mean values of these parameters for each of the populations of interest under conditions corresponding to maximum growth potential. Considerable effort was expended in the search for representative estimates, and to describe associated probability distributions which represented uncertainty about the true population mean. In comparison to other waterfowl species, available information on demographic rates for sea ducks is limited in both spatial and temporal scope. Frequently, published demographic rates were unavailable, were available only from dated studies, or were applicable only to local study populations (Appendix A). Vital rates presented typically applied to populations subjected to harvest and possibly experiencing density-dependent regulation and so were not directly applicable in estimating $r_{\max}$. Moreover, we were concerned that estimates of statistical variability associated with published demographic rates for local populations were not representative of the uncertainty about the true population means for those parameters. Therefore, we also conducted an expert elicitation process to supplement available information on demographic rates and their uncertainty.

Elicitation Process.— Increasingly, researchers have employed expert elicitation as a method of synthesizing expert opinions about uncertain quantities or processes (e.g., McBride et al. 2012). We used expert elicitation to generate probability distributions for age-specific survival and fecundity parameters (i.e., breeding propensity, clutch size, hatch success, nest success, and duckling survival), fall population size, and differential vulnerability (DV) of age classes to harvest (used to adjust harvest age ratios as an alternative measure of fecundity, see below). We summarized the information for each species obtained through the literature search (see Tables A1-A7) and provided this to an expert panel of sea duck ecologists and managers in the U.S. and Canada as background information for the elicitation. We requested that each panel member identify any additional published or unpublished data sources not summarized in the literature review. We also provided each panel member with an elicitation form and requested that they provide four values for each parameter (Tables B1-B2) in accordance with the four-point elicitation method described by Speirs-Bridge et al. (2010). The values requested for each parameter included the experts' best appraisal of the true population mean, the highest the mean could be, the lowest the mean could be, and a probability to describe their confidence that the true mean lies within the bounds they specified. Panel members only provided values for parameters for which they felt they could offer an informed opinion. We reviewed responses from each expert and when we perceived ambiguity in a panel member's response, followed up with that expert for clarification. We then compiled input from all experts and returned the compilation to them for review, without revealing the names of the panel members. Experts were asked to view their peers' responses, identify areas of concern or misunderstanding, and revise their own values as they deemed appropriate.

Parameters

Survival (p1, p2, p.ad).— To obtain survival rates, we compiled literature values from mark–recapture studies or banding data. Literature values generally represented survival of adults of populations subjected to harvest. Therefore, we also compiled modeled survival estimates based on allometric relationships predicting maximum survival as a function of body weight (Johnson et al. 2012, F. A. Johnson, U.S. Geological Survey, Southeast Ecological Science Center, unpublished data). Both literature-derived estimates and those based on published allometric relationships were provided to the elicitation panel as aids in specifying values for mean juvenile, sub-adult, and adult survival rates in the absence of harvest. The number of sub-adult age classes differed for each population depending on published information and expert opinion regarding the breeding propensity of each age class.

Fecundity (b). — Fecundity was defined as the number of female offspring fledged per reproductive female per year. To estimate $r_{\max}$, values for reproductive rates should be representative of a population undergoing unrestrained growth, without density-dependent regulation, under average environmental conditions (Runge et al. 2004). Two alternative and independent measures of fecundity were considered in this assessment. In the first approach we computed fecundity from component rates where b = breeding propensity × clutch size × nest success × hatching success (i.e., egg survival) × the proportion of ducklings that are female (assumed 0.50) × duckling survival. In the second approach, we computed fecundity directly from female harvest age ratios adjusted for differences in the vulnerability of juveniles and adults to being harvested (i.e., differential vulnerability).

We incorporated age structure by allowing age-specificity in breeding propensity. Limited published accounts (e.g., Iles et al. 2013) have indicated the potential for periodic spikes in fecundity for some species of sea ducks. Therefore, we allowed panel members to specify: (1) baseline and high reproductive values, and (2) the frequency of the spikes. Two experts provided these estimates for eider duckling survival, which we incorporated by sampling from two probability distributions (i.e., one representing baseline and one representing high duckling survival, based on the frequency specified by the experts).

For the second method of computing overall fecundity, we relied on the age composition of harvested birds estimated from fall age ratios adjusted by the relative vulnerability of juvenile and adult females to harvest (Cowardin and Blohm 1992). This method can produce reliable estimates of fecundity provided that the spatial distribution of juvenile and adult birds is such that the two age cohorts are equally available to hunters, and there is reliable information on DV. Uncorrected harvest age ratios were obtained from hunter-submitted wings collected during annual harvest surveys in U.S. and Canada. There are, however, no reliable estimates of DV for the seven populations of interest. Therefore, we elicited DV values from the expert panel and provided them, as background, DV estimates for several other species for which adequate band recovery data are available including mallards (*A. platyrhynchos*) and black ducks. Estimates of fecundity based on age ratios are reflective of current conditions (i.e., population density, weather conditions) and therefore may not represent conditions of unrestrained growth.

Population size (N).— With the exception of a survey of breeding western black scoters that covered >80% of the population's breeding range and was conducted annually from 2004-2012 (Bowman et al. 2015), sea duck abundance monitoring programs are poor. The best available abundance data for eastern and western surf scoters, eastern black scoter, white-winged scoter, and long-tailed duck were obtained through recently conducted winter surveys of the Atlantic and Pacific Coasts and Great Lakes (Silverman et al. 2012; SDJV, unpublished data). The Pacific Coast winter surveys also estimated winter abundance of western black scoters, and the winter estimate (based on the sum of estimates from a number of partial surveys conducted over a period of years) was credible when compared with the breeding population survey estimate, considering that the winter estimate includes young-of-the-year birds and the breeding population survey does not. Therefore, we used winter survey data to estimate population size. We estimated fall flight by adding winter population estimates and total harvest estimates from that particular season. This required an assumption that mortality from non-hunting sources was negligible between the fall flight and the winter surveys. We provided these estimates to the expert elicitation panel as background information in formulating their responses.

Harvest and harvest rate (H.obs, h.obs). — Harvest estimates utilized in this assessment consisted of both fall and winter sport harvest and subsistence harvest. We compiled estimates of fall and winter harvest obtained from the national harvest surveys that are conducted annually in the U.S. and Canada, and subsistence harvest estimates from a variety of sources (Tables A2-A7) to quantify observed take for scoters and long-tailed ducks. We represented fall and winter harvest (H_{FW}) as the 2004-2013 mean of the combined estimated harvest in the U.S. (H_{US}) and Canada (H_{Canada}), and adjusted that mean for a presumed bias in harvest estimates (0.731; see Padding and Royle 2012, Rothe et al. 2015).

We used Alaska subsistence harvest estimates for 2011 (Rothe et al. 2015), as well as subsistence harvest estimates obtained from several one-time surveys of aboriginal peoples that were conducted in specific regions across Canada from the 1980s to the 2010s (Natcher et al. 2011; Rothe et al. 2015; Tobias and Kay 1994; C. Lepage, personal communication). We list the values of total harvest and their uncertainty in Table B2. Take associated with hunting includes crippling loss, which is assumed to be about 30% of retrieved harvest (Rothe et al. 2015). We adjusted the combined total harvest estimates (H_{FW}) for crippling loss (C) to arrive at the total observed harvest (*H.obs*) used in the assessment using the formula:

$$H.obs = \frac{H_{\mathrm{FW}}}{(1-C)} \tag{6}$$

An estimate of the total harvest (in numbers of birds) of American common eiders was not needed because they have been banded frequently along the Atlantic coast since 2002, which provides information for estimating harvest rates for this population. We used banding and dead recovery data obtained from the Bird Banding Laboratory and the Brownie et al. (1985) recovery model to estimate recovery rates for nesting and molting female birds. The dead recovery model assumes that no mortality occurs between banding and exposure to harvest. Molting birds are likely to meet this assumption, but we were concerned that the molting birds might underrepresent breeding females. Therefore, we included nesting birds in the recovery analysis. We acknowledge that some mortality occurred between nesting and the hunting season; however, given the high annual survival rate of female eiders (Krementz et al. 1996), we assume that any violation of this assumption would lead to only a slight underestimate of recovery rates. We converted recovery rates to overall rates of retrieved harvest plus unretrieved kill, *h.obs*, by dividing the recovery probabilities by a reporting probability estimated for mallards in the Atlantic Flyway (Boomer et al. 2013), and the same estimate of crippling loss used to calculate observed kill. We incorporated the uncertainty in recovery rates, reporting rates, and crippling loss into probability distributions describing uncertainty in overall rates of retrieved harvest and unretrieved kill.

Combining Sources of Information and Generating Probability Distributions

We used the results from the elicitation to construct expert-specific probability distributions (for those parameters addressed by each expert) using the methods described by Conroy and Peterson (2013:186). We also described a probability distribution for reproductive parameters in cases where the literature provided ample information (see Appendix, Table B1). We generated beta distributions for binomial parameters (e.g., survival, nest success, hatch success, etc.) and either a normal or log normal distribution for all other parameters (e.g., population size, clutch size, or harvest). We used the *qmedist* function of the *fitdistplus* package in R to derive distribution parameters and then functions *rnorm*, *rlnorm*, or *rbeta* to generate probability distributions for the simulation (Delignette-Muller and Dutang 2015). We used Monte Carlo simulation (10,000 iterations) to select randomly, and with equal probability, from the probability distributions derived from expert responses as well as available published or unpublished data, to develop a single overall probability distribution characterizing uncertainty for each parameter across all experts and data sources.

We generated two independent distributions to characterize uncertainty in overall adult fecundity (ratio of juvenile females to adult females) for each population. The first fecundity distribution, based on individual reproductive rates, was generated by sampling 10,000 values independently from each of the five reproductive parameter probability distributions, and multiplying their product by 0.5 (for the ratio of female to total ducklings). The second adult fecundity distribution, based on harvest age ratios, was generated by sampling 10,000 values from the probability distribution constructed for the 2004-2013 mean female harvest age ratio and estimated variance, and then dividing these by a similar sample from the probability distribution of differential vulnerability generated from the values provided during the expert elicitation. Note that harvest age ratios were estimated as hatch-year:after-hatch-year birds, with sub-adults (i.e., second-and third-year eiders and second-year scoters and long-tailed ducks) included with adults in the “after-hatch-year” category. Therefore, they did not provide the same age-specific estimates of fecundity that could be calculated using reproductive rates. Thus, the two independent estimates and distributions of fecundity were similar but not exact equivalents.

We assumed the two independent methods of computing overall adult fecundity described above (i.e., computed from reproductive components and adjusted age ratios from wings) were equally credible, and generated a single probability distribution for overall fecundity by generating 10,000 samples from the two independent fecundity distributions, sampling with equal probability from both.

Sensitivity Analyses

A primary motivation for this assessment was to aid the SDJV in prioritizing investments in monitoring and research to inform harvest management. We used Monte Carlo simulation to propagate uncertainty in individual parameters into probability distributions that described uncertainty in allowable harvest, observed harvest, and the difference between allowable and observed harvest estimates. We interpreted the distribution of the difference between allowable and observed harvest, resulting from the simulations, as uncertainty about whether observed harvest levels were less than, or exceeded, allowable levels. We then conducted a sensitivity analysis to identify the individual demographic parameters whose uncertainty most influenced inferences about the appropriateness of observed harvest (or harvest rate) levels. Within the PTL framework, uncertainty in a given parameter will most greatly influence these inferences if $r_{\max}$ (and hence allowable harvest) is highly sensitive to the parameter, and there is a large degree of uncertainty about the true parameter mean.

We compared the sensitivity of the difference between allowable and observed harvest estimates (or harvest rate for eiders) to uncertainty in the component parameters of $r_{\max}$, fall flight, and observed harvest by comparing the slopes of the linear relationships between the allowable – observed harvest difference and each parameter. So that the slopes were directly comparable, we first standardized values of the probability distribution for each demographic parameter by mean and variance as:

$$Standardized\ (x_i) = \frac{x_i - mean(x_i)}{sd(x)} \tag{7}$$

where x_i represents the sampled parameter from the *i*th iteration of the simulation. We present a hypothetical example in Figure 1 representing a comparison of the difference between allowable and observed harvest and two standardized demographic rates.

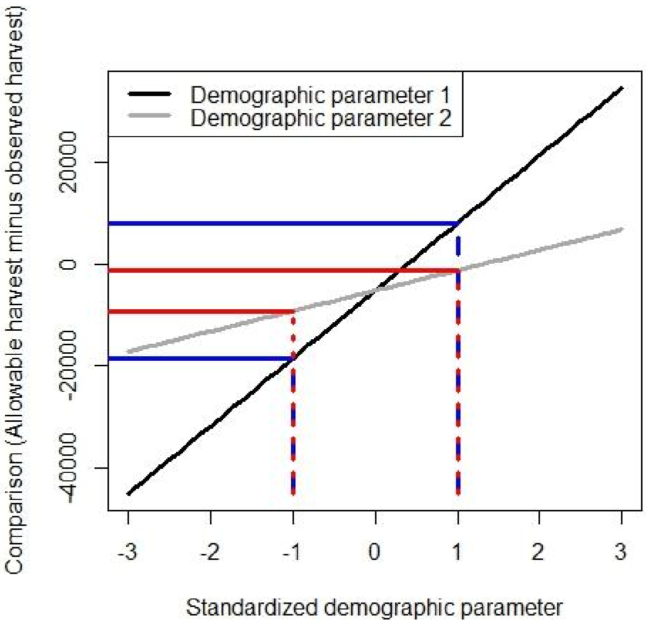

Figure 1. Hypothetical example demonstrating the use of slope for estimating sensitivity of the difference between allowable and observed harvest to demographic parameters.

It is possible to compare relative sensitivities of the difference between the allowable vs. realized harvest to each demographic parameter, by observing the change in *y*-axis values related to a similar change (–1 to +1) in the standardized values of each demographic parameter (*x*-axis). Note that a steeper slope in the relationship of one parameter (black line) results in a greater change in the difference between allowable and observed harvest (blue lines). In comparison, a second parameter (gray line) has little influence on the comparison between the allowable vs. realized harvest (red solid lines), given the same degree of change in the standardized parameters (red dashed lines). A steeper-sloped relationship could result from a high degree of uncertainty in an input demographic parameter relative to other parameters, a high inherent sensitivity of $r_{\max}$ to that parameter given a species' life history traits, or both.

We note that our simulations can result in probability distributions for $r_{\max}$ that include both positive and negative values. Our estimates of $r_{\max}$ are derived from projection matrices and are not theoretically constrained to be ≥0. The PTL formulation, which we use to derive estimates of allowable harvest based on estimated $r_{\max}$, is, however, founded on the assumptions of the logistic growth model. Under the logistic model, a negative value of $r_{\max}$, is not supported. A negative $r_{\max}$ implies that no allowable harvest exists. In such instances, we could reasonably constrain allowable harvest to be ≥ 0. However, our primary purpose was to assess the effect of parameter uncertainty on uncertainty about the difference between observed and allowable harvest. Considering probability distributions that include negative values of allowable harvest, though biologically nonsensical, facilitated that comparison. We, therefore, elected not to constrain values of allowable harvest to be ≥ 0 in the assessment. Subsequently, however, we re-ran the simulations and sensitivity analyses for all populations while constraining allowable harvest to be ≥ 0 to identify any differing conclusions. In those simulations, we fixed the random seed generation so the results were comparable to the initial simulations where values of allowable harvest were not constrained.

Priority Information Needs

We identified three criteria that can be used to prioritize populations and demographic information needs from this assessment. First, we ranked populations based on the degree to which they were at risk of overharvest, as assessed through the proportion of simulations where observed harvest exceeded allowable, and the uncertainty in the estimate of allowable harvest. Second, for individual populations, we categorized the demographic parameters into three levels of priority: highest priority, for those demographic parameters that were highly uncertain and were also influential to the comparison of allowable and observed harvest; medium priority, for those with lower uncertainty that were still highly influential to the harvest comparison (likely due to inherent sensitivity of $r_{\max}$ to the parameter); and low priority, for those with low influence regardless of uncertainty levels.

In order to provide guidance to the SDJV, we arbitrarily determined highest priority parameters to be those whose regression with the difference between allowable and observed harvest resulted in a slope in the top one-third (top 5 of 15) of parameters and whose relative proportional uncertainty (coefficient of variation, or concentration for binomial parameters) was ≥ 0.20. Concentration, a measure of relative proportional uncertainty for binomials, was computed as:

$$Conc = \frac{SD}{\sqrt{p(1-p)}} \tag{8}$$

where SD is the standard deviation and p is the median simulated value of the rate parameter (Link and Barker 2010). Medium priority parameters were those whose regression with the difference between allowable and observed harvest resulted in a slope in the top one-third (top 5 of 15) of parameters, but whose relative proportional uncertainty was <0.20. Other parameters were deemed of low priority from the standpoint of research and monitoring investments.

In the Simulation Results Section, we discuss instances where there were differences in the priority parameters identified by the simulations and sensitivity analyses where allowable harvest was constrained to be ≥0 and those simulations that were not constrained.

SIMULATION RESULTS

American Common Eider

The simulation for American common eiders produced a median $r_{\max}$ of -0.0018 (95% credible intervals -0.1503, 0.1488; Table 2) suggesting that, on average, there is little growth potential for this population under current environmental conditions. Again, we defined environmental conditions as reflecting the overall environmental setting in which the population occurs, to include weather and variation in weather, predator communities, and other density independent factors. Though the credible intervals overlapped, our simulated value of median $r_{\max}$ was considerably lower than the maximum theoretical $r_{\max}$ computed through the DIM of 0.1553 (0.1079, 0.2169).

The percent of simulations where observed harvest was less than allowable harvest was 20% suggesting that, under current harvest policies, there is a substantial risk of overharvest based on the current information available. The median allowable harvest rate was -0.0009 (–0.0812, 0.0692), compared to the observed harvest rate of 0.0256 (0.0208, 0.0327; Table 2). Therefore, our assessment indicates that although harvest rates of hen American common eiders were low, the harvest potential of this subspecies, given current environmental conditions, might be even lower. Most of the information used to describe fecundity of this population was based on field observations and experts from Maine and the Maritimes where the population is believed to be in decline. No information was provided during SDJV Harvest Management Subcommittee deliberations or through the formal elicitation from northern portions of this population’s breeding range in Newfoundland and Labrador, however, limited information suggests that the breeding population there may be increasing (Bowman et al. 2015).

The sensitivity analysis for eiders indicated that uncertainty surrounding comparisons of average harvest potential (allowable harvest) and contemporary harvest levels (observed harvest) is most influenced by adult survival, as well as several components of fecundity including duckling survival, the ratio of juvenile to adult female wings in samples submitted by hunters (i.e., age ratio), hatching success and clutch size (Table 3). This is indicated by the higher slopes for these parameters. Of these, substantial uncertainty and wide probability distributions are associated with age ratio, duckling survival, and hatching success, and they are categorized as the highest priorities for research or monitoring. The probability distributions surrounding median estimates of adult survival and clutch size suggest less uncertainty in these parameters, and they are classified as moderate priority information needs. Priority parameters were identical for simulations where maximum allowable harvest was constrained to be ≥ 0.

Eastern Black Scoter

We estimated a median $r_{\max}$ value of 0.1390 (0.0048, 0.4487) for eastern black scoter (Table 2). The median observed harvest of 29,118 (18,898; 41,641) was 2.7% less than the median allowable harvest of 29,940 (807; 93,753) derived from the simulation; however, allowable harvest was characterized by much greater uncertainty (Table 2). The percent of simulations where observed harvest was less than allowable harvest was 52% (Table 2), indicating that just over one-half of simulations resulted in observed harvest levels consistent with, or more conservative than, harvest objectives. The maximum theoretical $r_{\max}$ from the DIM, 0.1767 (0.1208, 0.2463) was larger than our simulated median $r_{\max}$ though the credible interval of the DIM estimate was entirely contained within the interval of our simulated $r_{\max}$.

The sensitivity analysis for eastern black scoter indicated that fall population size (N^{FF}) and duckling survival meet the criteria established for high priority information needs (Table 3). The high prioritization of population size is attributable to the large uncertainty surrounding the median estimate of fall population size. While this assessment is structured around a post-birth-pulse population, the importance of more reliable estimates of population size is a general conclusion. We recognize that biological, logistical, and fiscal considerations affect the desirability and feasibility of population monitoring at different times during the year. As with eiders, conclusions about appropriate harvest levels were influenced heavily by adult survival as evidenced by its high slope, though a lower degree of uncertainty associated with the estimate of adult survival relegated this parameter to a moderate priority information need. Other moderate priority needs included age ratio, differential vulnerability, and the proportion of hens first breeding at age 2. Of all the moderate priority fecundity parameters, adult survival was most critical to harvest inferences given its higher slope (Table 3). Priority parameters were identical for simulations where maximum allowable harvest was constrained to be ≥0 except that differential vulnerability fell to a low priority.

Western Black Scoter

Median $r_{\max}$ for western black scoters from the simulation was 0.1043 (-0.0502, 0.4052; Table 2). Median observed harvest was 17,137 (9,418; 25,767), while the median allowable harvest was 10,854 (-11,058; 37,219; Table 2). Median observed harvest was 57.7% greater than median allowable harvest (Table 2). Observed harvest was less than allowable harvest in 30% of the simulation iterations. The maximum theoretical $r_{\max}$ from the DIM, 0.1767 (0.1208, 0.2463) was larger than our simulated median $r_{\max}$ though, again, the credible interval of the DIM estimate was entirely contained within the interval of our simulated $r_{\max}$. Sensitivity analysis identified 3 fecundity parameters as the highest priorities for research or monitoring: nest success, duckling survival, and harvest age ratio. Observed harvest was also categorized as a high priority information need though it was less influential on comparisons between allowable and observed harvest than the fecundity parameters. Finally, similar to others species, and as expected given life history traits, adult survival was categorized as a moderate priority information need (Table 3). Priority parameters were identical for simulations where maximum allowable harvest was constrained to be ≥0.

Eastern Surf Scoter

Median $r_{\max}$ for eastern surf scoter was 0.1265 (-0.0260, 0.3418; Table 2) which, coupled with population size estimates (Table 1), resulted in the second highest median allowable harvest estimate of the seven populations studied. Median allowable harvest from the simulations was 23,149 (-9,308; 78,894); however, this population is also believed to be subjected to the second greatest harvest pressure of the seven populations with a median observed harvest of 37,947 (27,237; 51,733). The percent of simulations in which observed harvest was less than allowable harvest was 25% for eastern surf scoters, suggesting a high risk that this population could be overharvested, on average, with respect to harvest objectives (Table 2). The median observed harvest of eastern surf scoters exceeded the median allowable harvest by approximately 15,000 birds, or 63.9% (Table 2). The maximum theoretical $r_{\max}$ from the DIM, 0.1781 (0.1221, 0.2506 was larger than our simulated median $r_{\max}$ though, as for most other populations, the credible interval of the DIM estimate was entirely contained within the interval of our simulated $r_{\max}$. Three parameters of the eastern surf scoter assessment met the criteria for high priority information needs based on the sensitivity analysis: age ratio, nest success, and population size. Of these, age ratio and population size had the highest slopes, and were most influential on comparisons of allowable and observed harvest (Table 3). Adult survival and differential vulnerability were classified as moderate information needs based on the sensitivity analysis. Priority parameters were identical for simulations where maximum allowable harvest was constrained to be ≥0.

Western Surf Scoter

The median simulated value of $r_{\max}$ for western surf scoters was 0.0773 (-0.1344, 0.3137; Table 2). This population is believed to be lightly harvested and the median simulated observed harvest of 9,560 (5,399; 16,939) birds was 32.4% below the median simulated allowable harvest of 14,354 (-61,985; 82,110; Table 2). As was the case for all seven populations, the simulated values for allowable harvest suggested greater uncertainty than for observed harvest. Observed harvest was less than allowable harvest in 59% of simulation iterations (Table 2). The maximum theoretical $r_{\max}$ from the DIM, 0.1781 (0.1221, 0.2506) was larger than our simulated median $r_{\max}$ though its credible interval was again entirely contained within the interval of our simulated $r_{\max}$. Five parameters met the criteria for high priority information needs: population size, clutch size, adult survival, juvenile survival, and differential vulnerability (Table 3). Adult survival was highly influential on the comparison of observed and allowable harvest and was characterized by considerable uncertainty. The probability distribution describing uncertainty in population size was particularly wide and the CV for this parameter was 0.39. No parameters met the criteria for moderate priority information needs. Priority parameters were similar for simulations where maximum allowable harvest was constrained to be ≥0. The only difference for the constrained results was that duckling survival supplanted differential vulnerability and was characterized as a moderate priority information need.

White-winged Scoter

No experts were comfortable speculating on population-wide mean values for reproductive rates of white-winged scoters, though one individual did provide feedback on age-specific breeding propensity and age of senescence. Therefore, we used literature estimates with associated standard errors to create probability distributions for clutch size, hatching success, nesting success, and duckling survival (Appendix B). We assumed that hatching success and nesting success were correlated ($r = 0.8$), and therefore sampled these values from a multivariate normal distribution.

Median simulated $r_{\max}$ for white-winged scoter was 0.0509 (-0.1217, 0.2335; Table 2). Median simulated allowable harvest was 13,054 (-68,824; 61,072) while median simulated observed harvest was 24,399 (12,599; 37,647), almost double (86.9% greater than) the median allowable level. The percent of simulations in which observed harvest was less than allowable harvest was 36% (Table 2). The estimate of allowable harvest was extremely uncertain (Table 2). The maximum theoretical $r_{\max}$ from the DIM, 0.1721 (0.1178,0.2443) was larger than our simulated median $r_{\max}$ though the credible interval overlapped the interval of our simulated $r_{\max}$. Two parameters met the criteria for a high priority information needs for white-winged scoters, observed harvest and nest success (Table 3). While observed harvest met the criteria for highest priority because it ranked in the top 5 absolute slopes and its CV was > 0.20, the influence of its uncertainty on the harvest comparison was lower than the 3 parameters that were categorized as moderate priority information needs. As in most of the other sea duck populations investigated, adult survival was highly influential but not thought to be highly uncertain. The other moderate priority parameters were differential vulnerability and hatching success, both components of the two alternative fecundity estimation methods that were weighted equally and used to generate the overall fecundity probability distribution. Differences in priority parameters when maximum allowable harvest was constrained to be ≥ 0 were more substantial for white-winged scoters: population size and unadjusted age ratio from wing samples were elevated to moderate priorities, while nest success and hatching success fell to low priorities.

Long-tailed Duck

Median simulated $r_{\max}$ for long-tailed duck was -0.0669 (-0.2882, 0.1215). Reproductive rate estimates for long-tailed ducks from the literature were extremely low compared to all species/populations other than common eiders (Appendix A7). Fecundity estimates from age ratios (median = 0.45) were higher than the limited published values of component reproductive rates would suggest, but the final probability distribution for fecundity used in the simulation, as for the other populations, incorporated both adjusted age ratios and fecundity modeled from its component reproductive rates. The median value of the allowable harvest distribution was very low: -48,966 (-202,663; 60,561), and highly uncertain (Table 2). The percent of simulations in which observed harvest, 43,044 (32,151; 57,589), was less than allowable harvest was only 5% (Table 2). The maximum theoretical $r_{\max}$ from the DIM, 0.1843 (0.1266, 0.2560) was larger than our simulated median $r_{\max}$.

Four parameters, adult survival, population size, nest success, and second-year survival met the requirements for high priority information needs based on their influence on the comparison of allowable to contemporary observed harvest levels. Population size ranked as having the second highest slope of the 15 parameters evaluated in the sensitivity analysis. The proportion of first time breeders breeding at age 2 was classified as a moderate priority information need. Adult survival had both high uncertainty and a very large slope value, about twice the magnitude of its nearest competitor. Expert elicitation resulted in lower median estimates of adult survival for long-tailed ducks than the other species and the lower survival values were highly influential in the sensitivity analysis. As with white-winged scoters, differences in priority parameters identified through analyses where maximum allowable harvest was constrained to be ≥0 were more substantial for long-tailed ducks. For this population, observed harvest and rate of crippling loss became moderate priority information needs while nest success and the probability of first breeding at age 2 were classified as low priorities.

DISCUSSION

As expected, the subcommittee’s literature search revealed a dearth of empirical data for most of the species and populations under consideration. The few studies that provided such information were conducted in areas that were accessible and/or known to be important breeding areas (e.g., Redberry Lake, Saskatchewan for white-winged scoters), and likely were not representative at the population scale. Breeding ranges of North American scoter species and long-tailed ducks have only recently been delineated adequately through the SDJV's satellite telemetry projects (e.g., the Atlantic and Great Lakes sea duck migration study; http://seaduckjv.org/science-resources/atlantic-and-great-lakes-sea-duck-migration-study/), and the telemetry data indicate that much of the breeding range of these species lies outside of areas that are currently surveyed annually to estimate breeding waterfowl abundance in North America (Bowman et al. 2015). Those data also illustrate that reproductive parameter estimates for scoters and long-tailed ducks are absent across the great majority of their breeding ranges. Consequently, empirical estimates that were available to us did not necessarily represent population-wide demographics, nor did they adequately characterize the uncertainty around those parameters.

Our effort to augment the empirical data by eliciting opinions from subject-matter experts met with limited success. North America has few sea duck experts given the difficulties in studying these species in remote areas, and although we solicited input from most of them, several were unable to participate and others were reluctant to provide an educated guess for certain parameters, even though the elicitation process allowed them to express their lack of confidence. Most of the experts who did respond only provided estimates for a subset of parameters. Nonetheless, we believe that including the experts' responses resulted in probability distributions for parameters that better represented uncertainty. Combining the elicited data and limited available empirical data also ensured that the probability distributions reflected parameter values that were presumed representative of the population.

Accurate quantification of uncertainty is crucial both to evaluating the results of the assessment for harvest management and to identifying and prioritizing information needs, and the results of the assessment are conditional on adequate descriptions of uncertainty levels for each parameter. In the case of fecundity, where we incorporated 2 independent measures of fecundity into the overall probability distribution describing uncertainty in this parameter, the results of the assessment are also conditional on both estimation methods being retained in the analysis. If one of the 2 methods were used as the sole source for information on fecundity, in addition to dropping the parameters for the other method from consideration, the relative sensitivity analysis rankings could also shift somewhat. Other assumptions made in characterizing uncertainty, such as age at first breeding for different species or conversion of winter to fall population sizes, were made on the basis of published literature values or limited empirical data, but did constrain uncertainty descriptions for some populations.

The maximum theoretical values of $r_{\max}$ computed by the DIM exceeded the median values from our simulations for all populations. The DIM is based on fundamental relationships between survival, fecundity, and generation time that apply broadly across taxonomies. DIM estimates of $r_{\max}$ can be thought of as theoretical maximum values for a species. The DIM estimates of $r_{\max}$ may not apply to a specific population at a specific time. The most useful estimates of $r_{\max}$, from a management perspective, are context-specific, in that $r_{\max}$ should reflect the highest growth rate attainable in the specific environmental setting within with the population occurs. The differences between median values of $r_{\max}$ simulated in this assessment and the maximum theoretical values derived through the DIM may reflect suboptimal environmental conditions experienced by the population, the use of parameter values that do not reflect population growth unconstrained by density or harvest, or use of parameters values that do not reflect of the entire population, but rather a subpopulation with lower growth potential than the entire population.

Comparison of Allowable and Observed Harvest

Our PTL (allowable harvest or harvest rate) estimates were very uncertain, much more so than the estimates of observed harvest that were based on national harvest survey programs in the U.S. and Canada as well as generally less reliable estimates of subsistence harvest. In fact, for every population examined, the 95% credible interval for observed harvest fell entirely within the 95% credible interval for allowable take. This further illustrates that although a better understanding of the components of observed harvest (bias in fall and winter harvest estimates, comprehensive and up-to-date estimates of subsistence harvest, crippling loss) is desirable, a reduction in uncertainty in allowable harvest would likely be more informative to harvest management. For those populations where subsistence harvest estimates far exceeded sport harvest a reasonable case can be made for better subsistence harvest estimates.

The percentage of simulations for each of the seven sea duck populations in which observed harvest was less than allowable harvest (Table 2) provides a general framework for assessing the relative risk of each population to overharvest. Any conclusions are, of course, conditional on the probability distributions used to characterize uncertainty in each demographic parameter, the assumptions and limitations of the deterministic PTL framework applied (see below), and the implied harvest objective of maximum sustained yield. According to the simulation results, we subjectively categorized the seven populations into three categories reflecting the relative risk of overharvest. Highest risk populations include the long-tailed duck and American common eider. Moderate risk populations include eastern surf scoter, western black scoter, and white-winged scoter, and lowest risk populations include the eastern black scoter and western surf scoter.
This assessment addresses one of the recommendations of the 2013 Supplemental Environmental Impact Statement (SEIS) on annual migratory bird hunting regulations (USFWS 2013). The SEIS considered alternatives regarding the continuance and review of special season structures such as the special sea duck season in the Atlantic Flyway. The proposed action that the USFWS adopted called for periodic review of the special seasons to determine whether they are still justifiable. Although this assessment provides attempts to summarize current understanding of the harvest potential of American common eiders, the three scoter species, and long-tailed ducks, use of the results of this assessment to guide sea duck harvest management is outside the purview of the Sea Duck Joint Venture. Appropriate application of these results as well as final resolution of important policy issues pertaining to harvest management will be determined by the broader harvest management community through normal regulatory processes. However, this assessment emphasizes the need to reduce uncertainty in key demographic parameters to ensure harvest levels for these species are sustainable.

The most difficult aspect of our assessment was specifying distributions for demographic parameters. As we described above, demographic rates used in the estimation of $r_{\max}$ are rarely observed in nature, particularly in the case of a harvested population. While, through the elicitation, we attempted to adjust rates to reflect conditions under which $r_{\max}$ would apply, we acknowledge the difficulty in doing so, and that survival and reproductive parameters, and $r_{\max}$ may be under-estimated. In addition, we assumed $\theta = 1$ for all species, which also adds conservatism to our PTL assessment. In contrast, the DIM represents a theoretical maximum of

$r_{\max}$ and may overestimate harvest potential under prevailing environmental conditions. Therefore, applying our results to harvest management should carefully consider management objectives and risk tolerance with respect to population viability and hunting opportunity.

For the benefit of the managers who will be considering the harvest management implications of this assessment, key assumptions and limitations of the PTL approach are detailed below.

1) PTL assumes that carrying capacity is not changing. Carrying capacity, however, varies in response to environmental variation or to directional system changes. PTL is robust to changes in carrying capacity as long as F_o seeks to maintain the population at a fixed fraction of carrying capacity. This can be accomplished through a harvest rate formulation of PTL or through a time-specific, total harvest-based formulation which scales allowable harvest in accordance with population size (Runge et al. 2009, Johnson et al. 2012). For the latter, it is critical that periodically updated population estimates are available in order to adjust PTL.

2) PTL assumes $r_{\max}$ is fixed. Like carrying capacity, PTL is also sensitive to changes in $r_{\max}$. As we view $r_{\max}$ as being a reflection of species life history traits expressed within a specific environmental setting, it is possible that $r_{\max}$ could change over time. There is no remedy for this problem other than conservatism in decision-making and vigilance in monitoring (Runge et al. 2009) to ensure that periodic updating of $r_{\max}$ estimates is possible.

3) PTL as formulated assumes that hunting mortality is additive to other forms of mortality. This is a conservative assumption; however, PTL can be re-framed to allow for partial compensation (Williams 2012). Given the life history characteristics of sea ducks, and in the interest of simplification, we did not consider compensation or partial compensation of harvest mortality.

4) PTL, as applied here, is based on highly simplified models of population growth: the deterministic logistic model or deterministic theta-logistic model. While the methods we used to estimate $r_{\max}$ allow for limited incorporation of age/stage structure, sparse monitoring data for sea ducks limits rigorous examination of the effects of age/stage structure, population inertia/transient dynamics, environmental and other stochastic effects, cohort-targeted harvest, and other factors affecting the dynamics of structured populations. PTL was selected as an initial assessment framework because the simplicity of the underlying model permits broad application to a variety of sea duck species.

5) Though we used the theta-logistic PTL formulation, a lack of estimated values of θ for any of the populations of interest led us to the simplifying assumption that $\theta = 1$. Given the life history characteristics of sea ducks it is likely that true values of θ are >1, and that density dependence in these species is strongest as population size approaches carrying capacity. The functional form of the relationship between population density and growth rate and the proximity of the population to carrying capacity can have significant implications for harvest potential, increasing allowable take levels as the population approaches carrying capacity for more k-selected species. If θ is >1 in the populations that we simulated, our assessment will be conservative since we assumed $\theta = 1$ for all species.

6) PTL provides conservation assurances at the geographic resolution at which it is applied, or at coarser resolutions. No conservation assurances apply at finer geographic resolutions.

Priority Information Needs and Recommendations

Priority information needs identified through the sensitivity analysis for each sea duck population reflect the influence of uncertainty in individual parameters on inferences about appropriate harvest levels, given the harvest objective we assumed. In many instances, the probability distributions we described to characterize uncertainty of input parameters were heavily dependent on elicitation values that represented informed opinion of sea duck research ecologists and managers. Conclusions drawn from this analysis should be tempered by that fact, and overly fine-grained interpretation of the results should be avoided. Regardless, the general high, moderate, and low priority categorization of information needs for the 7 populations represents our best guidance to the SDJV about where reductions in uncertainty could most significantly improve harvest inferences and decision making. We recognize that other factors must be considered by the JV in order to prioritize actual resource investments in research and monitoring, including the feasibility and cost of obtaining required information, the capacity and interests of individuals and agencies engaged in research and monitoring activities, and available funds. Accordingly, we offer the SDJV the following general observations and recommendations for research and monitoring investments.

Our comparisons of allowable and observed harvest were highly influenced by adult survival in all populations. This was expected, given the life history characteristics of sea ducks and the structure of population projection models (Flint 2015), and is a reflection of the sensitivity of $r_{\max}$ to variation in adult survival. For most populations, the relative proportional uncertainty of adult survival was small in comparison to other parameters, suggesting low uncertainty about the true population median value. This led to adult survival being categorized as a moderate priority information need for all populations, except eastern and western surf scoter and long-tailed duck, where it was a high priority. While the conclusion that uncertainty about adult survival is low relative to other demographic parameters is likely reasonable, we recognize that few data exist to derive population-wide estimates of adult female survival for any sea duck populations. A possible exception is American common eider, but even in this case, estimation of population-level survival requires a combination of spring and preseason banding data sets and corresponding assumptions. The high sensitivity of maximum growth rate to adult female survival argues in favor of increased efforts to estimate this critical parameter and verify that median values and uncertainty distributions used in the assessment are reasonable. We recognize the challenges in banding sufficient preseason samples of adult female sea ducks and that estimation based on other banding periods or based on satellite telemetry data may be more feasible. Efforts to verify and scale published allometric relationships between body size and adult survival (Johnson et al. 2012) for sea ducks may also be of value if estimation of adult female survival proves intractable for some species.

Overall, uncertainty about fecundity had more influence on comparisons of allowable and observed harvest than adult survival (Table 3, see computed parameters), however, fecundity cannot be directly studied or monitored, rather it must be estimated from component parameters. The high influence of overall fecundity on our conclusions about harvest levels reflected wide probability distributions around this composite parameter. Those probability distributions, in turn, were a result of propagation of the uncertainty in individual fecundity parameters.

The probability distributions for fecundity used in this assessment were a composite of probability distributions derived from the 2 independent methods of estimating fecundity: from component reproductive rates and from adjusted female harvest age ratios. For several populations, uncertainty in individual reproductive rates greatly influenced inferences about harvest levels. Examples include duckling survival in American common eiders, and nest success and duckling survival of western black scoters. Estimating the true population means for individual reproductive rates such as nest success and duckling survival is likely impossible for any of the populations of interest. Local studies strategically conducted across the breeding range would provide spatial and temporal replication, but replication sufficient to describe these processes at a population scale may be prohibitively costly given their vast and remote breeding ranges. A possible exception may be American common eider given the colonial nesting habits of this species and a generally more accessible breeding distribution in comparison to the other populations assessed here. Emerging technologies that allow for remote monitoring of individual bird activity and physiology may open new avenues for research on the components of fecundity in these species.

Uncertainty about the uncorrected female harvest age ratio and/or differential vulnerability estimates, like individual reproductive rates, appeared as high or moderate priorities for all populations except long-tailed duck and white-winged scoter. Improved estimates of fecundity based on adjusted age ratios may be more tractable than estimation of individual recruitment parameters at the population scale, and we recommend increased efforts to improve estimates of adjusted harvest age ratios of female sea ducks. This would require increased preseason banding efforts, or some other novel method, to develop sea duck-specific estimates of differential vulnerability. For some species, differential vulnerability is suspected to vary between the U.S. and Canada in relation to cultural differences affecting the targeting of different sex and age cohorts by sport hunters. Differences in vulnerability of the female age classes in relation to sport and subsistence harvest should also receive increased attention. Efforts to increase the sample of sea duck wings obtained during annual harvest surveys in the U.S. and Canada would also be necessary to improve the utility of adjusted harvest age ratios as measures of fecundity for these populations. Increasing the wing sample is especially critical, and would be particularly difficult, in the Pacific Flyway, where sea duck harvest is low and sample sizes of hunter-submitted wings are small. Finally, a method of scaling age-ratios to be representative of conditions under which r_{max} could be attained would need to be devised.

Uncertainty about population size was also categorized as a high priority based on its significance to harvest inferences in four of the six populations where it factored into the assessment. The PTL assessment for American common eiders was based on harvest rate and, therefore, population size did not factor into the assessment for this population. The four populations where uncertainty in population size most affected harvest inferences were eastern black scoter, eastern surf scoter, western surf scoter, and long-tailed duck. Bowman et al. (2015) recently reviewed population status and monitoring programs for North American sea ducks. They report that little information exists to assess population status or trend for the eastern black scoter given a breeding range that poorly coincides with the spatial extent of population monitoring programs. Similarly, population monitoring programs are not ideally designed spatially for western surf scoters and limited data suggest some discrepancies among regional trends in breeding and wintering populations. The breeding range of eastern surf scoter corresponds better to the spatial extent of current monitoring programs and indications from limited data are that this population has been stable or increasing since 1990. Lastly, the vast and remote breeding range of the long-tailed duck has precluded effective breeding population abundance monitoring. In general, but with some exceptions, current population monitoring programs for sea ducks are poorly timed to sea duck breeding chronology, suffer from poorly understood species misclassification biases (especially for scoters), poorly account for detection and availability biases, and, in the case of winter surveys, are challenged by highly aggregated and variable distributions (Bowman et al. 2015, Silverman et al. 2013).

We recommend continued efforts to integrate the operating procedures and analysis of presently disparate breeding population surveys for sea ducks. In many instances long-term records of sea duck observations collected during surveys targeting other species (e.g., arctic and sub-arctic nesting geese) have not been analyzed, nor have consistency of protocols across surveys been formally assessed. Recent experimental breeding population surveys in the barrenlands of the Northwest Territories and Nunavut utilize methods complementary to the Waterfowl Breeding Population and Habitat Survey (WBPHS) and cover previously unsurveyed sea duck habitats. In addition, an ongoing review of the WBPHS will address species prioritization and the geographic extent of this critical annual survey. While available resources may preclude major shifts in annual spatial coverage or timing of the WBPHS it is possible that new procedures to accommodate climate change or address differences in waterfowl breeding chronologies could help address some limitations to breeding sea duck monitoring protocols. In addition, wintering surveys are receiving increased attention as alternative means of improving sea duck monitoring capacity. Significant challenges with aggregated distributions, mixed species flocks, species misclassification, and limited information on detection and availability processes hamper interpretation of winter surveys (Silverman et al. 2012). However, given the importance of uncertainty in population size estimates to harvest inferences, we recommend continued efforts to improve winter population surveys. Remote sensing-based designs offer promise in addressing many challenges of both wintering and breeding surveys, as well as increasing safety of survey personnel. Thus, we recommend continued SDJV support of remote sensing-based designs and improved automation of imagery post-processing and feature identification.

Uncertainty about observed harvest estimates was categorized as a high priority for only two populations: western black scoter and white-winged scoter. Nearly all of the western black scoter harvest results from subsistence hunting in Alaska, where harvest survey methodology relies on hunters to identify species correctly when they report their harvest. Rothe et al. (2015) stated that some of Alaska’s subsistence hunters probably report all scoters they harvest as black scoters, thereby inflating harvest estimates for that species. Efforts are currently underway to improve the Alaska subsistence harvest survey, which could result in reduced uncertainty around western black scoter harvest estimates. Continental harvest of white-winged scoters is less than that of any other species we assessed; thus, harvest estimates for that species would benefit from larger wing samples, as recommended above.

Based on the general categorization of the risk of overharvest for the seven populations assessed here, we recommend that the SDJV prioritize research and monitoring efforts on the long-tailed duck and American common eider. Based on limited breeding population data from surveys that covered only a small fraction of the breeding range of this species in North America, Bowman et al. (2015) speculated that the long-tailed duck population declined during the 1980s and 1990s, but has, since the early 2000s, stabilized. In contrast, our assessment indicates that r_{max} for the species is <0, indicative of a population with no long-term growth potential. Like Bowman et al. (2015), however, we have low confidence in our conclusion regarding the status of this long-tailed duck. We also lack confidence in estimates of allowable harvest for this species given that no biological hypothesis had been advanced to corroborate an ongoing high rate of population decline. We have greater confidence in our assessment of allowable harvest of American common eiders, though limited data suggest our assessment may be more applicable to the segment of this population breeding in Maine and the Maritimes. Here several biological hypotheses have been advanced, from gull predation on ducklings to climate-related regime shifts in the Gulf of Maine, which could account for a low r_{max} and low growth potential. Given the high potential for overharvest we recommend increased emphasis by the SDJV in reducing uncertainty in priority demographic parameters identified for this eider subspecies.

We also recommend that these PTL assessments be replicated at some reasonable time interval (perhaps five years), incorporating additional information derived through SDJV-funded or other studies. The results and conclusions of this study are conditional on the uncertainty described for all the demographic parameters for each of the seven populations. Changing the uncertainty distribution of one parameter can affect the relative sensitivity of harvest inferences to that parameter as well as others and would be expected to change the relative monitoring and research priorities described here. We note also that this assessment framework could also be used in a more prospective manner. By speculating about the expected reduction in uncertainty in individual parameters that are anticipated from specific research and monitoring proposals, the JV could use this framework to contrast individual proposals based on their expected effect on the comparison of allowable and observed harvest.

A more formal analysis of the effect of uncertainty and the reduction of uncertainty on harvest decision-making capability could be achieved through an assessment of the expected value of perfect information (EVPI) or expected value of partial information (EVPXI; Runge et al. 2011, Johnson et al. 2014). However, these approaches require a fully-specified decision framework to include a harvest management objective, a set of actions (frequently different regulatory packages in a harvest management context), and system and control models that specify the effects of regulatory actions and uncertainty about those effects. While a modeling approach as simple as PTL could provide a component of this larger decision framework, other elements, some technical, some policy-based, are presently unspecified. Nevertheless, as the information base for sea ducks grows and increasing attention is directed toward informed harvest decision-making for these species, it may be possible to more fully develop decision frameworks for sea ducks, apply more formal approaches to the evaluation of uncertainty, and provide better guidance to the SDJV on research and monitoring priorities to support effective harvest management.

ACKNOWLEDGMENTS

We would like to thank the many individuals who provided input, guidance and information regarding sea ducks, including: R. Alisauskas, B. Allen, B. Avers, T. Bowman, S. De La Cruz, P. Flint, S. Gilliland, J.-F. Giroux, J. Kelley, D. Kraege, M. Mallory, D. McAuley, R. Milton, S.

Olson, J. Osenkowski, E. Osnas, J. Pearce, E. Reed, A. Roberts, M. Robertson, D. Rosenberg, J. Schamber, K. Sullivan and R. Trost. We also thank M. Runge and F. Johnson for guidance and expertise regarding various approaches for assessing allowable take, and the leadership of K. Dickson for guiding the SDJV through a process for developing greater focus on supporting the information needs of harvest and habitat managers. This project was funded through the U.S. Fish and Wildlife Service, Division of Migratory Bird Management (Headquarters and Northeast Region), and the SDJV. This report is submitted to the SDJV in fulfillment of project No. 139. The findings and conclusions are those of the authors and do not necessarily represent the views of the U.S. Fish and Wildlife Service.

LITERATURE CITED

Alisauskas, R. T., J. J. Traylor, C. J. Swoboda, and F. P. Kehoe. 2004. Components of population growth rate for white–winged scoters in Saskatchewan, Canada. Animal Biodiversity and Conservation 27:451-460.

Alison, R. M. 1975. Breeding biology and behavior of the oldsquaw (*Clangula hyemalis*). Ornithological Monographs No. 18. American Ornithologists' Union.

Bellrose, F. C. 1980. Ducks, Geese and Swans of North America. 3rd edition. Wildlife Management Institute, Washington, D.C.

Bengtson, S. A. 1971. Variations in clutch-size in ducks in relation to the food supply. Ibis 113:523-526.

Bengtson, S. A. 1972. Reproduction and fluctuations in the size of duck populations at Lake Myvatn, Iceland. Oikos 23:35–58.

Boomer, G. S., G. Zimmerman, N. Zimpfer, P. Garrettson, M. Koneff, T. Sanders, K. Magruder, and J. A. Royle. 2013. Band reporting probabilities for mallards recovered in the United States and Canada. Journal of Wildlife Management 77:1059–1066.

Bordage, D. and J.-P. L. Savard. 1995. Black scoter (*Melanitta nigra*). *In* The Birds of North America, No. 177 (A. Poole and F. Gill, eds.). Academy of Natural Sciences, Philadelphia, and The American Ornithologists' Union, Washington, D.C.

Bowman, T. D., E. D. Silverman, S. G. Gilliland, and J. B. Leirness. 2015. Status and trends of North American sea ducks: Reinforcing the need for better monitoring. Pages 1-28 *in* J.-P. L. Savard, D. V. Derksen, D. Esler, and J. M. Eadie (editors). Ecology and conservation of North American sea ducks. Studies in avian biology (no. 46). CRC Press, Boca Raton, FL.

Boyd, W. S., T. D. Bowman, J.-P. L. Savard, and R. D. Dickson. 2015. Conservation of North American sea ducks. Pages 529-559 *in* J.-P. L. Savard, D. V. Derksen, D. Esler, and J. M. Eadie (editors). Ecology and conservation of North American sea ducks. Studies in avian biology (no. 46). CRC Press, Boca Raton, FL.

Brown, P. W., and L. H. Fredrickson. 1989. White-winged scoter, *Melanitta fusca*, populations and nesting on Redberry Lake, Saskatchewan. Canadian Field-Naturalist 103(2):240-247.

Brown, P. W., and L. H. Fredrickson. 1997. White-winged scoter (*Melanitta fusca*). *In* The birds of North America online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/274doi:10.2173/bna.274

Brownie, C., D. R. Anderson, K. P. Burnham, and D. R. Robson. 1985. Statistical Inference from Band Recovery Data – A Handbook. Second Edition. U. S. Fish and Wildlife Service Resource Publication 156, Washington, D.C., USA.

Caithamer, D. F., M. Otto, P. I. Padding, J. R. Sauer, and G. H. Haas. 2000. Sea ducks in the Atlantic flyway: population status and a review of special hunting seasons. U. S. Fish and Wildlife Service, Washington, D.C., USA.

Caswell, H. 2001. Matrix population models: construction, analysis, and interpretation, Second edition. Sinauer, Sunderland, Massachusetts, USA.

Conroy, M. J., and J. T. Peterson. 2013. Decision making in natural resource management: a structured, adaptive approach. Wiley-Blackwell, Oxford, UK.

Cowardin, L. M. and R. J. Blohm. 1992. Breeding population inventories and measures of recruitment. Pages 423-445 *in* Ecology and management of breeding waterfowl (B. D. J. Batt, A. D. Afton, M. G. Anderson, C. D. Ankney, D. H. Johnson, J. A. Kadlec and G. R. Krapu, Eds.), University of Minnesota Press, Minneapolis, Minnesota, USA.

Delignette-Muller, M. L., and C. Dutang. 2015. fitdistrplus: An R package for fitting distributions. Journal of Statistical Software 64: 1-34. URL http://www.jstatsoft.org/v64/i04/.

Descamps, S., M. R. Forbes, H. G. Gilchrist, O. P. Love, and J. Bêty. 2011. Avian cholera, post-hatching survival and selection on hatch characteristics in a long-lived bird, the common eider *Somateria mollissima*. Journal of Avian Biology 42:39-48.

Evenson, J., H. Tschaekofske, B. Murphie, T. Cyra, and D. Kraege. 2013. Status and summary of the 2013 WDFW winter sea duck aerial survey detectability project – Phase 3. Washington Department of Fish and Wildlife unpublished report.

Flint, P. L., 2015. Population dynamics of sea ducks. Pages 63-96 *in* J.-P. L. Savard, D. V. Derksen, D. Esler, and J. M. Eadie (editors). Ecology and conservation of North American sea ducks. Studies in avian biology (no. 46). CRC Press, Boca Raton, FL.

Fox, A. D., Æ. Petersen, and M. Frederiksen. 2003. Annual survival and site-fidelity of breeding female Common Scoter *Melanitta nigra* at Mývatn, Iceland, 1925–58. Ibis 145(2):E94-E96.

Gendron, M.H., and A.C. Smith. 2014. National Harvest Survey web site. Bird Populations Monitoring, National Wildlife Research Centre, Canadian Wildlife Service, Ottawa, Ontario. (URL: http://www.ec.gc.ca/reom-mbs/enp-nhs/index.cfm?do=b&lang=e)

Gilliland, S. G., H. G. Gilchrist, R. F. Rockwell, G. J. Robertson, J. L. Savard, F. Merkel, and A. Mosbech. 2009. Evaluating the sustainability of harvest among northern common eiders Someteria mollissima borealis in Greenland and Canada. Wildlife Biology 15:24-36.

Gilliland, S. G., E. Reed, C. Lepage, L. Lesage, J.-P. L. Savard, and K. McAloney. 2011. Harvest and survival of eastern North America surf scoters. Abstract of presentation at the 4rd North American Sea Duck Conference, Seward, AK. http://seaduckjv.org/pdf/sea_duck_conference_2011_abstracts_web.pdf

Gilliland, S., K. McAloney, S. Boyd, J.-P. Savard, E. Reed, D. Saalfeld, and M. Perry. 2013. Satellite tracking of Atlantic black scoters: What we have learned in just a few short years. Presentation at the Ecology and Conservation of North American Waterfowl Symposium, Memphis, TN.

Goudie, R. I., A. V. Kondratyev, S. Brault, M. R. Petersen, B. Conant, and K. Vermeer. 1994. The status of sea ducks in the North Pacific Rim: Toward their conservation and management. Transactions North American Wildlife Natural Resoource Conference 59:27-45.

Goudie, I., G.J. Robertson, and A. Reed. 2000. Common Eider (*Somateria mollissima*). *In* The Birds of North America No. 546 (Poole, A. and Gill, F. eds.), Philadelphia, and American Ornithologists' Union, Washington D.C.: Academy of Natural Sciences.

Iles, D. T., R. F. Rockwell, P. Matulonis, G. J. Robertson, K. F. Abraham, J. C. Davies, and D.N. Koons. 2013. Predators, alternative prey and climate influence annual breeding success of a long-lived sea duck. Journal of Animal Ecology 82:683-693.

Johnson, F. A., G. Hagan, W. E. Palmer, and M. Kemmerer. 2014. Uncertainty, robustness, and the value of information in managing a population of northern bobwhites. Journal of Wildlife Management 78:531-539.

Johnson, F. A., M. A. H. Walters, G. S. Boomer. 2012. Allowable levels of take for the trade in Nearctic songbirds. Ecological Applications 22:1114-1130.

Kehoe, P., D. Caithamer, J. Myers, R. Burrell, B. Allen, G. Haas, J. Goldsberry, M. Bateman, A. Bourget, and E. Butler. 1994. Status of Sea Ducks in the Atlantic Flyway. Canadian Wildlife Service, Fredericton, New Brunswick, Canada.

Kellett, D. K., and R. T. Alisauskas. 2014. Population ecology of long-tailed ducks at Karrak Lake, Nunavut. Poster Presentation, Fifth International Sea Duck Conference, Rejkavik, Iceland.

Kellett, D. K., R. T. Alisauskas, K. R. Mehl, K. L. Drake, J. J. Tray- lor, and S. L. Lawson. 2005. Body mass of Long-tailed Ducks (*Clangula hyemalis*) during incubation. Auk 122:313–318.

Korschgen, C. E. 1977. Breeding stress of female eiders in Maine. Journal of Wildlife Management 41:360–373.

Krementz, D. G., J. E. Hines, and D. F. Caithamer. 1996. Survival and recovery rates of American Eiders in east North America. Journal of Wildlife Management 60:855-862.

Leafloor, J. O., J. E. Thompson, and C. D. Ankney. 1996. Body mass and carcass composition of fall migrant Oldsquaws. Wilson Bulletin 108:567–572.

Link, W. A., and R. J. Barker. 2010. Bayesian Inference with Ecological Applications. Academic Press, Amsterdam.

McBride, M. F., S. T. Garnett, J. K. Szabo, A. H. Burbidge, S. H. M. Butchart, L. Christidis, G. Dutson, H. A. Ford, R. H. Loyn, D. M. Watson, and M. A. Burgman. 2012. Structured elicitation of expert judgments for threatened species assessment: A case study on a continental scale using email. Methods in Ecology and Evolution 3: 906-920.

Morrier, A., L., A. Lesage, A. Reed, and J.-P. L. Savard. 1997. Etude sur l’ecologie de la Macreuse a` front blanc au lac Malbaie, Reserve des Laurentides—1994–1995. Technical Report Series No. 301, Canadian Wildlife Service, Quebec Region, Quebec, Canada.

Natcher, D. C., L. Felt, K. Chaulk, A. Procter, and the Nunatsiavut Government. 2011. Monitoring the domestic harvest of migratory birds in Nunatsiavut, Labrador. Arctic 64:362–366.

NAWMP. 2012. North American Waterfowl Management Plan 2012: People Conserving Waterfowl and Wetlands. U.S. Department of the Interior, Washington, D.C., USA.

Nelson, A. L., and A. C. Martin. 1953. Gamebird weights. Journal of Wildlife Management 17:36–42.

Niel, C. and J-D Lebreton. 2005. Using demographic invariants to detect overharvested bird populations from incomplete data. Conservation Biology 19:826-835

Padding, P., and J. A. Royle. 2012. Assessment of bias in US waterfowl harvest estimates. Wildlife Research 39:336–342.

R Core Team. 2015. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051-07-0, URL http://www.R-project.org/

Robertson, G.J. and J.-P.L. Savard. 2002. Long-tailed Duck (*Clangula hyemalis*). *In* The Birds of North America, No. 651 (A. Poole and F. Gill, Eds.). The Birds of North America, Inc., Philadelphia, PA.

Rothe, T. C., P. I. Padding, L. C. Naves, and G. J. Robertson. 2015. Harvest of sea ducks in North America: A contemporary summary. Pages 417-467 *in* J.-P. L. Savard, D. V. Derksen, D. Esler, and J. M. Eadie (editors). Ecology and conservation of North American sea ducks. Studies in avian biology (no. 46). CRC Press, Boca Raton, FL.

Runge, M. C., S. J. Converse, and J. E. Lyons. 2011. Which uncertainty? Using expert elicitation and expected value of information to design an adaptive program. Biological Conservation 144:1214-1223.

Runge, M. C., W. L. Kendall, and J. D. Nichols. 2004. Exploitation. Pages 303-328 *in* W. J. Sutherland, I. Newton, and R. E. Green, editors. Bird ecology and conservation: A handbook of techniques. Oxford University Press, Oxford, United Kingdom.

Runge, M.C., J. R. Sauer, M. L. Avery, B. F. Blackwell, M. D. Koneff. 2009. Assessing allowable take of migratory birds. Journal of Wildlife Management 73:556–565.

Safine, D. E. 2005. Breeding ecology of white-winged scoters on the Yukon Flats, Alaska. MS Thesis. University of Alaska, Fairbanks, USA.

Savard, J.-P. L., D. Bordage, and A. Reed. 1998. Surf scoter (*Melanitta perspicillata*). *In* The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology. http://bna.birds.cornell.edu/bna/species/363doi:10.2173/bna.363

Schamber, J. L., P. L. Flint, J. B. Grand, H. M. Wilson, and J. A. Morse. 2009. Population dynamics of long-tailed ducks breeding on the Yukon-Kuskokwim Delta, Alaska. Arctic 62:190-200.

Schamber, J. L., F. J. Broerman, and P. L. Flint. 2010. Reproductive ecology and habitat use of Pacific black scoters (*Melanitta nigra americana*) nesting on the Yukon-Kuskokwim Delta, Alaska. Waterbirds 33:129-139.

Sea Duck Joint Venture Continental Technical Team. 2003. Species status reports. U.S. Fish and Wildlife Service, Anchorage, Alaska, USA.

Sea Duck Joint Venture Management Board. 2008. Sea Duck Joint Venture Strategic Plan 2008–2012. U. S. Fish and Wildlife Service, Anchorage, Alaska, USA, and Canadian Wildlife Service, Sackville, New Brunswick, Canada.

Silverman, E. D., J. B. Leirness, D. T. Saalfeld, M. D. Koneff, and K. D. Richkus. 2012. Atlantic coast wintering sea duck survey, 2008-2011. U. S. Fish and Wildlife Service, Laurel, MD, USA.

Silverman, E. D., D. T. Saalfeld, J. B. Leirness, and M. D. Koneff. 2013. Wintering Sea Duck Distribution Along the Atlantic Coast of the United States. Journal of Fish and Wildlife Management 4: 178-198.

Slattery, S., and B. Clark. 2008. Apparent annual survival of lesser scaup and white-winged scoter females from the Canadian western boreal forest. Abstract of presentation at the 3rd North American Sea Duck Conference, Quebec City, QU: http://seaduckjv.org/pdf/3rd_north_am_sea_duck_conference_quebec_program_and_abstracts.pdf

Speirs-Bridge, A., F. Fidler, M. McBride, L. Flander, G. Cumming, and M. Burgman. 2010. Reducing overconfidence in the interval judgments of experts. Risk Analysis 30(3):512-523.

Stubben, C. J., and B. G. Milligan. 2007. Estimating and analyzing demographic models using the popbio package in R. Journal of Statistical Software 22.

Tobias, T. N., and J. J. Kay. 1994. The bush harvest in Pinehouse, Saskatchewan, Canada. Arctic 47:207-221.

Traylor, J. J., and R. T. Alisauskas. 2006. Effects of intrinsic and extrinsic factors on survival of white-winged scoter ducklings. Auk 123:67–81.

Traylor J. J., R. T. Alisauskas, and F. P. Kehoe. 2004. Nesting ecology of white-winged scoters (*Melanitta fusca deglandi*) at Redberry Lake, Saskatchewan. Auk 121:950–962.

U. S. Fish and Wildlife Service. 2013. Final Supplemental Environmental Impact Statement: Issuance of annual regulations permitting the hunting of migratory birds. U. S. Department of the Interior, Washington, D.C., USA.

Vermeer, K., and N. Bourne. 1984. The White-winged Scoter diet in British Columbia waters: resource partitioning with other scoters. Pages 30–38 *in* D. N. Nettleship, G. A. Sanger, and P. F. Springer, editors. Marine Birds: Their Feeding Ecology and Commercial Fisheries Relationships; Proceedings of the Pacific Seabird Group Symposium, Seattle, Washington. Canadian Wildlife Service, Dartmouth, Nova Scotia.

Wade, P. R. 1998. Calculating limits to the allowable human-caused mortality of cetaceans and pinnipeds. Marine Mammal Science 14:1-37.

Williams, C. K. 2013. Accounting for wildlife life-history strategies when modeling stochastic density-dependent populations: a review. Journal of Wildlife Management 77:4-11.

Table 1. Median and 95% credible intervals of final probability distributions for parameters used in the harvest potential assessment of seven sea duck populations, based on available empirical data and expert elicitation.

Parameter[A]	COEI[B]	EBLSC[B]	ESUSC[B]	WBLSC[B]	WSUSC[B]	WWSC[B]	LTDU[B]
*p.first.br*2	0.21 (0.13,0.30)	0.24 (0.13,0.43)	0.24 (0.13,0.43)	0.25 (0.17,0.34)	0.22 (0.14,0.33)	0.28 (0.12,0.48)	0.28 (0.11,0.43)
*p.first.br*3	0.75 (0.60,1.00)						
bp.ad	0.92 (0.60,1.00)	0.92 (0.73,0.99)	0.93 (0.73,0.99)	0.91 (0.67,0.99)	0.85 (0.65,0.99)	0.94 (0.77,0.99)	0.88 (0.63,0.99)
cs	3.96 (2.93,5.23)	8.26 (5.83,10.77)	7.68 (5.91,9.73)	8.08 (5.79,9.91)	7.48 (3.63,9.93)	8.84 (7.30,10.37)	7.13 (5.41,9.04)
ns	0.68 (0.40,0.88)	0.63 (0.36,0.93)	0.55 (0.30,0.89)	0.48 (0.18,0.93)	0.66 (0.34,0.91)	0.24 (0.11,0.44)	0.46 (0.18,0.70)
hs	0.86 (0.56,0.95)	0.92 (0.60,1.00)	0.94 (0.58,1.00)	0.97 (0.89,1.00)	0.97 (0.49,1.00)	0.85 (0.81,0.88)	0.73 (0.19,0.94)
ds	0.15 (0.00,0.53)	0.45 (0.17,0.75)	0.42 (0.18,0.65)	0.34 (0.20,0.73)	0.40 (0.21,0.62)	0.36 (0.27,0.45)	0.24 (0.07,0.49)
wings	0.58 (0.21,1.55)	1.41 (1.15,1.72)	1.45 (0.94,2.25)	1.14 (0.63,2.06)	0.94 (0.64,1.41)	2.13 (1.74,2.60)	1.24 (1.02,1.52)
DV	2.41 (1.27,5.23)	2.50 (1.55,3.43)	2.50 (1.55,3.43)	2.25 (1.28,3.24)	2.25 (1.29,3.21)	2.50 (1.55,3.43)	2.74 (1.12,4.81)
p1	0.65 (0.40,0.85)	0.67 (0.50,0.80)	0.66 (0.50,0.80)	0.66 (0.50,0.80)	0.58 (0.29,0.81)	0.67 (0.58,0.75)	0.63 (0.45,0.78)
p2	0.89 (0.79,0.96)	0.82 (0.71,0.95)	0.82 (0.71,0.95)	0.82 (0.70,0.95)	0.88 (0.65,0.97)	0.75 (0.66,0.83)	0.71 (0.47,0.85)
p.ad	0.90 (0.78,0.96)	0.88 (0.79,0.95)	0.88 (0.79,0.95)	0.88 (0.79,0.95)	0.88 (0.65,0.97)	0.84 (0.75,0.90)	0.81 (0.58,0.91)
N^{FF}	NA	468218 (230861,721318)	398208 (69798,743716)	220022 (185367, 254971)	413687 (211809, 888205)	549322 (366132,734061)	794376 (477681,1153481)
H.obs	0.018 (0.016,0.021)	20394 (13593,27399)	26594 (19792,33598)	12034 (6770,17214)	6688 (3885,11493)	17072 (9045,25340)	30199 (23509,37089)
crip	0.30 (0.18,0.43)	0.30 (0.19,0.43)	0.30 (0.19,0.43)	0.30 (0.18,0.43)	0.30 (0.18,0.43)	0.30 (0.18,0.43)	0.30 (0.19,0.43)

[A] *p.first.br*2 (proportion of hens first breeding at age 2), *p.first.br*3 (proportion of hens first breeding at age 3), *bp.ad* (breeding propensity adult hens), *cs* (clutch size), *ns* (nest success), *hs* (hatching success), *ds* (duckling survival), *wings* (ratio of juvenile to adult hen wings submitted and collected in the US and Canadian national harvest survey), *DV* (differential vulnerability; vulnerability of juvenile to adult hens to harvest), *p*1 (juvenile, first-year survival), *p*2 (sub-adult, second year survival), *p.ad* (adult survival), N^{FF} (fall flight population size; post-birth pulse population size), *H.obs* (observed total sport and subsistence harvest unadjusted for crippling loss; for COEI this is observed harvest rate unadjusted for crippling loss), *crip* (crippling loss or unretrieved harvest).

[B] COEI (American common eider), EBLSC (eastern black scoter), ESUSC (eastern surf scoter), WBLSC (western black scoter), WSUSC (western surf scoter), WWSC (white-winged scoter), LTDU (long-tailed duck).

Table 2. Median and 95% credible intervals (in parentheses) of simulation-derived probability distributions for $r_{\max}$, allowable total harvest (or harvest rate for eider), observed total harvest (harvest rate for eider), and percent of simulations where observed harvest was ≤ allowable harvest for seven populations of sea ducks.

Population[A]	$r_{\max}$ (95% CI)	Allowable harvest (95% CI)[B]	Observed harvest (95% CI)[B]	Percent Observed < Allowable
COEI	-0.0018 (-0.1503; 0.1488)	-0.0009 (-0.0812; 0.0692)	0.0256 (0.0208; 0.0327)	20%
EBLSC	0.1390 (0.0048; 0.4487)	29,940 (807; 93,753)	29,118 (18,898; 41,641)	52%
WBLSC	0.1043 (-0.0502; 0.4052)	10,854 (-11,058; 37,219)	17,137 (9,418; 25,767)	30%
ESUSC	0.1265 (-0.0260; 0.3418)	23,149 (-9,308; 78,894)	37,947 (27,237; 51,733)	25%
WSUSC	0.0773 (-0.1344; 0.3137)	14,354 (-61,985; 82,110)	9,560 (5,399; 16,939)	59%
WWSC	0.0509 (-0.1217; 0.2335)	13,054 (-68,824; 61,072)	24,399 (12,599; 37,647)	36%
LTDU	-0.0669 (-0.2882; 0.1215)	-48,966 (-202,663; 60,561)	43,044 (32,151; 57,589)	5%

[A] COEI (American common eider), EBLSC (eastern black scoter), ESUSC (eastern surf scoter), WBLSC (western black scoter), WSUSC (western surf scoter), WWSC (white-winged scoter), LTDU (long-tailed duck).

[B] Total subsistence and sport harvest adjusted for crippling loss measured in total birds for scoters and long-tailed duck and harvest rate for common eider.

Table 3. Sensitivity of the computed difference between allowable and observed harvest to individual demographic parameters as measured by the slope of linear relationships between the computed harvest difference and the demographic parameters, where greater absolute slopes indicated higher sensitivity. Slope is affected by both the inherent sensitivity of growth rate and, hence, allowable harvest, and the relative proportional uncertainty described for a parameter.

COEI^{A}					EBLSC^{A}					ESUSC^{A}				
Parameter^{B}	Type	Slope	abs(slope)	U^{C}	Parameter^{B}	Type	Slope	abs(slope)	U^{C}	Parameter^{B}	Type	Slope	abs(slope)	U^{C}
p.ad	base	0.023	0.023	15	*p.ad*	base	11100	11100	13	*wings*	base	15500	15500	22
ds	base	0.018	0.018	39	N^{FF}	base	10900	10900	24	N^{FF}	base	12800	12800	38
wings	base	0.016	0.016	54	*ds*	base	9100	9100	31	*p.ad*	base	9210	9210	13
hs	base	0.011	0.011	31	*wings*	base	8140	8140	10	*ns*	base	8240	8240	30
cs	base	0.01	0.01	14	*p.first.br*2	base	-7100	7100	17	*DV*	base	-6920	6920	19
*p.first.br*2	base	-0.008	0.008	10	*DV*	base	-7100	7100	19	*ds*	base	5830	5830	22
*p.first.br*3	base	-0.007	0.007	36	*ns*	base	7010	7010	31	*cs*	base	5320	5320	12
bp.ad	base	0.005	0.005	40	*cs*	base	6310	6310	14	*p*2	base	5100	5100	19
ns	base	0.005	0.005	25	*p*1	base	5610	5610	16	*H.obs*	base	-5050	5050	13
DV	base	-0.003	0.003	38	*H.obs*	base	-5460	5460	17	*p.first.br*2	base	-4100	4100	17
*p*2	base	0.003	0.003	14	*hs*	base	5350	5350	43	*crip*	base	-3600	3600	14
h.obs	base	-0.002	0.002	1	*p*2	base	4470	4470	18	*hs*	base	2970	2970	53
crip	base	-0.002	0.002	14	*crip*	base	-2370	2370	14	*p*1	base	2800	2800	16
p1	base	0	0	28	*bp.ad*	base	-274	274	23	*bp.ad*	base	729	729	27
N^{FF}	base	NA	NA	NA	*p.first.br*3	base	NA	NA	NA	*p.first.br*3	base	NA	NA	NA
r_{max}	computed	0.037	0.037		r_{max}	computed	22400	22400		r_{max}	computed	18700	18700	
$b.ad_{model}$	computed	0.034	0.034	90	$b.ad_{model}$	computed	21900	21900	51	$b.ad_{model}$	computed	17800	17800	47
$b.ad_{total}$	computed	0.027	0.027	69	$b.ad_{total}$	computed	20100	20100	53	$b.ad_{total}$	computed	16400	16400	42
$b2_{model}$	computed	0.02	0.02	91	*Adj.Age.Rat*	computed	9990	9990	25	*Adj.Age.Rat*	computed	15800	15800	32
$b3_{model}$	computed	0.021	0.021	86	$b2_{model}$	computed	6680	6680	57	$b2_{model}$	computed	7360	7360	48
Adj.Age.Rat	computed	0.018	0.018	59	$b3_{model}$	computed	NA	NA	NA	$b3_{model}$	computed	NA	NA	NA

Table 3. Continued. Sensitivity of the computed difference between allowable and observed harvest to individual demographic parameters as measured by the slope of linear relationships between the computed harvest difference and the demographic parameters, where greater absolute slopes indicated higher sensitivity. Slope is affected by both the inherent sensitivity of growth rate and, hence, allowable harvest, and the relative proportional uncertainty described for a parameter.

WBLSC[A]					WSUSC[A]					WWSC[A]				
Parameter[B]	Type	Slope	abs(slope)	U[C]	Parameter[B]	Type	Slope	abs(slope)	U[C]	Parameter[B]	Type	Slope	abs(slope)	U[C]
ns	base	5960	5960	44	*p.ad*	base	20400	20400	25	*p.ad*	base	13600	13600	11
ds	base	5690	5690	31	*p*1	base	9380	9380	26	*DV*	base	-9670	9670	19
wings	base	4820	4820	30	N^{FF}	base	8730	8730	39	*ns*	base	8880	8880	20
p.ad	base	4600	4600	13	*DV*	base	-7780	7780	22	*hs*	base	7750	7750	5
H.obs	base	-3850	3850	22	*cs*	base	7420	7420	20	*H.obs*	base	-5110	5110	24
DV	base	-3590	3590	22	*ds*	base	7170	7170	20	*wings*	base	4670	4670	10
*p.first.br*2	base	2600	2600	10	*wings*	base	6850	6850	20	*cs*	base	4210	4210	9
cs	base	2150	2150	11	*ns*	base	6100	6100	32	*ds*	base	3630	3630	10
*p*2	base	1810	1810	19	*p*2	base	5070	5070	25	*p*2	base	3440	3440	10
*p*1	base	1510	1510	16	*hs*	base	4940	4940	79	*p*1	base	2940	2940	9
crip	base	-1350	1350	14	*bp.ad*	base	3510	3510	26	*crip*	base	-2410	2410	14
N^{FF}	base	872	872	8	*p.first.br*2	base	2890	2890	12	*bp.ad*	base	1760	1760	24
bp.ad	base	786	786	30	*H.obs*	base	-2750	2750	28	N^{FF}	base	1350	1350	17
hs	base	409	409	17	*crip*	base	-1540	1540	14	*p.first.br*2	base	898	898	21
$p.first.br_3$	base	NA	NA	NA	*p.first.br*3	base	NA	NA	NA	*p.first.br*3	base	NA	NA	NA
$b.ad_{model}$	computed	12600	12600	72	r_{max}	computed	31200	31200		r_{max}	computed	32900	32900	
r_{max}	computed	11300	11300		$b.ad_{model}$	computed	23800	23800	49	$b.ad_{total}$	computed	28900	28900	57
$b.ad_{total}$	computed	9830	9830	64	$b.ad_{total}$	computed	20300	20300	51	$b.ad_{model}$	computed	19200	19200	40
$b2_{model}$	computed	7480	7480	75	$b2_{model}$	computed	11900	11900	54	*Adj.Age.Rat*	computed	10900	10900	25
Adj.Age.Rat	computed	6010	6010	40	*Adj.Age.Rat*	computed	10500	10500	32	$b2_{model}$	computed	8170	8170	54
$b3_{model}$	computed	NA	NA	NA	$b3_{model}$	computed	NA	NA	NA	$b3_{model}$	computed	NA	NA	NA

Table 3. Continued. Sensitivity of the comparison of allowable to observed harvest (steeper slopes represent higher sensitivity) to base and computed parameters with each parameter's relative uncertainty (CV).

		LTDU[A]		
Parameter[B]	Type	Slope	abs(slope)	U[C]
p.ad	base	52500	52500	21
N^{FF}	base	26100	26100	21
ns	base	12700	12700	28
*p.first.br*2	base	12400	12400	18
*p*2	base	12000	12000	20
*p*1	base	10500	10500	17
H.obs	base	-4720	4720	11
crip	base	-4360	4360	14
hs	base	3640	3640	66
bp.ad	base	3460	3460	30
DV	base	-2870	2870	31
ds	base	2740	2740	26
wings	base	1670	1670	10
cs	base	-878	878	12
$b.ad_{\text{model}}$	computed	6050	6050	83
$r_{\max}$	computed	62800	62800	
$b.ad_{\text{total}}$	computed	32800	32800	65
$b2_{\text{model}}$	computed	11400	11400	91
Adj.Age.Rat	computed	1070	1070	39
$b3_{\text{model}}$	computed	NA	NA	NA

[A] COEI (American common eider), EBLSC (eastern black scoter), ESUSC (eastern surf scoter), WBLSC (western black scoter), WSUSC (western surf scoter), WWSC (white-winged scoter), LTDU (long-tailed duck).

[B] Component parameters: *p.first.br*2 (proportion of hens first breeding at age 2), *p.first.br*3 (proportion of hens first breeding at age 3), *bp.ad* (breeding propensity adult hens), *cs* (clutch size), *ns* (nest success), *hs* (hatching success), *ds* (duckling survival), *wings* (ratio of juvenile to adult female wings submitted collected in the US and Canadian national harvest survey), *DV* (differential vulnerability), *p*1 (juvenile, first-year survival), *p*2 (sub-adult, second-year survival), *p.ad* (adult survival), N^{FF} (Fall flight population size; post-birth pulse population size), *H.obs* (observed total sport and subsistence harvest unadjusted for crippling loss; for COEI this is observed harvest rate unadjusted for crippling loss), *crip* (crippling loss or unretrieved harvest). Computed parameters: $r_{\max}$, $b.ad_{\text{model}}$ (adult fecundity modeled from component rates), $b2_{\text{model}}$ (year 2 fecundity modeled from component rates), $b3_{\text{model}}$ (year 3 fecundity modeled from component rates), *Adj.Age.Rat* (fecundity estimated from female harvest age ratios adjusted for differential vulnerability), $b.ad_{\text{total}}$ (composite fecundity derived from the probability distributions of $b.ad_{\text{model}}$ and *Adj.Age.Rat*).

[C] Relative proportional uncertainty defined as the coefficient of variation for non-binomial parameters or concentration for binomial parameters (i.e., rates constrained to values between 0 and 1).

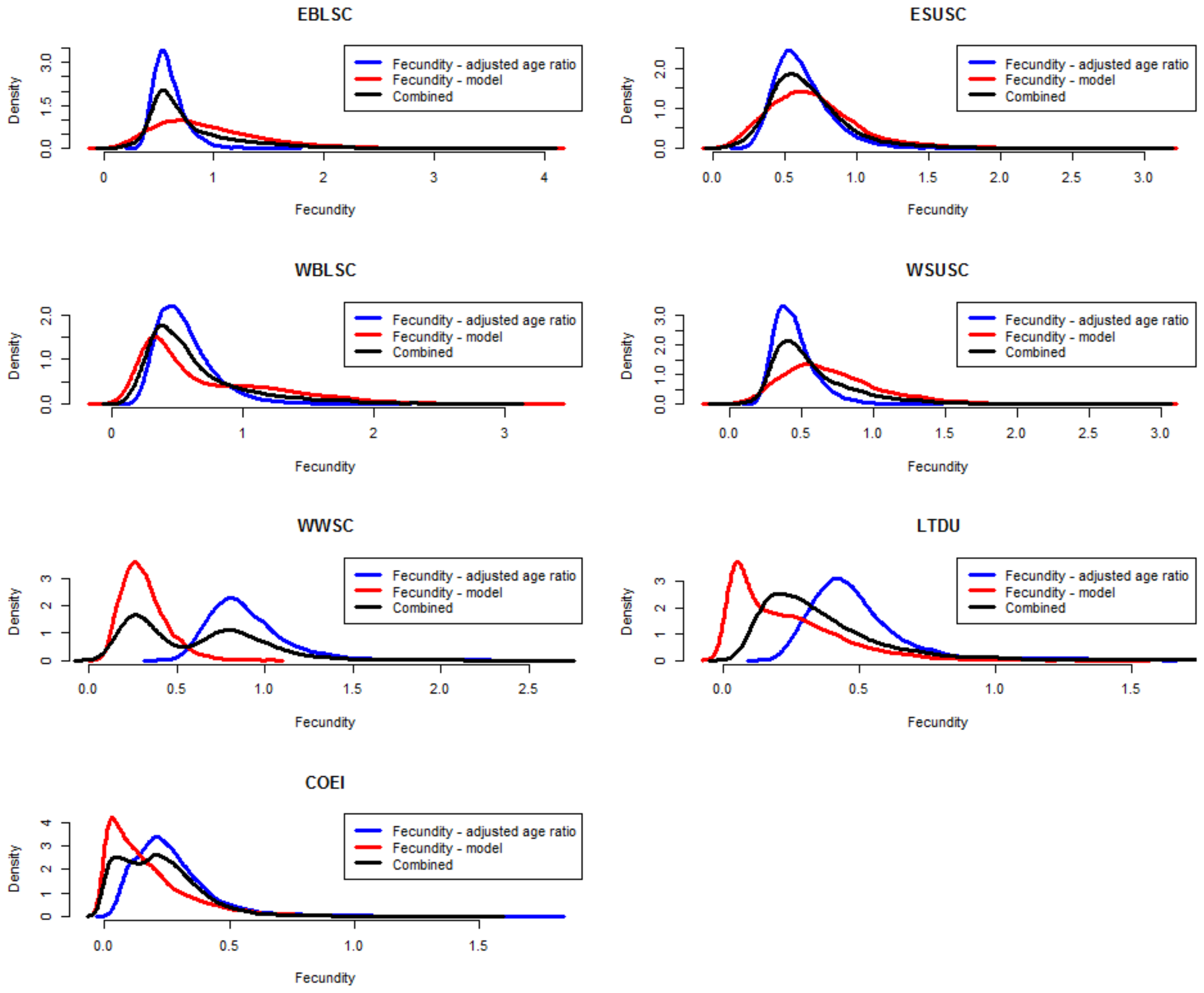

Figure 2. Probability distributions depicting uncertainty in two alternative methods of estimating fecundity for seven populations of North American sea ducks as well as a combined probability distribution that weights both methods equally (EBLSC = eastern black scoter, WBLSC = western black scoter, ESUSC = eastern surf scoter, WSUSC = western surf scoter, WWSC = white-winged scoter, LTDU = long-tailed duck, and COEI = American subspecies of common eider).

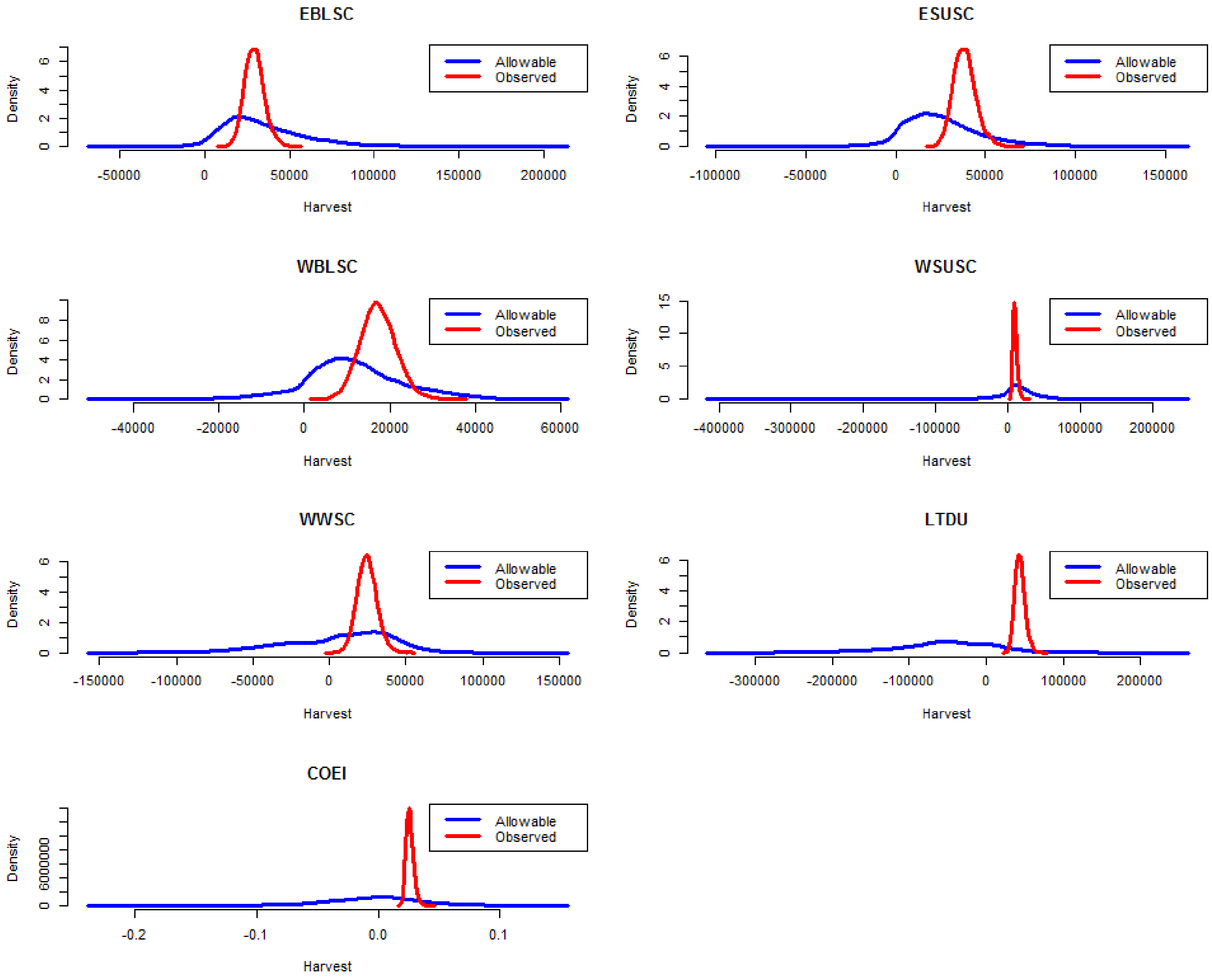

Figure 3. Distributions of allowable and observed harvest (or harvest rate, for common eiders) for seven populations of North American sea ducks based on Monte Carlo simulation (EBLSC = eastern black scoter, WBLSC = western black scoter, ESUSC = eastern surf scoter, WSUSC = western surf scoter, WWSC = white-winged scoter, LTDU = long-tailed duck, and COEI = American subspecies of common eider).

Table A1. American common eider demographic parameter estimates obtained from a search of published literature and unpublished data, and from solicited expert opinion.

Parameter	**Estimate and variability**	**Scale**	**Source**
Breeding propensity at age 1	0	Eastern North America	Gilliland et al. 2009
Breeding propensity at age 2	0.22	Eastern North America	Gilliland et al. 2009
Breeding propensity at age 3	0.40	Eastern North America	Gilliland et al. 2009
Breeding propensity at age 3	0.80	Eastern North America	Expert opinion
Breeding propensity at age $\geq$4	1	Eastern North America	Gilliland et al. 2009
Clutch size	$\bar{x}$ = 4, range 1-8	Species	Goudie et al. 2000
Nest success	0.90	Eastern North America	Gilliland et al. 2009
Nest success	$\bar{x}$ = 0.75, SE = 0.075	Eastern North America	Expert opinion
Hatching success	0.90, SE = 0.09	Eastern North America	Gilliland et al. 2009
Duckling survival to fledging	$\bar{x}$ = 0.10, SE = 0.06	Local (Nunavut)	Descamps et al. 2011
2nd year survival rate	0.87	American subspecies	Krementz et al. 1996
2nd year survival rate	0.89	Maine	B. Allen, unpublished data
Adult ($\geq$3rd year) survival rate	0.87	American subspecies	Krementz et al. 1996
Adult ($\geq$3rd year) survival rate	0.89	Maine	B. Allen, unpublished data
Adult ($\geq$3rd year) survival rate	0.91, SD = 0.05	Species	Johnson et al. 2012
Age at last breeding	23	North America	BBL longevity records
Harvest rate for adult females	0.027, SE = 0.002	Maine (molting)	Allen et al. (in prep.)
	0.018, 95%CI (0.016-0.021)	Maine-So. Labrador (molting+nesting)	Zimmerman (unpub. data)

Table A2. Eastern black scoter demographic parameter estimates obtained from a search of published literature and unpublished data, and from solicited expert opinion.

Parameter	**Estimate and variability**	**Scale**	**Source**
Age at first breeding	2nd or 3rd year	Species	Bordage and Savard 1995
Adult breeding propensity	Unknown, assume 1	Eastern North America	Bordage and Savard 1995
Clutch size	$\bar{x}$ = 7.7, SD = 1.7	Local (Alaska)	Bordage and Savard 1995
Clutch size	$\bar{x}$ = 8.7, SD = 1.37	Local (Iceland)	Bengtson 1971
Nest success	$\bar{x}$ = 0.5, SD = 0.05	Eastern North America	Expert opinion
Hatching success	$\bar{x}$ = 0.95, SD = 0.09	Local (Iceland)	Bengtson 1972
Duckling survival to fledging	$\bar{x}$ = 0.55, SD = 0.05	Local (Iceland)	Bengtson 1972
Adult survival rate	0.85, SE = 0.08	Eastern North America	Gilliland et al. 2013; expert opinion
Adult survival rate	0.89, SD = 0.05	Species	Johnson et al. 2012
Age at last breeding	16	Species	EURING longevity records
Winter population count	$\bar{x}$ = 211,300, CV = 11%	Atlantic coast and Great Lakes	Silverman et al. 2012
Winter population detection rate	0.416, SD = 0.011	Winter aerial surveys	Evenson et al. 2013; J. Leirness, unpublished data
Retrieved fall and winter harvest, U.S.	$\bar{x}$ =12,678, CV = 16%	Atlantic and Mississippi Flyways	USFWS annual harvest reports
Retrieved fall and winter harvest, Canada	$\bar{x}$ = 2,408, CV = 32%	Atlantic and Mississippi Flyways	Gendron and Smith 2014
Retrieved subsistence harvest	$\bar{x}$ = 6,280, CV = 70%	Eastern Canada	Natcher et al. 2011; C. Lepage, personal communication; Rothe et al. 2015
Crippling loss	0.3, CV = 7%	Species, North America	Rothe et al. 2015

Table A3. Western black scoter demographic parameter estimates obtained from a search of published literature and unpublished data, and from solicited expert opinion.

Parameter	**Estimate and variability**	**Scale**	**Source**
Age at first breeding	2nd or 3rd year	Species	Bordage and Savard 1995
Adult breeding propensity	Unknown, assume 1	Western North America	Bordage and Savard 1995
Clutch size	$\bar{x}$ = 8.7, SD = 1.37	Local (Iceland)	Bengtson 1971
Clutch size	$\bar{x}$ = 7.5, 95% CL 7.2-7.6	Local (Alaska)	Schamber et al. 2010
Nest success	$\bar{x}$ = 0.83, SD = 0.083	Local (Iceland)	Bengtson 1972
Nest success	0.02-0.37	Local (Alaska)	Schamber et al. 2010
Hatching success	$\bar{x}$ = 0.95, SD = 0.09	Local (Iceland)	Bengtson 1972
Duckling survival rate (to fledging)	$\bar{x}$ = 0.55, SD = 0.05	Local (Iceland)	Bengtson 1972
Duckling survival rate (to 30 days)	0.09-0.35	Local (Alaska)	Schamber et al. 2010
Adult survival rate	0.78, 95% CL 0.72-0.84	Local (Iceland)	Fox et al. 2003
Adult survival rate	0.89, SD = 0.05	Species	Johnson et al. 2012
Age at last breeding	16	Species	EURING longevity records
Winter population count	$\bar{x}$ = 100,216	Pacific coast	D. Kraege, unpublished data
Winter population detection rate	0.416, SD = 0.011	Winter aerial surveys	Evenson et al. 2013; J. Leirness, unpublished data
Retrieved fall and winter harvest	$\bar{x}$ = 550, CV = 47%	Pacific Flyway	USFWS annual harvest reports; Gendron and Smith 2014
Retrieved subsistence harvest	$\bar{x}$ = 9,606, CV = 70%	Alaska	Rothe et al. 2015
Crippling loss	0.3, CV = 7%	Species, North America	Rothe et al. 2015

Table A4. Eastern surf scoter demographic parameter estimates obtained from a search of published literature and unpublished data, and from solicited expert opinion.

Parameter	**Estimate and variability**	**Scale**	**Source**
Age at first breeding	2nd or 3rd year	Species	Savard et al. 1998
Adult breeding propensity	Unknown, assume <1	Eastern North America	Savard et al. 1998
Clutch size	$\bar{x}$ = 7.6, SE = 0.02	Local (Quebec)	Morrier et al. 1997
Nest success	$\bar{x}$ = 0.55, SE = 0.103	Local (Quebec)	Morrier et al. 1997
Hatching success	$\bar{x}$ = 0.97	Local (Quebec)	Morrier et al. 1997
Duckling survival to fledging	0.424, SE = 0.024	Local (Quebec)	Morrier et al. 1997
Adult survival rate	0.91, SE = 0.053	Eastern North America	E. Reed, updated analysis of Gilliland et al. 2011 data
Adult survival rate	0.89, SD = 0.05	Species	Johnson et al. 2012
Age at last breeding	20	Species	Longevity of similar spp. (BLSC, WWSC; Euring)
Winter population count	$\bar{x}$ = 150,826, CV = 11%	Atlantic coast and Great Lakes	Silverman et al. 2012
Winter population detection rate	0.416, SD = 0.011	Winter aerial surveys	Evenson et al. 2013; J. Leirness, unpublished data
Retrieved fall and winter harvest, U.S.	$\bar{x}$ = 25,404, CV = 10%	Atlantic and Mississippi Flyways	USFWS annual harvest reports
Retrieved fall and winter harvest, Canada	$\bar{x}$ = 3,114, CV = 29%	Atlantic and Mississippi Flyways	Gendron and Smith 2014
Retrieved subsistence harvest	$\bar{x}$ = 3,562, CV = 70%	Eastern Canada	Natcher et al. 2011; C. Lepage, personal communication; Rothe et al. 2015
Crippling loss	0.3, CV = 7%	Species, North America	Rothe et al. 2015

Table A5. Western surf scoter demographic parameter estimates obtained from a search of published literature and unpublished data, and from solicited expert opinion.

Parameter	**Estimate and variability**	**Scale**	**Source**
Age at first breeding	2^{nd} or 3^{rd} year	Species	Savard et al. 1998
Adult breeding propensity	Unknown, assume <1	Eastern North America	Savard et al. 1998
Clutch size	$\bar{x}$ = 7.6, SE = 0.02	Local (Quebec)	Morrier et al. 1997
Nest success	$\bar{x}$ = 0.55, SE = 0.103	Local (Quebec)	Morrier et al. 1997
Hatching success	$\bar{x}$ = 0.97	Local (Quebec)	Morrier et al. 1997
Duckling survival to fledging	0.424, SE = 0.024	Local (Quebec)	Morrier et al. 1997
Adult survival rate	0.91, SE = 0.073	Eastern North America	D. Kraege, unpublished data
Adult survival rate	0.89, SD = 0.05	Species	Johnson et al. 2012
Age at last breeding	20	Species	Longevity of similar spp. (BLSC, WWSC; Euring)
Winter population count	$\bar{x}$ = 222,983	Pacific coast	D. Kraege, unpublished data
Winter population detection rate	0.416, SD = 0.011	Winter aerial surveys	Evenson et al. 2013; J. Leirness, unpublished data
Retrieved fall and winter harvest	$\bar{x}$ = 4,798, CV = 37%	Pacific Flyway	USFWS annual harvest reports; Gendron and Smith 2014
Retrieved subsistence harvest	$\bar{x}$ = 2,809, CV = 70%	Alaska	Rothe et al. 2015
Crippling loss	0.3, CV = 7%	Species, North America	Rothe et al. 2015

Table A6. White-winged scoter demographic parameter estimates obtained from a search of published literature and unpublished data, and from solicited expert opinion.

Parameter	**Estimate and variability**	**Scale**	**Source**
Age at first breeding	2nd or 3rd year	Species	Brown and Frederickson 1997
Adult breeding propensity	0.72	Local (Alaska)	Safine 2005
Clutch size [1]	$\bar{x}$ = 8.8, 95% CL 8.6-9.1	Local (Saskatchewan)	Traylor et al. 2004
Clutch size [1]	$\bar{x}$ = 8.1, SE = 0.2	Local (Alaska)	Safine 2005
Nest success [1]	$\bar{x}$ = 0.35, 95% CL 0.27-0.43	Local (Saskatchewan)	Traylor et al. 2004
Nest success [1]	$\bar{x}$ = 0.25, 95% CL 0.13-0.49	Local (Alaska)	Safine 2005
Hatching success [1]	$\bar{x}$ = 0.85, 95% CL 0.80-0.88	Local (Saskatchewan)	Traylor et al. 2004
Hatching success [1]	$\bar{x}$ = 0.92, SE = 0.02	Local (Alaska)	Safine 2005
Duckling survival rate (to fledging)	0.18-0.40, $\bar{x}$ = 0.28	Local (Saskatchewan, 1977-80)	Brown and Frederickson 1989
Duckling survival rate (to 30 days)	$\bar{x}$ = 0.0044	Local (Saskatchewan, 2000-01)	Traylor and Alisauskas 2006
Duckling survival rate (to 30 days)	0.08, 95% CL 0.03-0.15	Local (Alaska, 2003)	Safine 2005
Duckling survival rate (to 30 days)	0.64, 95% CL 0.51-0.75	Local (Alaska, 2004)	Safine 2005
Adult survival rate [1]	0.69, SE = 0.04	Local (Northwest Territories)	Slattery and Clark 2008
Adult survival rate [1]	0.84, 95% CL 0.77-0.91	Local (Saskatchewan)	Alisauskas et al. 2004
Adult survival rate	0.90, SD = 0.05	Species	Johnson et al. 2012
Age at last breeding	21	Species	EURING longevity records
Winter population count	$\bar{x}$ = 161,890 (Pacific coast) + 58,600, CV = 19% (Atlantic coast)	North America	Silverman et al. 2012; D. Kraege, unpublished data
Winter population detection rate	0.416, SD = 0.011	Winter aerial surveys	Evenson et al. 2013; J. Leirness, unpublished data
Retrieved fall and winter harvest, U.S.	$\bar{x}$ = 8,379, CV = 22%	United States	USFWS annual harvest reports
Retrieved fall and winter harvest, Canada	$\bar{x}$ = 1,891, CV = 32%	Canada	Gendron and Smith 2014
Retrieved subsistence harvest	$\bar{x}$ = 1,871, CV = 70%	North America	Natcher et al. 2011; C. Lepage, personal communication; Rothe et al. 2015
Crippling loss	0.3, CV = 7%	Species, North America	Rothe et al. 2015

---

[1] Additional estimates from older studies are given and referenced in Brown and Frederickson (1997).

Table A7. Long-tailed duck demographic parameter estimates obtained from a search of published literature and unpublished data, and from solicited expert opinion.

Parameter	**Estimate and variability**	**Scale**	**Source**
Age at first breeding	2nd year	Species	Robertson and Savard 2002
Adult breeding propensity	Unknown, assume <1	North America	Robertson and Savard 2002
Clutch size[1]	$\bar{x}$ = 7.1, 95% CL 6.5-7.7	Local (Alaska)	Schamber et al. 2009
Nest success [1]	$\bar{x}$ = 0.30, 95% CL 0.24-0.36	Local (Alaska)	Schamber et al. 2009
Nest success[1]	0.14-0.90, $\bar{x}$ = 0.30	Local (Nunavut)	Kellett and Alisauskas 2014
Hatching success	$\bar{x}$ = 0.804	Local (Manitoba)	Alison 1975
Duckling survival rate (to 30 days)	0-0.25, $\bar{x}$ = 0.10	Local (Alaska)	Schamber et al. 2009
Adult survival rate	0.74, 95% CL 0.57-0.86	Local (Alaska)	Schamber et al. 2009
Adult survival rate	0.85, 95% CL 0.76-0.92	Local (Nunavut)	Kellett and Alisauskas 2014
Adult survival rate	0.88, SD = 0.05	Species	Johnson et al. 2012
Age at last breeding	17	North America	BBL longevity records
Winter population count	$\bar{x}$ = 174,029 (Pacific coast) + 236,552, CV = 11% (Atlantic coast); 53,770 (Great Lakes)	North America	Silverman et al. 2012; D. Kraege, unpublished data; M. Schummer, unpublished data
Winter population detection rate	0.814, SD = 0.034	Winter aerial surveys	J. Leirness, unpublished data
Retrieved fall and winter harvest, U.S.	$\bar{x}$ = 26,383, CV = 10%	United States	USFWS annual harvest reports
Retrieved fall and winter harvest, Canada	$\bar{x}$ = 2,189, CV = 32%	Canada	Gendron and Smith 2014
Retrieved subsistence harvest	$\bar{x}$ = 9,841, CV = 70%	North America	Natcher et al. 2011; C. Lepage, personal communication; Rothe et al. 2015
Crippling loss	0.3, CV = 7%	Species, North America	Rothe et al. 2015

[1] Additional estimates from older studies are given and referenced in Robertson and Savard (2002).

Table B1. Expert elicitation and literature estimates of sea duck demographic rates, differential vulnerability to harvest, and fall population size (with associated uncertainty) used to construct sample distributions for simulation of harvest potential.

COEI (American common eider)

	Expert 1					Expert 2				Expert 3				Expert 4					Data from Literature	
Parameter	Low	Est.	High	Prob	Freq	Low	Est.	High	Prob	Low	Est.	High	Prob	Low	Est.	High	Prob	Freq	Estimate	Distributions
2nd year breeding propensity		0.20									0.22									
3rd year breeding propensity		0.50									1.00									
Adult breeding propensity	0.85	0.90	0.95	60		0.90	0.95	0.99	70	0.70	1.00	1.00	90	0.60	0.80	0.90	90			
Clutch size	3.50	4.00	4.50	90		3.00	4.00	5.00	80	3.00	4.00	5.00	75							
Nest success	0.65	0.70	0.75	70		0.50	0.60	0.80	60	0.50	0.75	0.80	75	0.40	0.68	0.85	95			
Hatch success	0.85	0.90	0.95	80		0.75	0.85	0.90	70	0.85	0.90	0.95	75						0.69 (SD = 0.09)	Beta(16.14,7.25)
Duckling survival: background	0.10	0.20	0.30	55	90%	0.08	0.20	0.30	60	0.05	0.10	0.50	75	0.027	0.032	0.164	80	75%	0.10 (SD = 0.06)	Beta(2.40,21.60)
Duckling survival: boom	0.20	0.40	0.60	55	10%									0.172	0.420	0.420	80	25%		
1st year survival	0.40	0.50	0.60	50		0.65	0.75	0.60	50	0.50	0.71	0.80	75							
2nd year survival	0.85	0.90	0.95	70		0.85	0.90	0.95	70	0.80	0.87	0.95	90							
Adult (and 3rd year) survival	0.85	0.90	0.95	90		0.85	0.90	0.95	90	0.85	0.90	0.95	90	0.80	0.90	0.95	95			
Differential vulnerability	1.50	2.00	2.50	50										2.00	3.50	5.00	90			

EBLSC (Eastern black scoter)

	Expert 1				Expert 2				Expert 3				Expert 4				Data from Literature	
Parameter	Low	Est.	High	Prob	Low	Est.	High	Prob	Low	Est.	High	Prob	Low	Est.	High	Prob	Estimate	Distributions
2nd year breeding propensity	0.15	0.25	0.45	80		0.25				0.2								
Adult breeding propensity	0.80	0.90	1.00	90	0.75	0.90	1.00	75	0.85	0.90	0.95	70						
Clutch size					6.00	7.50	10.00	80	7.50	8.50	9.00	90					8.60 (SD =1.40)	Normal(8.60,1.40)
Nest success					0.40	0.83	0.90	75	0.40	0.60	0.70	60						
Hatch success					0.90	0.95	1.00	90	0.60	0.85	0.90	60						
Duckling survival					0.20	0.55	0.70	75	0.20	0.40	0.60	50					0.47 (SD = 0.16)	Beta(4.43,4.89)
Age at last breeding	16	20	22	90	12	16	20	99	9	13	16	70						
1st year survival	0.64	0.68	0.72	50	0.50	0.56	0.70	80	0.60	0.70	0.80	50						
2nd year survival	0.72	0.77	0.81	80	0.80	0.90	0.95	80	0.75	0.80	0.95	60						
Adult survival	0.80	0.85	0.90	90	0.80	0.90	0.95	80	0.85	0.90	0.95	60						
Fall flight (in thousands)	350	450	550	80	238	511	714	80	438	505	543	60	310	400	570	55		
Differential vulnerability	1.5	2.5	3.5	95														

Table B1 Continued. Expert elicitation and literature estimates of sea duck demographic rates, differential vulnerability to harvest, and fall population size (with associated uncertainty) used to construct sample distributions for simulation of harvest potential.

WBLSC (western black scoter)										
	Expert 1				Expert 2				Data from Literature	
Parameter	Low	Est.	High	Prob	Low	Est.	High	Prob	Estimate	Distributions
2nd year breeding propensity		0.25								
Adult breeding propensity	0.75	0.90	1.00	75						
Clutch size	6.00	7.50	10.00	80					8.10 (SD =0.10)	Normal(8.10,0.10)
Nest success	0.40	0.83	0.90	75	0.20	0.35	0.50	60		
Hatch success	0.90	0.95	1.00	90						
Duckling survival	0.20	0.55	0.70	75					0.29 (SD = 0.16)	Uniform(0.2,0.6) and Uniform(0.09,0.35)[A]
Age at last breeding	12	16	20	99						
1st year survival[B]										
2nd year survival[B]										
Adult survival[B]										
Fall flight (in thousands)					190	221	250	85		
Differential vulnerability										

[A] Two studies provided ranges only.

[B] No data available, used information for EBLSC as a surrogate.

Table B1 Continued. Expert elicitation and literature estimates of sea duck demographic rates, differential vulnerability to harvest, and fall population size (with associated uncertainty) used to construct sample distributions for simulation of harvest potential.

ESUSC (eastern surf scoter)																		
	Expert 1				Expert 2				Expert 3				Expert 4				Data from Literature	
Parameter	Low	Est.	High	Prob	Low	Est.	High	Prob	Low	Est.	High	Prob	Low	Est.	High	Prob	Estimate	Distributions
2nd year breeding propensity	0.15	0.25	0.45	80		0.20				0.25								
Adult breeding propensity	0.80	0.90	1.00	90	0.85	0.95	0.97	60	0.75	0.90	1.00	75						
Clutch size					6.00	7.00	8.00	90	6.00	7.60	10.00	80					7.70 (SD = 0.02)	Normal (7.70,0.02)
Nest success					0.30	0.50	0.60	55	0.40	0.70	0.90	75					0.56 (SD = 0.10)	Beta (12.29,9.93)
Hatch success					0.60	0.80	0.90	70	0.95	0.97	1.00	90						
Duckling survival					0.20	0.40	0.60	50	0.20	0.42	0.60	75					0.42 (SD = 0.02)	Binomial (0.42,422)
Age at last breeding	16	20	22	90	14	16	17	60	15	20	25	99						
1st year survival	0.64	0.68	0.72	50	0.60	0.70	0.80	50	0.50	0.56	0.70	80						
2nd year survival	0.72	0.77	0.81	80	0.75	0.80	0.95	60	0.80	0.90	0.95	80						
Adult survival	0.80	0.85	0.90	90	0.85	0.90	0.95	70	0.80	0.90	0.95	80						
Fall flight (in thousands)	300	350	700	80	333	381	429	80	238	359	714	50	160	285	750	66		
Differential vulnerability	1.5	2.5	3.5	95														

Table B1 Continued. Expert elicitation and literature estimates of sea duck demographic rates, differential vulnerability to harvest, and fall population size (with associated uncertainty) used to construct sample distributions for simulation of harvest potential.

WSUSC (western surf scoter)														
	Expert 1				Expert 2				Expert 3				Data from Literature	
Parameter	Low	Est.	High	Prob	Low	Est.	High	Prob	Low	Est.	High	Prob	Estimate	Distributions
2nd year breeding propensity		0.20				0.25								
Adult breeding propensity	0.70	0.80	0.90	60	0.75	0.90	1.00	75						
Clutch size	3.00	7.00	9.00	60	6.00	7.60	10.00	80	6.00	6.70	7.00	60	7.60 (SD = 0.02)	Normal (7.60,0.02)
Nest success	0.40	0.70	0.80	60	0.40	0.70	0.90	75						
Hatch success	0.50	0.90	1.00	80	0.95	0.97	1.00	90						
Duckling survival	0.30	0.40	0.50	80	0.20	0.42	0.60	75						
Age at last breeding	8	10	20	60	15	20	25	99						
1st year survival	0.25	0.48	0.72	90	0.50	0.56	0.70	80	0.42	0.62	0.83	80		
2nd year survival	0.73	0.91	0.97	90	0.80	0.90	0.95	80	0.66	0.77	0.99	80		
Adult survival	0.73	0.91	0.97	90	0.80	0.90	0.95	80	0.66	0.77	0.99	80		
Fall flight (in thousands)	223	387	550	70	238	509	714	50						
Differential vulnerability	1.50	2.25	3.00	75										

Table B1 Continued. Expert elicitation and literature estimates of sea duck demographic rates, differential vulnerability to harvest, and fall population size (with associated uncertainty) used to construct sample distributions for simulation of harvest potential.

WWSC (white-winged scoter)						
	Expert 1				Data from Literature	
Parameter	Low	Est.	High	Prob	Estimate	Distributions
2nd year breeding propensity	0.15	0.25	0.45	80		
Adult breeding propensity	0.80	0.90	1.00	90		
Clutch size					8.85 (SD =0.77)	Normal(8.85,0.77)
Nest success					0.24 (SD = 0.09)	MVNORM[A](mean=-1.15,1.72,vc matrix= $\begin{vmatrix} 0.22 & 0.05 \\ 0.05 & 0.02 \end{vmatrix}$
Hatch success					0.85 (SD = 0.02)	
Duckling survival					0.36 (SD = 0.05)	Beta(38.29,68.07)
Age at last breeding	16	20	22	90		
1st year survival	0.60	0.68	0.72	50		
2nd year survival	0.68	0.77	0.81	80		
Adult survival	0.75	0.85	0.90	90		
Fall flight (in thousands)	400	550	700	80		
Differential vulnerability	1.5	2.5	3.5	95		

[A] Mean transformed using logit link function.

Table B1 Continued. Expert elicitation and literature estimates of sea duck demographic rates, differential vulnerability to harvest, and fall population size (with associated uncertainty) used to construct sample distributions for simulation of harvest potential.

LTDU (long-tailed duck)																		
	Expert 1				Expert 2				Expert 3				Expert 4				Data from Literature	
Parameter	Low	Est.	High	Prob	Low	Est.	High	Prob	Low	Est.	High	Prob	Low	Est.	High	Prob	Estimate	Distributions
2nd year breeding propensity	0.15	0.25	0.45	80	0.15	0.25	0.35	60	0.10	0.20	0.30	70		0.25				
Adult breeding propensity	0.80	0.90	1.00	90	0.70	0.90	0.97	70	0.60	0.80	0.95	90						
Clutch size					6.00	7.50	8.00	80	6.00	7.00	8.00	70	4.30	7.30	10.30	95	7.20 (SD = 0.46)	Normal(7.20,0.46)
Nest success					0.20	0.40	0.65	75	0.20	0.40	0.70	70	0.29	0.39	0.52	95	0.59 (SD = 0.05)	Beta(55.32,38.60)
Hatch success					0.50	0.85	0.90	75	0.70	0.80	0.95	90	0.18	0.20	0.22	95		
Duckling survival					0.10	0.30	0.40	50	0.10	0.15	0.40	70						
Age at last breeding	16	20	22	90	7	9	12	50	4	7	9	70						
1st year survival	0.60	0.66	0.70	50	0.50	0.60	0.70	50	0.50	0.60	0.80	60						
2nd year survival	0.68	0.74	0.78	80	0.55	0.65	0.85	60	0.50	0.70	0.80	60						
Adult survival	0.75	0.82	0.87	90	0.60	0.75	0.90	75	0.60	0.80	0.85	70	0.76	0.85	0.92	95		
Fall flight (in thousands)	550	650	1000	80	600	750	1200	60	600	800	1000	70						
Differential vulnerability	1.5	2.5	3.5	95	1.5	2.5	5.0	60	2.0	3.0	4.0	70						

Table B2. Elicitation used to estimate harvest and crippling loss for scoters and long-tailed ducks.

Parameter: population	Low	Est	High	Prob
Crippling loss: all	0.2	0.3	0.4	0.8
Harvest: EBLSC	15,000	20, 069	26,000	0.8
Harvest: WBLSC	8,000	12,190	16,000	0.7
Harvest: ESUSC	21,000	27,065	32,000	0.8
Harvest: WSUSC	4,000	7,354	10,000	0.8
Harvest: WWSC	11,000	17,318	23,000	0.7
Harvest: LTDU	25,000	30,727	35,000	0.7

Appendix

Response to Reviewers

Reviewer #1:

This report represents an impressive body of work. It is extremely well written, and should be of great utility to waterfowl managers. The results and conclusions seem sound and well supported, and I especially liked how the Discussion section thoroughly explained the limitations of the study. My chief concern with the report, however, is that the results may be overly pessimistic. I suspect it would be very difficult for experts to identify demographic rates under "average" environmental conditions but absent any density dependence. Thus, it would have been useful to see how the theoretical rmax (e.g. from allometric relationships) compared with those elicited from the literature and experts. Also, as the authors point out, theta in the logistic model is likely to be >1 for sea ducks. If so, then h(MSY) would be somewhat higher than that reported. Again, a comparison with theoretical values would be instructive I think. Most of my remaining comments are relatively minor.

(page 3) "However, a lack of published information and limited monitoring data for these populations led us to assume linear density dependence throughout this assessment" The implications of this assumption should probably be stated here (i.e., generally more conservative than theta >1).

Response: We agree and have added an explicit statement.

(page 4) "We used a post-birth pulse formulation of the projection matrix with four age classes for eider and three for the other species." I suggest you say explicitly that this matrix is for three age classes.

Response: We agree and have addressed this in the report.

(page 5) "Moreover, estimates of statistical variability associated with published demographic rates did not represent the uncertainty about the true population means for those parameters." Not sure about this. If these are means of annual estimates then the variance of the mean captures both sampling and true variability. One could partition the variance if sampling variability for the annual estimates were available to get out "true" annual variation.

Response: We clarified the statement to indicate that variance around parameter estimates from one or a few local studies is not indicative of uncertainty about the population-level values for those parameters.

(page 8) "We used Monte Carlo simulation (10,000 iterations) to select randomly, and with equal probability, from the probability distributions…" Not sure what is being said here. Do you mean that for every draw from the literature you took one from the experts. Obviously draws from a given distribution are not done with equal probability.

Response: Correct, we sampled with equal frequency from the distributions based on the literature and each of the experts.

(page 9) "The second fecundity distribution, based on harvest age ratios, was generated by sampling 10,000 values from the probability distribution constructed for the 2004-2013 mean female harvest age ratio and estimated variance, and then dividing these by a similar sample from the probability distribution of differential vulnerability generated from the values provided during the expert elicitation." Probably should mention here that, in contrast to the other method, harvest age ratios do not provide age-specific fecundities. Thus, depending on the age structure of the population, the two methods could provide very different results.

Response: We agree, and we amended the text accordingly.

(page 17) "While the methods we used to estimate $r_{\max}$ allow for limited incorporation of age/stage structure, sparse monitoring data for sea ducks limits rigorous examination of the effects of age/stage structure, population inertia/transient dynamics, environmental and other stochastic effects, cohort-targeted harvest, and other factors affecting the dynamics of structured populations." This is always a little unsettling when deriving allowable harvests for an age-structured population. In reality, populations are probably rarely, if ever, at their stable age distribution (i.e., transient dynamics are the norm). In practice, this means the impact of a given level of harvest may be entirely unpredictable. Perhaps you could provide the damping ratio (the ratio of the first two eigenvalues; see Caswell 2001) to provide some sense of the expected strength of the transient dynamics. Other diagnostics for transient dynamics are also available – e.g., Caswell 2007 Eco. Ltrs. But perhaps it is sufficient to provide this warning here.

Response: We agree that damping ratio, which provides a dimensionless measure of the rate at which a population returns to equilibrium dynamics following a perturbation, as well as other measures of transient dynamics, provide useful information for harvest managers. Such metrics may provide managers with useful information in assessing risk related to management based on asymptotic dynamics in data-poor situations. We will further consider measures of transient dynamics in future analyses and believe such considerations are highly relevant to harvest management.

Reviewer #2

(page 3) "The best available population estimates of the sea duck species of interest are from winter surveys." Not always. WBLSC survey is a breeding survey - a winter survey is not feasible for that population.

Response: We agree. We have changed the sentence to read, "Most of the best available population estimates…." and added text late in the report to specify (1) that a breeding population survey of western BLSC has been conducted, and (2) why we did not use estimates from that survey. While a complete winter survey for western BLSC may not be logistically feasible, the winter estimate that we used (based on the sum of estimates from a number of partial surveys conducted over a period of years) was credible when compared with the breeding population survey estimate, considering that the winter estimate includes young-of-the-year birds and the breeding population survey does not.

(page 11) "The simulation for American common eiders produced a median $r_{\max}$ of -0.0018 (Table 2) suggesting that, on average, there is little growth potential for this population under current environmental conditions." But pops in northern part of range are apparently increasing.

Response: The guidance provided by the SDJV was to address American common eiders as a single population. On a population-wide scale, the experts from whom we elicited information on dresseri indicated that production is poor, and this resulted in the low r_{max} estimate. No data or expert opinion were presented during the deliberations of the SDJV Harvest Management Subcommittee or the formal elicitation for dresseri breeding in the northern portions of their range to contrast with that available for Maine and the Maritimes.

(page 12) "As with eiders, conclusions about appropriate harvest levels were influenced heavily by adult survival as evidenced by its high slope, though low levels of uncertainty associated with estimates of adult survival…" Why low levels of uncertainty? - we have no data on survival for this population.

Response: The adult survival distribution for eastern black scoters came from the estimates of 3 experts. We had provided them with 3 sources of information on eastern black scoter survival: (1) an estimate based on PTT-tagged black scoters, (2) the consensus of opinion "best guess" of the SDJV Harvest Management Working Group members, and (3) estimates based on allometric relationships (Johnson et al. 2012). Apparently the 3 experts felt that this was enough information to make them more sure of adult survival rate than they were of other black scoter demographic parameters. Note that "low" uncertainty means low relative to other parameters of interest, in this case the fecundity parameters.

(page 12, Western Black Scoter) "Sensitivity analysis identified 3 fecundity parameters as the highest priorities for research or monitoring: nest success, duckling survival,…" Population size probably didn't come out as a high priority because we have (had) reliable and relatively precise estimates of pop size (based on a survey that has since been discontinued). By not identifying pop size as high priority, it suggests that continued monitoring is less important than fecundity parameters. I think that sends the wrong message - we should applaud the "good" survey data and recommend that the survey be resurrected. Realistically, we're not going to estimate the fecundity parameters.

Response: The primary purpose of the assessment was to identify information needs to support harvest management. We believe that the fact that population size was not identified as a priority need is tantamount to a "job well done" statement about the quality of the existing population data for the western black scoter. While we don't know as much as we'd like about population size, the uncertainty we have about contemporary population size is not as influential on our comparisons of allowable and observed harvest as uncertainty in some other demographic parameters. If periodic surveys of abundance are not conducted in the future, it would be necessary to update the uncertainty distribution for population size which could change relative information priorities. Also, while we acknowledge present challenges, we are not as skeptical about the potential to estimate critical fecundity measures in the future. It may be impossible to estimate individual fecundity parameters on a population scale using conventional (local field study) methods, but maybe if this is identified as a priority need someone will come up with an alternative method of estimating fecundity, perhaps using recently developed or emerging technology (e.g., high resolution aerial photography to identify HY vs AHY birds during winter surveys). Finally, we believe that it is the task of the SDJV to consider *the priorities identified in this report, which reflect solely the influence on uncertainty in specific parameters on inferences about contemporary harvest levels, within a broader context that includes consideration of technical feasibility, cost, and other logistical or administrative factors.*

(Table A2, Eastern black scoter nest success) In absence of data for EBLSC, why not use the values for WBLSC?

Response: The value we used was the consensus of opinion among members of the harvest management working group. That value is similar to the nest success estimate for eastern surf scoter; presumably the working group members felt that factors affecting nest success of eastern surf and eastern black scoters were similar and affected nest success of both populations about the same.

Reviewer #3

In general, this document is longer than the analysis of sparse data would justify. We understand there are few studies that provide estimates of parameters and the expert opinion approach was a good idea, but there were also so few experts (though the number of experts is not exactly documented) that it makes this approach very uncertain in itself.

Response: Table B1 provides the number of experts who participated.

There is so much uncertainty, we're not sure all the analyses are warranted beyond simply concluding that 2 spp (LTDU and COEI) from the review of all parameter estimates show the greatest urgency of parameter improvement. Table 2 says it all. Both of these spp have allowable take estimates <0 (i.e., they don't support any take) and r_{max} estimates <0 (i.e., they show no growth). We fully understand and appreciate review and summary of all known data and best guesses to prioritize research, but these 2 parameters for these 2 spp jump out as the definite areas urgently needing more work.

Response: We agree there is sparse information to draw from for these species, and the report certainly has limitations. However, it is noted in previous comments and in the introduction of the report, that identifying the areas in which to focus limited resources on for improving the science needed to inform harvest management decisions is the driving factor behind this assessment.

Reviewer #4

I have read the "Sea Duck Harvest Potential" report and, in my opinion, it is a really good piece of work. As everyone knows, there is tremendous uncertainty in sea duck demographic rates and population size, both of which are needed to estimate a sustainable harvest. The authors of the report seem to have used the best available information as well as standard methods of expert elicitation to derive uncertainty distributions for maximum sustainable take (or harvest rate).

They then used this with estimates of actual harvest (also uncertain) to derive the probability that the actual harvest is greater than the predicted maximum sustainable take. On its own, this quantity might be useful to decision makers, but they then did a sensitivity analysis that shows which parameter would be most important to reduce uncertainty in order to better estimate the difference between allowable and actual harvest. If sustainability of harvest is the primary issue of concern with decision makers, then this sensitivity analysis should be very useful to decisions allocating research dollars. On a technical note, the regression based method they used for the sensitivity analysis is very appropriate and useful in this context. The authors have also done a great job stating the assumptions in their analysis (there are a lot) and in providing discussion about their recommendations. This report should meet their stated goal of providing guidance for SDJV research priorities (and hopefully funding allocations).

A few minor comments:
1. It would be useful to have a < 1 page summary of the major results and recommendations.

Response: We agree. We have added an executive summary.

2. I think it would be nice to give 95% credible intervals on quantities in the text so that the level of uncertainty comes across immediately and clearly. Now, the reader is forced to find Table 3 or put off thinking about the level of uncertainty until reading through the report. This is distracting to people wondering about the range of estimates.

Response: We agree, text amended accordingly.

3. I was hoping for a little more discussion of the decision context and, specifically, the application of expected value of (perfect or sample) information (EVI) calculations. However, the authors addressed this very topic in a paragraph at the end of the report. Given their thoughts, I can see that the sensitivity analysis is appropriate, but I wonder if further development of the harvest decision problem might lend itself to an EVI calculation and might strengthen the report even more. Are there any suggestions along these lines that can be given?

Response: As indicated in the Discussion section, we agree that this would be a natural follow-up action, and we would anticipate that such formal analyses would occur once the decision problem is more completely specified (i.e., objective function, alternate system and control models, expressions of relative confidence in models, etc).

Reviewer # 5

First, we question if you can draw inferences about American common eider and long-tailed duck in this modeling framework after the finding of negative values for $r_{\max}$. Runge (2004) is cited in the section of the report where rmax is discussed, but Runge (2004) notes that this overall approach assumes logistic growth and density dependence. Given these assumptions, it is not possible to have r max values that are zero or negative. Our thoughts are that either the parameter estimates used to estimate r_{max} are biased (or limited as we know they are for sea ducks) or these species do not conform to logistic growth. In either scenario, we wonder if it is appropriate to draw inference for these species using these data/modeling approach.

Response: We do not believe that a finding of negative median values of r_{max} *invalidates this approach.* r_{max}*, under our definition, which we believe is most relevant to harvest management, is not constrained to positive values, nor does the estimation method (projection matrix) make any assumptions about density dependence. We view* r_{max} *as reflecting the life history characteristics of a species expressed in a given "environmental" setting when no density dependent regulation or harvest is occurring. Under that definition, it is possible that prevailing environmental factors (all density independent factors limiting population growth other than harvest) could result in conditions where positive growth is not possible. It is true that the PTL framework and the equations for computing PTL are derived from a logistic (or theta logistic) growth model, however, we don't feel that that invalidates our results or conclusions. Rather, estimated values of* r_{max} *that are 0 or negative would imply that there is no harvest potential on average. It is important to note also that given the probabilistic nature of this assessment, and assuming the range of uncertainty described for the input parameters is reasonable, the true median values of* r_{max} *for these species could be positive and we felt it was important to demonstrate the uncertainty in estimated values of allowable harvest, recognizing that negative values of allowable harvest are nonsensical. Perhaps truncating the distributions of* h_{max} *would be more appropriate. We certainly don't dispute your conclusion that the probability distributions for some of the demographic parameters may not reflect true median values and we maintain a healthy skepticism about the conclusions given the large degree of uncertainty, but we hope it's a starting point to target key uncertainties and improve estimates.*

The models in the harvest assessment assume a population size objective of K/2 (i.e., the population level that yields MSY). Given that both r_{max} and a median optimal harvest are estimated, it should be possible to easily derive the actual number of this population size objective (or at least the median of this objective). We recommend presenting the inferred population size objective levels and incorporate those into the conclusions of over/under harvest. The question then becomes are we above or below the population size objective given the harvest. We calculated this for several of the species and while it's true harvest levels are above sustainable levels, the estimated population sizes are also above the unstated population size objectives. If we are above the population level objective, then overharvest is not an issue. Similarly, if we are under the population objective, then under-harvest is desirable. The worst case scenario would be under the objective and overharvest.

The main point is that conclusions regarding under or overharvest must consider the size of the population relative to the management objective.

Response: With respect to your suggestion that we incorporate our implied population objective (i.e., equilibrium population size under a MSY harvest strategy, N_{MSY}*) we are not quite certain how you are able to derive* N_{MSY} *and ask for further clarification or an example calculation. We estimated* r_{max} *from individual component demographic rates using a projection matrix. We did not have sufficient time series of data necessary to actually develop logistic growth models and directly estimate* r_{max} *and K. To our knowledge, the determination of* N_{MSY} *would require an estimate of K or the assumption that our contemporary harvest rates are close to* h_{MSY}*. Given* *our findings, we have reason to doubt that assumption. We agree that what you suggest would be a very useful addition to this assessment, particularly in assisting harvest managers in evaluating relative risk, but also in developing priorities for SDJV research and monitoring investments. We are just not certain that it is possible to derive* N_{MSY} *with the information at hand. Again, we would very much welcome further clarification of the approach you took and continued dialogue.*

Follow-up comments from Reviewers #6

As for the applicability of this approach to the eiders and long-tailed ducks, I guess my view is that if all the processes that regulate a population are density independent, then you really have the wrong model from which to be drawing inference.

Response: We disagree. By definition, r_{max} *occurs under conditions where no density dependent regulation is occurring, so only density independent factors and the life history traits of a species affect the magnitude of* r_{max}.

You note in your manuscript that your approach assumes a logistic model applies, but as soon as you estimate an $r_{\mathrm{max}} < 0$, you have demonstrated that you have violated that assumption.

Response: This assessment results in probability distributions for r_{max} *for some populations that include both positive and negative values. Our estimates of* r_{max} *are not derived on the basis of the logistic model and are not theoretically constrained to be greater than or equal to zero. The PTL formulation, from which we derive estimates of allowable harvest based on estimated* r_{max}*, is, however, founded on the assumptions of the logistic growth model. We agree, that in the case of a simulated negative value of* r_{max}*, the implication is that the logistic model does not apply, that no allowable harvest exists, and that we could reasonably constrain allowable harvest to be no less than zero. However, our primary purpose is to assess the effect of parameter uncertainty on uncertainty about the difference between observed and allowable harvest. Considering probability distributions that include negative values of allowable harvest facilitates this comparison, therefore we elected not to constrain values of allowable harvest to be greater than or equal to zero in the initial assessment. In order to determine the effect of constraining allowable harvest to values greater than or equal to zero, we subsequently constrained allowable harvest and re-ran the simulations and sensitivity analyses. In many instances, the prioritization of parameters from the sensitivity did not change, however, we note where they did in the results section of the report.*

As I am sure you realize, the yield curve is flat and there is no sustainable intersection between a fixed rate line and the yield curve. Under that scenario, over-harvest does not simply yield an equilibrium population size somewhat less than MSY, it theoretically causes extinction. So can you demonstrate that the results from your sensitivity analyses are still valid if the eider and long-tailed duck populations don't behave according to your underlying model? Of course the other possibility is that your parameter estimates are biased yielding a low (biased) estimate of r_{max}. But throughout this manuscript you went to great lengths to achieve good estimates of parameters, presumably because actual parameter estimates matter in calculation of sensitivities. So if one or more of the parameter estimates are substantially biased, are your sensitivity estimates still valid? Perhaps you can demonstrate that my concerns don't really matter, or discuss these issues and argue that your conclusions are robust to violations of the assumptions.

With regard to the second question, in looking at your equation 4, I interpreted H_{msy} as the harvest level at K/2. That is, the optimal harvest level for the population. Given that, all I did was take your mean allowable harvest and rearrange equation 4 to estimate the population size N = K/2. I think I see now that is not the case. Functionally what you are doing is assuming that N_t=K/2 and then asking if we are over or under harvesting. Thus, using the logic from Runge et al., any rate between 0 and r_{max} is sustainable. So now I am left wondering why you did not just use your estimates of harvest and population size to model harvest rate, and compare the distribution of harvest rate with the modeled distributions of $r_{max}/2$? If you did that you could utilize equations 4-6 of Runge et al. and suggest where that would lead the population in relation to MSY. So, upon further consideration, I think I was misinterpreting the parameters in your equations. Overall, however, the logic of my comment still applies and would seem to warrant some discussion. That is, your models assume we have a management objective of K/2, but we really have no idea what this number is or where our current population level falls in relation to that objective. Hence our population objective is really unknown. About all you can conclude is that assuming your estimate of r_{max} is valid and the populations behave according to the model, maintenance of a fixed harvest rate at $r_{max}/2$ should ultimately yield a population at MSY. Interpretation of under/over harvest really needs to consider current population status in relation to the objective which clearly cannot be done here. As I noted in my previous comments, over-harvest is not really an issue if the population is above the objective. If you believe that the long-term decline in sea duck populations occurred because carrying capacity declined, then it is plausible that populations are currently much closer to K than they are to MSY.

So now that I have wrapped my head around this a bit more, we really don’t even have a population objective. But if we define our objective to be N that achieves MSY (that way we don’t have to have a number), then the desired harvest rate is $r_{max}/2$. So you used the best demographic information to estimate $r_{max}/2$, and you use population and harvest data to estimate harvest rate. Then you can compare the 2. The conclusion of over- or under-harvest needs to be made in relation to the harvest rate (not the number of birds harvested).

Response: We agree that it is not possible to derive N_{msy} given the data at hand. We do not agree that a median observed harvest which exceeds median allowable harvest is inconsequential if population size is thought to be high, unless a management goal exists to reduce population size. Consistent extraction of harvest that exceeds allowable harvest would, at best, cause the population to equilibrate at a lower average population size. When allowable harvest reflects a management goal of maximum sustained yield, a consistent exceedance of allowable harvest level is particularly disconcerting. Given uncertainty and stochasticity, this situation would suggest a high degree of risk of overexploitation. A lack of refined and frequently updated population size estimates further exacerbates this risk (e.g., the collapse of some commercial fisheries managed under a fixed quota system and inadequate population monitoring).

Thanks for giving me the opportunity to read this material, it has made me consider sea duck harvest in far more detail than I had previously.

Reviewer # 6

Comments on the report titled, "Implications of demographic uncertainty for harvest management of North American sea ducks." Comments are brief, general (thus, not technical) and regard the Pacific Flyway populations of sea ducks considered in the report.

This is a useful 'first-step' toward assessing the influence of uncertainty in demographic parameters on harvest.

Comments:

* The PTL framework appears most useful for highlighting important research and monitoring information needs and prioritizing those needs with regard to SDJV harvest management (i.e. a gap analysis).

* However, addressing the identified high and medium priority needs are (for the most part) improbable or very difficult (at least with current technologies and budgets; and at a large enough spatial/temporal scale), so the models may not be well-suited for guiding future SDJV investments. I guess it gives a good indication of the most important information needs that are unlikely to be addressed in the near and most likely distant future. So in a sense it may force scientists to think about novel ways (or workarounds) to obtain this information.

* Given the uncertainty in model parameters is largely unsupported by data, the PTL framework is currently a useful (for some things) modeling exercise; but for the Pacific Flyway, it's a weak tool to guide harvest management decision making. However, it does add focus to populations with low values of r_{max} and over-harvest risk.

These comments are likely fairly obvious to those that reviewed this report (so take them at their worth) and the authors acknowledge many of these inadequacies in the report. This is a good beginning to a difficult and labor-intensive task. But the harvest assessment in its current state is most useful for identifying information needs, at least with regard to harvest of western populations of sea ducks.

Reviewer # 7

This is a useful first-step in assembling data available for assessment of harvest in sea ducks and provides a foundation for setting SDJV research priorities. I think that this assessment should not be the final evaluation of priorities but rather that it be used and built upon to develop those priorities as a group.

Perhaps the most important shortcoming of the assessment is that differences in quality of parameter estimates are not recognized, at least not entirely. Parameter estimates derived from large-scale studies published in peer-reviewed journals should be considered to be more reliable of actual population parameters than those derived from unpublished reports, with those derived from expert opinion being less reliable. The expert elicitation process was used to circumvent this issue but in the absence of alternative sources of information, one would expect experts to substantiate the estimates previously derived, especially since these same experts were involved the determination of parameter estimates in the first place.

The result is that all estimates are treated equally, albeit some component of uncertainty is taken into account. There is no distinction between assessments that are mainly based on expert opinion or local studies (e.g. LTDU) than from those that are mainly based on published data (e.g. COEI). When the results from such assessments are used, in conjunction with measured harvest levels, to determine priority information needs, the problem becomes compounded.

Response: See response later under recommendations.

During one of the original conference call of the Harvest Assessment Working Group, the use of different methods to estimate r_{max} was discussed. Some of these approaches are simpler and require fewer parameters (e.g. the Demographic Invariant Method requires estimates of adult survival and age-at-first breeding) while others are more complex and provide a potentially more realistic portrait of a species' demography. Importantly, peer-reviewed publications have established that despite differences in complexity, these different approaches provide comparable estimates of r_{max}, even in the case of species with life-histories similar to that of sea ducks. Different approaches may be appropriate for different populations and as such, the group had discussed comparing a suite of approaches on the Northern Eider (Borealis) data set. This population, despite not being shared between US and Canada, was closely related to the ones being assessed, presented the most robust estimates of demographic parameters of all species under study, and a fully parameterized population model was available. Unfortunately, it was determined later by the writing team that more simple approaches would not be considered in this assessment. The reasons why were never clear to me.

Response: We did initially compare projection matrix, Slades, and the demographic invariant method (DIM) to estimate rmax. The decision to not include the DIM method was based on the dissimilarity of the results to the other methods employed (projection matrix, Slades) and more importantly on the fact that we felt that the DIM method has properties that are more limiting to management application. In the original proposal for this work we state:

"We note that there are likely to be inherent advantages and limitations of each approach to estimating r_{max} in the context of this take assessment. The Slade et al. (1998) method has the disadvantage of requiring estimates of a larger number of input parameters, however, it has the advantage of greater generality and flexibility in the face of changing survival and recruitment processes which could affect r_{max}. Slade's method also might enable formal analysis of the sensitivity of take prescriptions to age-specific survival which could be useful if there is an age-bias in the harvest. The DIM method of Niel and Lebreton (2005), on the other hand requires the estimation of only 2 parameters, namely adult survival and age at first

reproduction. However, it sacrifices generality that may limit flexibility in harvest management applications since it offers no means of updating r_{max} *estimates to reflect changes in underlying recruitment processes." Though we did not discuss it in the proposal, we note that the DIM method would also result in a lower* r_{max} *estimate with an increase in adult survival. Therefore, if underlying density independent limitations to survival were relieved and adult survival in the absence of harvest increased (e.g. reduction of predation or contaminants), the DIM estimate of* r_{max}*, and thus allowable harvest, would actually decline. That would provide little incentive for conservation measures. Our belief is that DIM is more useful for broad comparisons of* r_{max} *across taxonomies, less so for recurrent regulatory or conservation decision making."*

We did, in fact, compare projection matrix, Slades, and DIM estimates of r_{max} *based on the data published for borealis in Gilliland et al. 2009. We found projection matrix and Slades estimates to be slightly less than 0.07, nearly identical to those published by Gilliland, while the DIM estimate approached 0.11. Despite these results, based on your comments as well as Reveiwer 8's and those made during the recent Harvest Management Working Group presentation, we have, however, reconsidered the utility of computing median DIM* r_{max} *estimates and contrasting these with the estimates we derived though projection matrices and estimates of both survival and fecundity parameters (i.e., best estimates of those parameters at low population density and no harvest).*

The DIM method, based on fundamental relationships between survival, fecundity, and generation time, and requiring only estimates of adult survival and age-at-first breeding, can be thought of as a "theoretical maximum" estimate of r_{max}*. We believe that the DIM* r_{max} *could be thought of as* r_{max} *under the most optimal environmental backdrop. From a harvest management perspective, population-specific estimates of* r_{max}*, if obtainable, have greater utility as they apply to the specific environmental setting in which a population exists at a given time. Comparing those theoretical maximums to our* r_{max} *distributions will highlight differences that are due to one, or both, of 2 factors: 1) poorer than optimal environmental backdrop (density independent factors like climate, contaminants, predation, etc), or 2) use of parameter values that are not reflective of values that would occur at low population density for an unharvested population.*

We believe that comparison of r_{max} *values computed through the projection matrices and those computed via the DIM will provide another means to contemplate, based on biological intuition, the degree to which baseline environmental conditions, poor estimates of parameters, or use of parameter values not representative of an entire population might be contributing to the differences.*

All that said, the DIM method comes with its own assumptions and caveats that are important to keep in mind as well.

I would personally not advocate use of the results from certain populations for making harvest management decisions because of the quality of the data it rests upon, but doing so should not cause prejudice to the resource given the apparent conservatism of the estimates of sustainable harvest. I am concerned however about the way the information was used for informing research priorities of the SDJV. The modeling approach chosen for each population had a direct impact on priority-setting, despite the fact that alternative methods (that were not tested in the process) may have led to different conclusions. In my opinion, more complex approaches are appropriate when data to support them is robust and representative. When this is not the case, more simple approaches that rely on fewer parameters may be more appropriate (i.e. you don’t compound lack of accuracy among a large number of parameters).

Response: See above response.

Finally, there was no consideration given to the impact of a demographic parameter on population growth rate in the priority information need analysis, other than its influence on estimated potential harvest level. This is an important consideration when determining appropriate harvest management strategies and should be highlighted in the SDJVs research priorities. In contrast, the assessment highlights nest or hatch success as being a high priority information need in 5 out of 7 populations. Given the relative lack of influence on this parameter on population growth rate of species such as sea ducks, I doubt that efforts to reduce uncertainty of this parameter will translate into better efficiency of harvest management.

Response: We disagree. The slope of the relationship between standardized parameter values and the difference between allowable and observed harvest is influenced by the classic sensitivity of growth rate to a parameter as well as the uncertainty surrounding the parameter estimate.

Population-specific comments:

<u>American Common Eider</u>: probably of the species with the most robust parameter estimates. Unfortunately, several are representative of only a segment of the population (Atlantic Canada + Maine). Productivity data used in the assessment does not appear to represent conditions in the St. Lawrence Estuary, Gulf and Newfoundland. Including information from other important segments of the population should be a high priority of the SDJV.

Response: We agree and this point is emphasized in the report.

This is also the only species for which the assessment was based on harvest rate. This decision de facto made that parameter irrelevant in the priority-setting process even though a perfectly good assessment could have likely been made using population size. This would have also had the added benefit of not relying on harvest rates derived from only part of the range.

Response: We understand the concern. The decision to include only the harvest rate based assessment of PTL was based on this being a more robust harvest assessment framework than that based on total harvest. The harvest rate based framework requires only good estimates of harvest rate, while the total harvest formulation of PTL requires estimates of both total harvest and population size, which are currently lacking. For American common eider we felt that improving the banding program to ensure representativeness was more tractable than developing unbiased population estimates and estimates of total harvest, which would have to include the need to develop a population survey with adequate sampling frame from ground up, *good estimates of subsistence harvest, parsing apart borealis and dresseri sport harvest, etc. That said, we were aware that not including a total harvest based formulation selects de facto against improved population surveys and total harvest estimates as priority information needs. Incorporation of a total harvest based formulation of PTL for American common eider could be addressed relatively easily in the future.*

Eastern and Western Surf Scoter: There is little basis for modelling r_{max} differently for those two populations. Tables A4 and A5 show that almost all parameters used are the same for the two populations (with the exception of population size and retrieved harvest), yet the identified priority information needs are completely different for each population. Only population size comes out as a high priority (agree on this one). Why does adult survival come out as a high priority for western SUSC but not for eastern? The 'quality' of both estimates is pretty similar (accuracy of the eastern SUSC survival is probably not great as the analysis is based on a pretty small sample) but yet their ranking is markedly different. Juvenile survival is a high priority for westerns but doesn't even make the medium priority list for easterns. The latter was based on expert opinion and as such any differences between the two populations is purely speculative.

Response: We agree that there was little basis for separate assessments. However, the guidance we received from the JV was to consider these as distinct populations. The differences in the conclusions are a result of differences in input parameter values that were based on limited data, expert opinion, or both.

Long-tailed Duck:

Most parameters are based on expert opinion or small scale local studies. I would not give much weight to the assessment for this species. All parameters need to be better estimated. However, for a species such as this, a simpler modeling approach may have been more appropriate (use more complex approaches when data can support it), which could have influenced the prioritization process. A focus *on adult survival and age-at-first breeding would allow an assessment of sustainable harvest.*

Response: We agree with the concern over the estimated value of r_{max} and indicate our low confidence in this result in the report. We don't, however, believe that this result invalidates the modeling approach (see discussion above contrasting the DIM and other approaches to estimating r_{max}), but only that it suggests that the input parameter values are not reflective of actual population means. The simpler DIM is rooted in fundamental relationships among survival, fecundity, and generation time and requires only adult survival and age-at-first-breeding as input. Therefore, DIM can be thought of as producing theoretical maximum r_{max} estimates. If the environmental conditions experienced by a specific population are suboptimal, it may not be possible for r_{max} to ever attain theoretical maximum values, and DIM could overestimate harvest potential. An example might be dresseri in Maine and the Maritimes where *we believe fecundity is very low, possibly due to predation on ducklings. Here the population-specific r_{max} may be substantially lower than the estimate provided by DIM which assumes that general relationships among survival and fecundity apply. The situation is a little different with long-tailed duck. Here, no good biological hypothesis exists to explain the negative estimate of r_{max} for the widely distributed continental population of long-tailed duck. That brings the median values of input demographic rates into question, we agree. We report the median estimate of r_{max} for long-tails and the associated parameter distributions in the interest of documenting the extreme uncertainty associated with this population and highlighting the need for better information.*

<u>*Recommendations:*</u>

- Use some measure of input parameter quality in the priority-setting process

  Response: We believe this is already a part of the assessment given the effort to characterize the uncertainty distribution about each parameter estimate.

  Ensure that parameters that are not being used in this assessment but that could be used if a different approach was used are not eliminated from the priority-setting process. SDJV priority research needs, as identified through this process, could influence funding decisions for several years so we should make sure that we don’t let anything important fall off the table. Examples include COEI population size and harvest rate information from important parts of the range

  Response: See the response above.

- Take into account classic sensitivity of parameters (ie their degree of influence on population growth) in then priority-setting process

  Response: See the response above.

Reviewer # 8

I still remain somewhat uncomfortably with the results of the harvest assessment. I will only touch on a couple of major issues.

I find the concept r a bit confusing, especially since there more than one growth rate that uses r. My understanding now, which maybe incorrect, is that there are discrete and continuous forms of r, there’s rmax, and that rmax is a theoretical value that we would likely never observe in a population. I’ve included a Figure 1 from Pianka (2000) that I think captures the relationships between fecundity, mortality, r and rmax, and population size.

Response: Yes there are different expressions for continuous and discrete growth processes. All the work in the report is based on discrete growth. In the logistic growth model, r_{max} *is never achieved but the growth rate approaches* r_{max} *as population density declines toward 0. In actual populations it is possible that growth rates very close to* r_{max} *could be attained, say in a new release where population density is very low and there is no density regulation occurring and the population is not being harvested. Its important to note, however, that* r_{max} *is not a single value for a given species.* R_{max} *is context-specific, that is, it is dependent on the background environment in which a population exists (i.e., all the density independent factors influencing the population with the exception of the mortality source of interest, in our case harvest). See the discussion in the response to Reviewer 8's comments of the comparison of theoretical maximum values of* r_{max} *derived through the DIM method to those derived in the report using projection matrices and estimates of fecundity and survival parameters.*

I am still struck that harvest potential is so low for most populations of sea ducks. We generally think of sea ducks as being on the k end of the spectrum: i.e. long lived and low reproduction. As such, I think it is reasonable to expect that most populations of sea ducks should be around k unless they are some other limitation (e.g. harvest, habitat degradation, etc.). If a population is near k, then we should expect that levels of fecundity should be low, and mortality be high, and the any r that is estimated from a projection matrix with parameter values measured around k would approach 0. Using this estimate of r would result in little or no harvest potential. This would be incorrect, as rmax should be estimated when the population size is very low. My first major concern is that the r's used in the assessment maybe a very poor substitute for rmax as they are estimated at a relatively high population density.

Response: As the populations under study are subjected to harvest we don't believe that they are at K, however, given their life histories h_{msy} *may occur at equilibrium population sizes closer to K than for more r-selected species. We do not account for this in the report because we assume that theta of the theta logistic growth model is 1 in this first iteration of the assessment. We disagree with your expectation of mortality and fecundity at K. At K, by definition, fecundity and mortality should be identical. As we have indicated before, we did our best through the literature review and expert elicitation to specify parameter values that would be associated with low population density and no harvest. This is a very difficult exercise and there is much uncertainty so it is certainly possible that median parameter estimates don't reflect true population values.*

I think we have underestimated the growth potential for sea ducks. For example, in the NCOEI model were built a projection matrix for an un-hunted population that had a lamda of 1.07, a growth rate that has been observed in other non-hunted populations. In 2008 (Gilliland et al. 3rd Sea Duck Conference), we re-evaluated the NCOEI model after the harvest restrictions in Greenland were implemented in 2001. We were very surprized to see the performance of NCOEI. In Greenland, colonies that were monitored had lambas of 1.1 to 1.15 and Grant's East Bay colony 1.11 following the restriction. With observed harvest, the model only predicted a lamda of 1.05 for Greenland. We felt the underestimate of the model could be due to underestimating population size, or that the population can grow more rapidly at low population density. I think it is also interesting to note that we are observing similar growth rates in dresseri along the Quebec north shore and northern Newfoundland (lamda = 1.11-1.15), and that these estimates come from populations that are hunted.

Response: See discussion immediately above and in response to Reviewer 8's comments. We have included estimates of theoretical maximum r_{max} values from the DIM method to contrast with r_{max} values estimated from both fecundity and survival parameters. While we, too, have low confidence in the estimated r_{max} for long-tailed ducks in particular, we note that the differences between the theoretical maximum r_{max} values derived through DIM and those derived via the projection matrices may be due to one or both of 2 factors: 1) poorer than optimal environmental backdrop (density independent factors like climate, contaminants, predation, etc), or 2) use of parameter values that are not reflective of values that would occur at low population density for an unharvested population. The objective of the assessment was to attempt to characterize uncertainty and the implications of that uncertainty for research and monitoring to improve harvest management. The wide distributions associated with r_{max} for all populations speak to the degree of uncertainty given available information.

I think it is clear from Figure 1 that if hunting mortality is included in the estimate of r, then r is not equivalent to rmax. My second issue that that hunting mortality is likely included in most of the survival estimates. The worst is for longtails where the measured survival rates for the adults are measure apparent survival and not true survival. It seems to me that for Long Tails, the experts were highly influenced but the published values and did not take into account that the estimates were not true survival. I also feel that the experts did not take into account that most of the estimates of survival for all populations include harvest mortality.

Response: See previous responses. We too have low confidence in the result for long-tails.

You provided 2 responses related to these issues to my last comments:

> *Response: We attempted to utilize demographic parameter estimates that would be consistent with the conditions under which r_{max} would be achieved, i.e., low density and no harvest. The large degree of uncertainty defined for many parameters speaks to the challenges of developing appropriate estimates of certain parameters under these conditions, which typically required some degree of speculation beyond available data or experience.*
>
> *Response: In the report, we defined r_{max} for the population as the maximum growth rate achievable by a population when that population is not exposed to the source of mortality of interest (in this case harvest), is not under any resource limitations causing density-dependent regulation, and is experiencing otherwise average environmental conditions. We asked experts to provide information on demographic rates under these*

> *specific conditions. With respect to specific comments related to breeding propensity, there are a number of mechanisms which might induce density-related regulation. In addition, we cannot find evidence to support the idea that breeding propensity in sea ducks is density dependent, and the small number of studies of this demographic rate in waterfowl seem to contradict this claim (e.g., see Coulson. 2010. Bird Study 57: 1-18; Sedinger et al. 2001. Journal of Animal Ecology 70:798-807; page 566 in Esler et al. 2015. Chapter 15, in Savard and Derksen 2015. Ecology and conservation of North American sea ducks, CRC press).*

However, I do not think you addressed this issues. I had Greg Robertson review the assessment and it was his opinion that you are “killing many birds twice”. I don’t think that the large levels of uncertainty included in the parameter estimates necessarily addresses thus issue unless the trials were drawn from something like a uniform distribution. I think you need to provide strong evidence that the various parameter values were estimated for populations at low population density and that harvest mortality is not included in the estimates of survival.

Response: See previous responses. We note that this work attempted to characterize uncertainty and its implications for decision making and research and monitoring priorities. Our approach was heavily supplemented by expert opinion, especially for species like long-tails where data availability was particularly poor.

In my initial comments I had suggested that you at least increase breeding propensity to 1 for all birds of breeding age. Above you quote Coulson 2010 and Sedenger et al. 2001 as examples that there is no density dependant affects on breeding propensity. First, I’m not sure that geese provide a good surrogate for sea ducks and waste grain has changed the state of this system, I only wish we had this problem for sea ducks. As for the Coulson study, I have not talked to Dr. Coulson, but I expect the problem there is that k has changed, and the population is adjusting to the new state. I would point out that there is is extreme annual variability in the proportion of adults that were estimated to be non-breeding each year (Figure 4). I would speculate that this is a result in the annual variability in female condition which to me suggests that there is annual variability in the amount of food that is available each year – if this is the case this is density dependence. I would also point out that in this study clutch size (Figure 5) and mean adult survival (Figure 9) were much higher early in this study suggesting the importance of using parameter values estimated at low population density.

Response: We are unclear of your point here, but do feel that we addressed the concerns in your initial set of comments previously.

In my initial comments I also stated:

> “Another potential weakness is that it appears to me that maximum sustainable harvest rate was just calculated as a function of r (or r_{max}), and does not take into account the differential vulnerabilities of the various age cohorts to the harvest. I think that the

> method used to estimate maximum allowable take assumes that the age composition of harvest is proportional to their availability in the population. Younger cohorts have a lower reproductive value than older birds; hence, if the younger age cohorts are more vulnerable to harvest then older birds, then population may sustain a higher overall harvest rate."

And in your response you acknowledged this limitation:

> *Response: We recognize that the age structure of the harvest and the relative reproductive value of the various age classes has implications to maximum allowable harvest if harvest pressure is, or can be, directed toward cohorts of low reproductive value. Sparse data for most sea ducks led us to employ a simple modeling framework that, while allowing for limited consideration of population age structure does not permit direct assessment of the effect of harvest age structure on allowable harvest. We acknowledge this explicitly in the limitations section of the report and would encourage future data collection and refined modeling to better address the management of structured populations*

However, I bring it up again as all the potential biases in the assessment, i.e. survival with harvest, estimates of mortality and fecundity at high population density, and over-estimating the impact of harvest by not account for differential vulnerability of the age classes to harvest are all biased in the same direction which may result in an ultra-conservative assessment on the impact of harvest.

I have a couple of suggestions:

1: for all populations estimate rmax using the maximum value for the various parameters. The justification for this is that the performance of best individuals at high population density may similar to the average individual at low density.

Response: We have already done this by incorporating the probability distributions for the individual parameters and propagating that uncertainty into probability distributions for r_{max} *and* h_{max} *(or* H_{max}*). Simply pull a value from the right tail of the* r_{max} *probability distribution. We have now taken the additional step of computing and comparing the theoretical maximum* r_{max} *estimates from the DIM method with those estimated from both fecundity and survival parameters using projection matrices. See previous responses on the utility and limitations of this comparison.*

2: For long tails, use an alternate method to estimate rmax. I feel the estimates provided are too poor to use the projection matrix approach

Response: See previous response to a similar comment by Reviewer 8.

3: dresseri. If you opt for using the maximum value of the various parameter estimates, then I think the results would apply to generally to the population. But this does not address concerns we have for breeders the Maritimes and Maine. I would run a second model with the fecundity set very low to simulate the very low recruitment for this segment of the population.

Response: The fecundity estimates used in the assessment are low and likely reflective only of the Maritime and Maine segment of this population. We acknowledge this in the report. Again r_{max} *estimates corresponding to maximum values of all the input parameters could be obtained from the assessment as presented by extracting a value from the right tail of the* r_{max} *distribution.*