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An Assessment of the Potential for Recovery of West Coast Transient Killer Whales Using Coastal Waters of British Columbia

Évaluation du potentiel de rétablissement de l'épaulard migrateur du Pacifique Nord-Est vivant dans les eaux côtières de la Colombie-Britannique

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ABSTRACT

Mammal-eating ‘transient’ killer whales off Canada’s Pacific coast are listed as Threatened under the Species-at-Risk Act. A draft Recovery Strategy for transient killer whales was prepared by DFO in 2007, but insufficient information was available to set quantitative recovery goals in that document. Here, we present a Recovery Potential Assessment for West Coast Transient (WCT) killer whales to provide a basis for on-going recovery planning for this population. For this assessment, we used an archive of photo-identifications of individual WCT whales collected during 1479 sightings between 1974 and 2006. We applied a ‘capture-recapture’ approach to the analysis of this dataset and used Bayesian statistical techniques to estimate population abundance and dynamics. These analyses indicate that the WCT population grew rapidly from the mid-1970s to mid-1990s as a result of high survival and recruitment, the latter including immigration of animals into the nearshore study area. Population growth began slowing in the mid-1990s and has continued to slow in recent years as the population approaches an equilibrium point of around 262 whales. The rapid growth of the WCT population in the mid-1970s to mid-1990s coincided with a dramatic increase in the abundance of the whales’ primary prey, harbour seals, in nearshore waters. The recent slowing of WCT population growth suggests a carrying capacity in the mid to high 200s, given the current abundance of marine mammal prey in coastal waters. A Potential Biological Removal (PBR) of 1.60 animals/year suggests that the population could sustain very little human-induced mortality without declining.

RÉSUMÉ

L’épaulard migrateur de la côte canadienne du Pacifique, qui se nourrit de mammifères, est inscrit sur la liste de la *Loi sur les espèces en péril* en tant qu’espèce menacée. Un programme préliminaire de rétablissement de l’épaulard migrateur a été préparé par le MPO en 2007, mais l’information était alors insuffisante pour fixer des objectifs de rétablissement quantitatifs. Nous présentons donc ici une évaluation du potentiel de rétablissement de l’épaulard migrateur de la côte ouest (MCO) pouvant servir de base à la planification continue du rétablissement de cette population. Aux fins de l’évaluation, nous avons utilisé des archives de photographies d’identification de différents épaulards MCO, recueillies au cours de 1 479 observations, entre 1974 et 2006. Nous avons appliqué une méthode de capture-recapture à l’analyse de ces données et utilisé des techniques statistiques bayesiennes en vue d’estimer l’abondance et la dynamique de la population. Ces analyses montrent que la population d’épaulards MCO a connu une croissance rapide à partir du milieu des années 1970 jusqu’au milieu des années 1990, favorisée par un haut taux de survie et de recrutement, ce dernier incluant l’immigration d’animaux dans la zone d’étude littorale. Le taux de croissance de la population a commencé à ralentir au milieu des années 1990 et a poursuivi son ralentissement ces dernières années, tandis que la population s’approche d’un point d’équilibre d’environ 262 épaulards. La croissance rapide observée entre le milieu des décennies 1970 et 1990 a coïncidé avec une hausse radicale de l’abondance de la principale proie des épaulards, le phoque commun, dans les eaux littorales. Le récent ralentissement de la population d’épaulards MCO indiquerait que la capacité de charge oscillerait entre 250 et 300 bêtes environ, compte tenu de l’abondance actuelle des proies de la famille des mammifères marins dans les eaux côtières. Puisque la plupart de ses proies sont vraisemblablement à un niveau d’abondance historique ou presque, si l’épaulard MCO se stabilise comme on le prévoit, il pourrait être considéré comme étant « rétabli ». Toutefois, compte tenu de sa petite taille, la population d’épaulards MCO continuerait d’être vulnérable face à toute augmentation des mortalités. Un taux de prélèvement biologique potentiel de 1,60 bête par année semble indiquer que la population ne pourrait soutenir que très peu de mortalités d’origine anthropique sans diminuer.

INTRODUCTION

In April 1999, the northeastern Pacific transient killer whale population was designated Special Concern by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC). The status of this population was reassessed in 2001 based on an existing status report (Baird 1999) and an addendum containing updated information (Trites and Barrett-Lennard 2001), and uplisted to Threatened in November 2001. Reasons for this designation were that it is a "small population that eats marine mammals", and "individuals have high levels of toxic pollutants". This population became legally listed on Schedule 1 with the proclamation of the Species-at-Risk Act (SARA) in 2003.

As required by SARA, a Recovery Strategy for Transient Killer Whales in Canada was prepared by Fisheries and Oceans Canada and posted for public comments in 2007 (Fisheries and Oceans Canada 2007). Once accepted by the Minister of Fisheries and Oceans, an Action Plan will be developed to achieve recovery goals and objectives developed in the Recovery Strategy.

DFO Science has recently established a Recovery Potential Assessment (RPA) process to provide information and science advice for meeting SARA requirements for listed species, and for deciding whether to add species to the list. An RPA is intended to assess current population status, identify the scope of human induced mortality, and describe the characteristics and availability of critical habitat. At the time the Recovery Strategy for transient killer whales was drafted, an RPA was not available. Our intention in this report is to provide an RPA for this population that will serve as the scientific basis for establishing population targets and assist in efforts to achieve other recovery objectives and goals described in the Recovery Strategy.

BIOLOGY AND LIFE HISTORY

The killer whale *Orcinus orca* is the largest member of the family Delphinidae and one of the most widely distributed mammals. It occurs in all the world's oceans and most seas, but is most commonly found in coastal waters in high latitude regions. There is an estimated global population of at least 50,000 (Forney and Wade 2006). It is the apex marine predator, capable of feeding on a great diversity of prey, from the largest whales to small schooling fish. It has no natural predators. Despite being a generalist predator as a species, different populations of killer whales may have highly specialized foraging strategies and diets.

Three distinct assemblages of killer whales have been described in coastal waters of the northeastern Pacific Ocean. These assemblages, named *transient, resident* and *offshore*, differ in diet and foraging behaviour, acoustic behaviour, morphology, and genetic characteristics. Despite having overlapping ranges, these assemblages do not mix and are thus socially and reproductively isolated from each other. Transient killer whales specialize on marine mammal prey, though they occasionally kill and eat seabirds as well. There is no evidence from decades of field observations that they feed on fish. Resident killer whales prey mainly on fish, particularly salmon, and some squid. Offshore killer whales are poorly known, but appear to also be fish feeders. Neither residents nor offshores have been observed to prey on marine mammals. These foraging specializations appear to be fixed behavioural traits maintained by cultural transmission within populations.

Three putative populations of transient killer whales have been described from studies in nearshore waters of the northeastern Pacific. These are the so-called *West Coast transients*, distributed along the west coast of the mainland US and Canada, the *AT1 transients*, centred in Prince William Sound and Kenai Fjords, Alaska, and the *Gulf of Alaska transients,* usually sighted in waters of the central and western portion of the Gulf of Alaska (Ford and Ellis 1999; Matkin et al. 1999; Angliss and Outlaw 2007). The AT1 population has declined sharply in recent years and is believed to be comprised of only 8 animals (Saulitis et al. 2005). The Gulf of Alaska transient population numbers at least 314 individuals, and are most reliably seen between southeastern and western Alaska (Angliss and Outlaw 2007).

The West Coast Transient (WCT) population is the only one known to frequent Canadian waters, and is the focus of this RPA. This population is distributed throughout coastal waters of British Columbia. In 1999, this population was estimated from long-term photo identification studies to contain 219 whales, though long gaps between sightings of some individuals added considerable uncertainty to this estimate (Ford and Ellis 1999). An additional 100 or so transient killer whales identified off central California (Black et al. 1997) were in the past considered to be an extension of this population because of acoustical similarities and occasional mixing with WCT individuals in BC waters (Ford and Ellis 1999). However, a recent reassessment indicated that the available evidence was insufficient to warrant inclusion of these whales in the WCT population (Fisheries and Oceans Canada 2007). This is also the case for Gulf of Alaska transients, which are seen occasionally within the range of WCTs but have only been observed to travel in association with WCTs on one occasion.

Killer whales are long lived animals that have a low reproductive potential. Although few life history parameters are yet available for transient killer whales, values for resident killer whales presented in Olesiuk et al. (2005) may be generally representative. Survival patterns are typical of mammals, being ∪-shaped with highest mortality rates in very young (neonate) and very old age classes. Survival rates of juveniles and adults are high (0.97-0.99), particularly among mature females and during periods of population growth. During a period of growth in the northern resident killer whale population, females had a mean life expectancy (at age 0.5 yr) of 46 yrs and a maximum longevity of about 80 yrs. Males had a mean life expectancy of 31 yrs, with maximum longevities of 60-70 yrs.

Females give birth to their first viable calf at approximately 14 yrs, and produce an average of 4.7 calves over a 24-yr reproductive lifespan. Gestation is relatively long at 16-17 months, and the minimum calving interval is about 3 yrs (mean = 4.9 yrs). Females give birth to their last calf at around 40 yrs, then become reproductively senescent for the remainder of their lives. Calving is diffusely seasonal, with a peak in fall and winter.

Killer whales tend to live in long-term matrilineal groups (Bigg et al. 1990; Ford et al. 2000). In resident killer whales, social structure is extremely stable, as there is no dispersal from the natal group by either sex. Thus, the basic social unit, known as a *matriline*, can be comprised of up to 4 generations of whales, generally a post-reproductive female matriarch and her living descendants. The typical travelling unit of resident killer whales is the *pod*, which may consist of a single matriline or several closely-related matrilines that spend most of their time in close association. Each pod has a distinctive set of stereotyped underwater calls, or dialect, that is maintained by cultural transmission. Dialect similarities reflect pod genealogies and likely serve as acoustic ‘family badges’ that promote group cohesion and integrity. Pods often join to form short-term associations lasting several days or more, especially during the summer months when prey is abundant. Mating takes place between pods, particularly those that are acoustically dissimilar (Barrett-Lennard 2000). Dialects may thus serve as an inbreeding avoidance mechanism in resident killer whales.

Transient killer whale society is matrilineally based, but is considerably more dynamic than that of residents. A key difference is the common dispersal of individuals from the natal matriline in transients (Baird and Dill 1996; Ford and Ellis 1999). Once dispersed, whales will travel with other transient groups for variable periods, and may only associate with their natal matriline occasionally. As a result of dispersal, transient matrilines tend to be smaller than those of residents, and long-term associations of closely-related matrilines equivalent to resident pods do not exist. Typical group sizes of transients are 3-6 individuals, though temporary associations of up to 25 whales have been observed. Transients are acoustically quiet compared to residents, probably because their hunting strategy relies on stealthy approaches to unwary marine mammals. When vocal, WCTs share a common set of distinct stereotyped calls that do not have dialect variations as seen in residents.

HABITAT AND ECOLOGY

Transient killer whales range widely in coastal waters of the northeast Pacific. They occur along the exposed outer coast as well as in protected inshore channels, straits, and inlets. They are most commonly seen within 10 km of the coast, but this may reflect a strong bias in sighting effort in nearshore waters. Transients have been encountered up to 30 km offshore during shipboard cetacean surveys (DFO-Cetacean Research Program, unpubl. data). They often swim very close along shorelines and around rocky reefs and islets, particularly while foraging for their most common prey species, the harbour seal (*Phoca vitulina*). They also can be found in open water areas, where they forage for small cetaceans and pinnipeds. Because their hunting strategy depends on stealth and surprise attack, transients seldom spend long in any single location. Transients are non-migratory and can be found throughout their range in any month of the year. However, there is some seasonal trends in their movement patterns that may coincide with availability of prey (Ford and Ellis 1999).

Transient killer whales prey on a variety of marine mammal species. Along the coast of British Columbia and southeastern Alaska, their primary prey is the harbour seal, which represented roughly one-half of observed kills and harassments (unsuccessful attacks) documented by Ford et al. (1998). Other important prey species included Steller sea lions *Eumetopias jubatus*, Dall’s porpoise *Phocoenoides dalli*, and harbour porpoise *Phocoena phocoena*. There was no apparent specialization by different transient groups on particular prey species or types (i.e., pinnipeds versus small cetaceans). Minke whales *Balaenoptera acutorostrata* are occasionally attacked and killed by transient killer whales, as are calves of grey whales *Eschrichtius robustus* (Ford and Ellis 1999; Ford et al. 2005). Various species of seabirds have been observed to be harassed or killed, but they are seldom eaten and do not represent a significant part of transient diet (Ford et al. 1998).

RECOVERY POTENTIAL ASSESSMENT

Phase I: Assess Current/Recent Species Status

West Coast Transient killer whales have been studied by means of photographic identification of individuals continuously since the early 1970s (Bigg et al. 1987, 1990; Ford and Ellis 1999). This long-term effort has resulted in an archive of identification photographs collected from 1971 sightings of WCT whales by over 100 collaborators between 1958 and 2006. This archive, maintained by the Cetacean Research Program (CRP) at the Pacific Biological Station (Fisheries and Oceans Canada, Nanaimo, B.C.), has resulted in a database containing 8155 confirmed individual whale identifications made from these photographs. Almost four hundred (398) unique identification numbers have been assigned. However, to define the study population, we identified a social network of 308 associated whales that have been identified repeatedly within the coastal study population. Specifically, each of these whales associated more than once with other whales in this network. This association rule eliminated 90 whales that were seen in 24 sightings (primarily at the periphery of the study area), but did not associate repeatedly with the WCT network (Figure 1). Some of these whales were known to be more regularly sighted in adjacent populations in California (18 CA whales, Black et al. 1997) and Alaska (12 AT whales; Matkin et al. 1999). Of the 308 whales, 155 (50%) were calves born during the study period that could be assigned to a known WCT mother. Figure 2 shows the rate of discovery of new non-calf individuals in the study population. The curve has reached an asymptote, with no new non-calf whales being identified in the last 6 years.

Capture-Recapture Population Modeling

To assess population abundance and dynamics more formally, we adopted a capture-recapture (or mark-recapture) approach, treating photographic identifications and re-identifications as “captures” and “recaptures” to which analytical techniques can be applied (Hammond 1986, 1987 1990). Our aim was to ensure that the included data conformed to the assumptions of the statistical models as closely as possible, in order to minimize bias in parameter estimates (Seber 1982; Pollock 2000). This is particularly important for populations that are sampled using non-conventional capture-recapture approaches. Additionally, we not only estimated the most likely value for these parameters, but also fully documented the associated uncertainty by estimating probability distributions through a Bayesian approach to model fitting and statistical inference (e.g. Gelman et al. 1995; Brooks 2000).

Capture-recapture models require sampling to be “instantaneous” so that demographic changes do not take place within the sampling intervals. Furthermore, to estimate meaningful trends in annual demographic rates we identified a consistent and relatively short annual sampling interval of five months between May and October. Over 75% (1479 out of 1947) of the sightings occurred within this time period, and 302 out of the 308 WCTs were identified in summer intervals. The dataset was further restricted to the years between 1974 and 2006, with multiple sightings within the summer interval in each of these years, comprising a total of 6194 individual identifications. The number of individual whales identified in each summer interval increased over the study period, from a minimum of 6 in 1977 to a maximum of 164 in 2005.

Individual whales were seen in an average (median) of 6 different annual intervals, with a range of 1 to 29 years. However, 16 animals had gaps in their photographic histories, with absences from the dataset for at least 10 consecutive summer intervals before re-identification, with a maximum gap of 17 years. These absences from the record imply periods of temporary emigration beyond the study area, as it is unlikely that whales would simply go undocumented in as many as 10 years if they were present. However, it should be noted that while this is likely due in part to the movement patterns of whales, it may also reflect a degree of shrinkage and expansion of the effective study area due to changes in the level and geographical coverage of the photo-identification effort. Either way, these gaps highlight the need to allow for temporary emigration in the capture-recapture modeling.

The popular Cormack Jolly Seber model for estimating survival (Lebreton et al. 1993) does not allow animals to emigrate from the study area and return later. Instead, we follow the lead of Whitehead (1990) in developing a capture-recapture model that parameterises emigration and re-immigration, in addition to survival. Our model is based on an individual-specific factorization similar in concept to that of Dupuis (1995, 2002). This factorization allows “realistically complex” models for the capture probability to be developed, that can be thought of as the product of conditional distributions. Specifically, the model could be separated into conditional distributions for capture, availability for capture (temporary emigration), and death, and each of these components of the model could then be developed to approximate key features of the data and study population.

Starting from the annual interval when whale *i* was first identified, we denote by O_{ij} whether the whale was captured (i.e. photographically identified) in interval t_j (1 for identified, 0 for not identified) and by X_{ij} whether the whale was known to be alive (1 for alive, 0 for dead). Values of 0s were inputted for X following the known death of three whales: two live-captured and removed in 1976 and one recovered dead-stranded in 1979. Similarly, values of 1 were inputted for X for years when the whale was not observed (O = 0) between years of repeated captures (O = 1). Where the status was unknown, typically following the interval of last capture, we treated X_{ij} as a missing value about which inference may be made. Similarly, we defined a third matrix F, denoting whether the whale was available to be captured within the study area ($F_{ij} = 1$) or outside the study area and unavailable to be captured ($F_{ij} = 0$). When an individual was actually observed then $F_{ij} = O_{ij} = 1$. However when the individual was not observed then the availability status was unknown and F_{ij} was treated as a missing value.

The model had parameters ($\varphi_{ij}, p_{ij}, \lambda_{ij}, \mu_{ij}$), where φ_{ij-1} is the probability that whale *i* survives from time t_{j-1} to time t_j given that it was alive at time t_{j-1}; λ_{ij-1} is the probability that whale *i* emigrates from the study area at time t_{j-1} given that it was alive at time t_{j-1} ; μ_{ij} is the probability that whale *i* re-immigrates back into the study area at time t_j given that it was alive at time t_j; *and* p_{ij} is the probability that whale *i* is captured at time t_j given that it was alive AND available to be captured at time t_j. For whale *i*, the sequence X_{ij} is modeled as a Markov chain with $Pr\{X_{ij} = 1 \mid X_{ij-1} = 1\} = \varphi_{ij-1}$. Similarly the sequence F_{ij} is modeled as a Markov chain with $Pr\{F_{ij} = 1 \mid F_{ij-1} = 1\} = (1 - \lambda_{ij-1})$ and $Pr\{F_{ij} = 1 \mid F_{ij-1} = 0\} = \mu_{ij}$. Consequently, given the whale is alive at time t_{j-1}, X_{ij} has a Bernoulli distribution with parameter $X_{ij-1}.\varphi_{ij-1}$; given the whale is alive and available to be captured at time t_{j-1}, F_{ij} has a Bernoulli distribution with parameter $F_{ij-1}.(1 - \lambda_{ij-1})$, and given the whale is alive and unavailable to be captured at time t_{j-1}, F_{ij} has a Bernoulli distribution with parameter $F_{ij-1}.\ \mu_{ij}$. Then, the observation point O_{ij} has a Bernoulli distribution with parameter $Xij.\ Fij.\ pij$, thus conditionally linking all parameters to the observed data.

In order to assess trends in recruitment, as well as survival, we adopted the approach of Pradel (1996) to condition on the time of last observation for each whale, and read the identification histories backwards. This induces a simple “duality” relationship between the survival and recruitment analyses, in which the natural counterpart of the survival probability is the seniority probability, γ_{ij} or the probability that individual i present at interval j was already present in the population at time $j-1$. The recruitment analysis of the dataset was identical to the survival analysis of its dual (reverse) data set. To simplify the model and facilitate model fitting, we initially assumed common probabilities of survival, recruitment, movement and capture across individuals so that $\varphi_{ij}=\varphi_j$, $p_{ij}= p_j$, $\lambda_{ij} = \lambda_j$, $\mu_{ij} = \mu_j$, and $\gamma_{ij} = \gamma_j$. However, as an extension we also fit the model with separate probabilities of survival and recruitment for 5 age / sex classes, so $\varphi_{ij}=\varphi_{aj}$ and $\gamma_{ij} = \gamma_{aj}$. Each individual i was linked to one of 5 age / sex classes a through a matrix A which specified the age / sex class a_{ij} for each individual i in each year j. Here, $a = 1$ represented newborn calves in their first year; $a = 2$ depicted juveniles, defined as females prior to birth of their first viable calf, or males that had a dorsal fin height:width ratio less than 1.4; $a = 3$ represented adult females after the age of producing their first viable calf; $a = 4$ represented sub-adult males with a fin height:width ratio >1.4 but with the fin not yet fully developed, that were likely sexually but not physically mature; and $a = 5$ depicted fully mature adult males.

Under the Bayesian approach to inference, we must specify prior distributions for all model parameters. We adopted Beta prior distributions for all model parameters so that probability mass was constrained to lie between 0 and 1. Specifically, we followed Kass and Raftery (1995) in specifying a separate Beta (b_1, b_2) hierarchical distribution for each set of parameters (e.g. one prior for each of φ, p, λ, μ and γ) where $b_1 = \xi/\omega$ and $b_2 = (1-\xi)/\omega$. This distribution was thus determined by two hyper-parameters: ξ represented the mean value across each set of parameters and ω represented the year to year variability over the set. Flat prior distributions were placed on each of these hyper-parameters by assigning Beta(1,1) prior distributions, in order to learn about the mean and variability across each set. This hierarchical prior allowed years in the set with relatively informative data to contribute largely towards the mean, and estimates from sparse data years were drawn towards the overall mean. This had the effect of smoothing estimates across each set so that notable variability from the mean was detected, but there was “borrowing strength” across each set to allow for more precise estimates in sparse data years. For the age / sex stratification of parameters, a separate Beta prior was specified for each age / sex class for each parameter to reveal key differences.

Once these priors had been assigned, the model could be thought of as a full probability model, which described a series of conditional probability distributions for the unknown parameters and missing data components conditional the observed data. Missing data are thus treated the same as the other unknown parameters, and updated based on the observed data. We used the freely available WinBUGS software (Lunn *et al.* 2000) to implement Markov Chain Monte Carlo (MCMC) sampling to make repeated draws from the conditional distributions. In particular, we used Gibbs sampling, which is a well known technique for generating samples from multivariate distributions (Brooks 1998). Gibbs sampling is used to update each variable in turn from its conditional or “posterior” distribution given all other variables in the system, and the entire distribution is explored as the number of Gibbs sweeps grows large. We sampled 10,000 values from the posterior distribution of each parameter, after discarding a burn-in of 10,000 values. The sampled values were then used to estimate summary statistics for the posterior distributions.

MCMC approaches can similarly be used to sample from the posterior distribution of quantities that can be derived as functions of parameters. For example, by estimating both recruitment parameters and survival parameters in a single MCMC run, we were able to estimate the posterior for derived population growth rates $\rho_j = \varphi_{ij-1} / \gamma_{ij} = \gamma_{aj}$ (Pradel, 1996). This approach allows for an estimate of population change that does not require the calculation of specific abundance estimates. However, we can also use estimates of the capture probabilities p_j and the movement probabilities λ_j and μ_j to estimate both the abundance *in* the study area N_j in any given interval j and the size of the parent population M_j of animals that are alive and use the study area over the course of the study, but may not all enter the study area during interval j (following Whitehead 1990). These parameters were linked to the observed data by specifying the number of individuals actually observed in the study area n_j as a Binomial sample from the study area abundance N_j with the Binomial proportion given by p_j. However, N_j was itself modeled as a Poisson random variable, with the Poisson mean given by m_j / π_j, where $\pi_j = (\lambda_j + \mu_j) / \mu_j$ was a derived parameter representing the ratio of the parent population size compared to the abundance of individuals in the study area. M_j was itself assigned a Poisson prior distribution with its mean m_j initially assigned a Gamma(0.001, 0.001) distribution where G(g_1, g_2) denotes an inverse gamma distribution with mean g_1 / g_2 and variance g_1 / g_2^2. This Gamma prior constrained values to be positive, but was essentially flat over a wide region encompassing the likely values. By placing a two stage prior on M_j, with the Poisson mean itself being treated as an unknown random variable, the prior distribution of M_j was therefore the negative binomial, allowing for over-dispersion relative to the Poisson distribution (Madigan and York, 1997).

Modeling Population Dynamics

The estimates of the population size, M_j, for each year j did not take into account the estimates for adjacent years. As an alternative, we investigated if we could gain information about the population by modeling the changes in the abundance estimates, and making inference about model parameters. This was accomplished by adopting a hierarchical prior density for the Poisson means $m_j \sim G(\theta_j \delta, \delta)$, so that the mean of m_j was given by a model θ_j. However, we did not assume that the m_j (and thus M_j) fell exactly on the model trajectory, or had a common variance. By including the additional parameter δ in the gamma mixture density for m_j we allowed additional variability (over dispersion) compared to fitting a model with more restrictive Poisson errors. This allowed for estimates of abundance that show greater variance than trend variance alone, and the marginal variance of M_j thus became $\mathrm{Var}(P_t) = \theta_j + \theta_j / \delta$. Smaller values of δ indicated more over dispersion relative to the Poisson assumption that $\mathrm{Var}(P_t) = m_t$. We set a Gamma(0.1,0.1) prior on δ to allow for the inclusion of the extra-Poisson variability.

We used the “stochastic logistic” model structure (Dennis and Taper, 1994) for θ_j, where the population size in year j was modeled as a function of population size in year $j-1$, with this function being characterized by an intrinsic growth rate β_0 that was modified by a density mediated effect β_1:

$$\log(\theta_j) <- \log(M_{j-1}) + \beta_0 + \beta_1(M_{j-1})$$

The parameters β_0 and β_1 are constants in the model, such that $\beta_1 < 0$ indicates density dependence in population growth with the realized growth rate decreasing as the population state M_{j-1} becomes larger, $\beta_1 > 0$ indicates inverse density dependence, and β_1

= 0 indicates no density-dependence. We assigned Normal(0, 100) prior distributions with high variance to the hyper-parameters $\beta_0 + \beta_{1.}$ Rather than perform this trend component independently of the annual capture-recapture estimation in the sampling component, we combined these two components into a single MCMC sampling from a linked Bayesian full probability model, so that uncertainty could effectively be propagated through the conditional distributions. We were particularly interested in using Bayesian inference to examine for density dependence in population growth, allowing for the estimation of an equilibrium population size that could be interpreted as the current carrying capacity for this population. If negative, the parameter β_1 would impart ergodic behaviour to the model: large populations would decline towards the equilibrium point K, and small populations tend to increase towards K (Dennis and Taper, 1994). We estimated K from a function of the estimated intrinsic growth rate and density-dependent parameter, $K = -\beta_0 / \beta_1$.

Results of Population Modeling

Survival

The survival probability, pooled across all individuals was remarkably high, and relatively stable, across the study period. The average annual survival probability had an estimated median of 0.98 (95% probability interval = 0.95-0.99) (Figure 3). The estimate of survival probably was relatively imprecise during the first half of the time series, due to small sample sizes. However, higher capture probabilities over the latter part of the time series led to more precise estimates.

Recruitment

Estimates of seniority probabilities (the probability of being in the population in the previous year, or the inverse of recruitment) were similarly imprecise at the start of the time series. However the seniority probabilities appear to have increased over time, with a relatively low level (high recruitment) for the first half of the time series and a relatively high level (low recruitment) at the second half of the time series (Figure 4). This increase can be seen in the average estimates for seniority over the four 8-year increments, starting in 1975: 0.91 (95%PI = 0.66-0.99), 0.91 (0.80-0.96), 0.96 (0.90-0.99) and 0.97 (0.93-0.99); with corresponding decreasing recruitment probabilities of 0.09, 0.09, 0.04 and 0.03. During the final 6 years, when only new calves were documented (Figure 2), the average recruitment was 0.03 (0.01-0.07).

Sex specific survival / recruitment

Estimates of both survival were relatively imprecise for calves and sub-adult males due to small sample sizes. Nonetheless, survival was notably high and stable for all age / sex classes (Figure 5). Estimates of average survival show the highest rate for adult females (0.98, 0.97-0.99) and juveniles (0.98, 0.95-0.99), with a slightly lower rate but similar rate for adult males (0.97, 0.94-0.98). The average survival rate was lower for sub-adult males (0.95, 0.87-0.98) and calves (0.92, 0.82-0.97).

Age / sex specific seniority estimates were also relatively imprecise for the smaller sample of calves and subadult males. However, it is clear that there was negligible recruitment of new non-calves into the population during the second half of the time series, with almost all the additions coming as first year calves (Figure 5). However, there is some evidence of a small level of recruitment for other age classes during the first half of the time series, indicating the recruitment of new non-calves into the population. This could reflect movement of non-calf individuals into the study area. However, this may partly reflect the discovery of new individuals in the population with increasing geographical coverage over time.

Population growth

High survival and recruitment has led to a growing population. The average annual population growth rate was 1.06 (0.99-1.23), indicating growth of 6% per year. However, there is also evidence of a decline in population growth, mirroring the decline in recruitment during the second half of this time series (Figure 6). This can be seen in the average estimates for growth over the four 8-year increments, starting in 1975: 1.08 (0.96-1.50); 1.11 (1.02-1.29); 1.02 (0.98-1.10); and 1.02 (0.98-1.07). Growth at the start of the time series was therefore rapid, tracking relatively high rates of recruitment. However, growth was typically around 2% in the final half of the time series, driven almost entirely by the production of new calves. Interestingly, 3 of the last 8 years had a posterior median estimate of growth rate that was actually less than 1 (i.e., population decline). This decreasing growth rate corresponds to increasing seniority (decreasing recruitment) at the end of the time series.

Population abundance

Abundance estimates produced from capture probabilities were consistent with these estimates of population growth rate derived from demographics, showing a growing population but a decreasing rate of population growth (Figure 7). Population size was initially very low (posterior median = 24, 95% probability interval = 11-53) in 1974, grew rapidly for the first half of the time series, but growth appears to have slowed towards the end of the series. This can again be seen in the average estimates for the parent population size M_j over the four 8-year increments: 34 (13-80); 84 (52-146); 169 (120-253); and 210 (155-289). The population estimate at the end of the time series was 243 (95% probability interval = 180-339) in 2006.

We described population growth using the stochastic logistic model, which assumes the rate of population growth decreases linearly with increasing abundance. Our analysis of population growth from demography supports this assumption of an approximately linear decrease in growth rate, and the SL model fits the abundance estimates extremely well (Figure 7). All the confidence intervals from the abundance estimates overlap with the median population trajectory predicted by the model. Furthermore, following years with unusually high estimated abundance (e.g. 1996) the subsequent abundance estimate was considerably lower, as predicted by the model. Based on this model fit, there was an extremely high probability (>0.99) of a density-dependent decrease in population growth, with 100% of the posterior estimate for β_1 sampled from values below zero. This density-dependent model defined an equilibrium point of K = 262 whales, which may be interpreted as the current carrying capacity of this system. At the end of the time series the population appeared to be close to reaching this equilibrium point.

Temporary emigration

Long-term gaps in the capture histories suggest that whales are not limited to the coastal study area. To support this, we have produced non-negligible estimates of temporary emigration in every year of the study (Figure 8). Temporary emigration away from the study area appears particularly high during the early part of the time series, but this may also reflect a limitation in the extent of the effective study area due to geographically-limited survey effort. The average estimate of temporary emigration was 0.20 (0.03-0.50) during the first 8 years of the study, and was lower for the next two 8-year intervals: 0.07 (0.01-0.21) and 0.09 (0.03-0.19); and lowest for the final interval 0.04 (0.01-0.12), implying higher fidelity to the study site or more complete coverage of the site.

Estimates of the re-immigration rates of temporary emigrants were high throughout the study period, indicating a typically fast return to the study area (Figure 9). The average re-immigration rate was 0.25 (0.20-0.30), and there was limited variability in the average re-immigration rates for the 8-year intervals: 0.24 (0.10-0.46); 0.21 (0.09-0.38); 0.25 (0.12-0.42); and 0.24 (0.12-0.42). Note that re-immigration was therefore still high at the end of the time series when emigration was at its lowest. This implies a high return rate of the few whales that did appear to emigrate from the study area at the end of the time series, indicating a much higher degree of site fidelity at the end of the time series.

Estimates of the derived parameter $\pi_j = (\lambda_j + \mu_j) / \mu_j$ varied from 1.02 to 3.25 with an average of 1.40 (1.07-2.44), indicating that on average there was more than two-thirds of the population using the study area during an annual interval. The ratio of abundance of the parent population size compared to the abundance of individuals in the study area was notably high at the start of the study, and low at the end, with 8-year averages of 1.79 (1.13-3.86); 1.31 (1.03-2.35), 1.34 (1.09-1.94); and 1.17 (1.03-1.59). This represented an increasing proportion of the population in the study area that reflects the decline in temporary emigration and high re-immigration. Note that this is a *proportional* increase in study area use, but because the parent population size also increased, there was a fairly consistent *number* of individuals outside of the study area in any given year (Figure 10).

Evaluating Possible Sampling Biases

To evaluate if the population growth (particularly during the early part of the time series) was due in part to the discovery of new animals inherent in a developing capture-recapture study, we re-started the time series in 1984. We reconstructed identification and survival histories, omitting any observations or survival data from the first ten years of the study, and naively ran the model using this truncated data. The estimates of population size were remarkably similar to the estimates from the full data set (Figure 11), supporting real population growth.

To evaluate if population growth was due in part to increasing coverage of some (extreme) portions of the study area (which may have increased the available portion of the study population), we further sub-sampled the dataset to a restricted area with close to constant effort. This was DFO's Pacific Fisheries Management Area 12, an area off the northeastern coast of Vancouver Island, including Queen Charlotte and western Johnstone straits, that has been consistently surveyed by one of the authors (GE) for the full duration of this study. The population estimates for the pool of whales using this area are smaller than for the full study area (Figure 11), which is to be expected given the limited size of this restricted area. Nonetheless, these estimates show a similar pattern of population growth, which cannot be attributed to changes in the size of the effective study area, thus supporting growth of the population using the full study area.

Population Abundance and Distribution Targets

An overall recovery goal and set of specific recovery objectives were developed in the draft Transient Killer Whale Recovery Strategy (Fisheries and Oceans Canada 2007). Without any objective means of estimating historical abundance, carrying capacity or biological limiting factors for WCTs, it was not possible to set a quantitative population abundance target for recovery. Instead, the goal of the Recovery Strategy is:

> *To attain long-term viability of the West Coast transient killer whale population by providing the conditions necessary to preserve the population's reproductive potential, genetic variation, and cultural continuity.*

Specific population and distribution objectives in the Recovery Strategy include the following:

- The population size, averaged over the next five years, will remain at or above the current level.
- The number of breeding females in the population, averaged over the next five years, will remain at levels that will provide a neutral or positive growth rate.
- Studies will be undertaken to determine numerical and demographic population objectives that represent long-term viability for this population.
- Transient killer whales will continue to utilize their known range.
- Prey will be available, in quantities adequate to support recovery, throughout the currently known range of transient killer whales.

Being the top marine predator, mammal-hunting killer whales are ultimately limited by prey availability. As the population analyses described above indicate, the WCT population has grown rapidly over the last three decades. Although more recent growth can be attributed almost entirely to the recruitment of new calves to known females in the population, the rapid growth during the first half of the time series also reflects recruitment of non-calf individuals. Population growth of WCTs has clearly been limited at increasing densities, and is levelling off as the population approaches an equilibrium point of around 262 whales. This reduction in growth is primarily due to a reduction in recruitment. This population regulation implies resource limitation and suggests that carrying capacity for transients in the study area may be in the mid to high 200s, given the current densities of marine mammal prey.

The diet of transient killer whales consists of a mixture of pinniped and small cetacean prey species. The proportion of different prey species observed in the diet of WCTs over the study period is shown in Figure 12. Approximately two-thirds of prey items consumed by WCTs were pinnipeds, and one-third were cetaceans. Pinnipeds were represented primarily by harbour seals (54% of total kills) and Steller sea lions (12%), and cetaceans primarily by harbour porpoise (15%), Dall's porpoise (10%) and Pacific white-sided dolphins (5%). There is no evidence of specialization by individuals or matrilines on particular prey species or types (e.g., pinnipeds vs cetaceans) within the WCT population (Ford et al. 1998).

The availability of pinniped prey for WCT killer whales has fluctuated dramatically over the past century. In British Columbia, intensive harvesting and culling between the late-1890s and late-1960s reduced harbour seal abundance to roughly 10% of historical levels (Olesiuk and Bigg 1989; Olesiuk 1999). Harbour seals were similarly depleted in Washington State (Jeffries et al. 2003). Steller sea lions populations were culled extensively in British Columbia between 1912 and the mid-1960s, which reduced abundance to about one-quarter of historical levels (Bigg 1985; Olesiuk and Trites 2003). Culls of this species also took place in Washington and Oregon waters (Pitcher et al. 2007). This depletion of these pinniped populations very likely had a major effect on the carrying capacity for transient killer whales, and may have resulted in reduced survival and/or emigration, leading to population decline.

The abundance of both harbour seals and Steller sea lions in the WCT range has increased dramatically since directed culls and harvesting of these species ended with protection provided by the Canadian Fisheries Act in 1970, and the U.S. Marine Mammal Protection Act in 1972. Harbour seal abundance in the well-surveyed Strait of Georgia, B.C., increased at 11.5% per annum during the 1970s and 1980s, then growth rates slowed and stabilized in the early to mid 1990s (Olesiuk 1999). A similar population trend was apparent in other regions of coastal British Columbia (Olesiuk 1999). Harbour seal abundance increased 7 to 10-fold during the same period in Washington, and stabilized by the mid 1990s (Jeffries et al. 2003). Harbour seal abundance has generally increased in southeastern Alaska, though declines have been observed in some areas (Small et al. 2003). The abundance of Steller sea lions in British Columbia and southeastern Alaska increased at an overall annual rate of 3.1% over the last 25 years, and has yet to show signs of stabilizing (Pitcher et al. 2007).

There is little information on trends in abundance of important cetacean prey of transient killer whales over the study period. Although there is some anecdotal evidence of a long-term decline in harbour porpoise abundance in industrialized inshore waters (Baird 2003), Dall's porpoise and Pacific white-sided dolphins appear to be common over much of the WCT range (Carretta et al. 2007; Angliss and Outlaw 2007). The distribution patterns of Dall's porpoise and Pacific white-sided dolphins can vary considerably from year to year due to changing oceanographic conditions (Carretta et al. 2007). Pacific white-sided dolphins were rarely seen in nearshore waters of British Columbia prior to the mid 1980s, but became abundant in this area during the late 1980s and 1990s. This species was first recorded as prey of WCT transients in 1995 (DFO-CRP unpubl. data).

Overall, it would appear that WCTs have benefitted primarily from the dramatic increase in the availability of pinniped prey since the early 1970s, particularly harbour seals. Although harbour seal population growth levelled off in the mid 1990s, the WCT population continued to increase into the early 2000s and has only recently begun to show signs of stabilizing. Although Steller sea lion abundance in the region continues to increase, it seems doubtful that this one prey species would be sufficient to drive a substantial population growth of WCTs.

Gaps in the capture histories and corresponding estimates of emigration and re-immigration probabilities provide evidence that members of the WCT population temporarily emigrate to waters outside the study area, although this temporary emigration appears to have decreased during the second half of the time series. Transient killer whales are mobile predators that are capable of ranging widely to sample their environment. Such movement could dramatically change the availability of different prey resources, and as such the carrying capacity for this population can also change quickly. This has likely happened in the past, and the dynamics that we have described here may be the response to movement into coastal waters, with prey preferences switching from offshore cetaceans and pinnipeds (e.g., northern fur seals *Callorhinus ursinus*) to coastal pinnipeds. Continued monitoring of prey preferences and ranging patterns is therefore required to understand the potential for population change.

Phase II: Scope for Management to Facilitate Recovery

Probability That Recovery Targets Can Be Achieved

If, as the evidence suggests, West Coast Transients are prey limited and are approaching the carrying capacity of their current range, there is limited capacity for future population growth without an increase in prey densities. As discussed above, there is no indication of an increasing trend in populations of the WCT's most important prey species, with the exception of the Steller sea lion. Provided there is no decline in prey availability or unexpected source of increased mortality, it can be anticipated that the population recovery target of the Transient Killer Whale Recovery Strategy – a stable abundance over the next five years – is achievable. Recruitment in the WCT population has slowed over the past decade, but continued monitoring will be necessary to determine if this trend continues and whether there may be future density-dependent decreases in survival.

Killer whales are long-lived, upper trophic-level predators with a very low reproductive potential. They also have a tendency to live in very small, behaviourally specialized and reproductively isolated populations. These factors make killer whale populations such as the WCTs highly vulnerable to even minor increases in levels of mortality.

Potential Sources of Human-Induced Mortality

Since the live-capture of killer whales for public display ended in the mid 1970s, there has been no recorded directed takes of transient killer whales in the region. Indiscriminate shooting of killer whales, once common on the coast, now appears to be very rare (Ford et al. 2000). However, there are a variety of potential sources of mortality to West Coast Transient killer whales that could result from human activities. These threats are described in detail in the draft Recovery Strategy (Fisheries and Oceans Canada 2007), and are summarized below. It should be noted that some of these threats have not been demonstrated to be a direct cause of mortality in WCTs or killer whales generally, and may not represent a lethal risk on their own. However, it is possible for multiple stressors to act synergistically to cause stronger negative and possibly lethal effects. It should also be kept in mind that through much of the last three decades, the WCT population has been increasing at rapid rate, through immigration into the study area, recruitment of calves, and high survival rates. This indicates that no significant limiting factors have restrained population growth until recently, which may be density-dependent effects caused by resource limitation.

Contaminants

Killer whales in coastal waters of the northeastern Pacific carry significant concentrations of Persistent Bioaccumulating Toxins (PBTs) in their tissue. Of greatest concern are polychlorinated biphenyls (PCBs), which are found at extremely high concentrations in WCT killer whales due to their consumption of marine mammals that are already contaminated with PCBs (Ross et al. 2000, 2004). These compounds are not typically acutely toxic, but can potentially have chronic, slow-acting effects as ‘hormone mimics’ or ‘endocrine disruptors’. Although health effects have not been demonstrated in killer whales, high levels of PCBs in harbour seals have been associated with immunosuppression and endocrine disruption (Mos et al. 2006). Although PCB levels are declining in the environment, recent models suggest that it will take decades before PCB levels in killer whales decline below the thresholds for adverse effects seen in other species (Hickie et al. 2007).

Also of concern are rapidly increasing levels of polybrominated diphenyl ethers (PBDEs), which have recently become widely used as flame retardants in a variety of products. As with PCBs, the potential direct effect of PBDEs on transient killer whale health is not clear, there is growing evidence of endocrine disruption and immunotoxicity in other species (Fisheries and Oceans Canada 2007).

Biological Pollutants

Transient killer whales may be at heightened risk to the impacts of exotic diseases or ‘biological pollution’ due to their consumption of marine mammal prey. Transients may be exposed to pathogens such as viruses and bacteria that are endemic to their mammalian prey or from terrestrial sources, such as domestic pets or livestock. Biological pollutants may have direct effects by causing disease in transient killer whales, which may be predisposed to increased risk or severity of infection due to the immunotoxic nature of PBTs found at high levels in transients. Biological pollutants and pathogens could also affect transients indirectly through mortality and subsequent reduced availability of prey species. For example, a widespread epidemic of *Morbillivirus* among harbour seals, such as that which caused mass mortalities of seals in northwestern Europe, could have serious consequences for WCT killer whales (Fisheries and Oceans Canada 2007).

Acoustic Disturbance

There has been increasing concern in recent years about the potential effects of underwater noise on cetaceans. Acoustic disturbance can be of two types: chronic and acute. Chronic noise is primarily associated with motorized vessel traffic of all types, from commercial shipping to whale watching. Chronic noise can result in masking of communication signals used for social contact or behavioural coordination, or interfere with echolocation signals used for navigation and discrimination. Transient killer whales often forage in silence and may rely on passive listening to locate their prey (Barrett-Lennard et al. 1996). Masking effects of increasing background noise could thus reduce their foraging efficiency.

Sources of acute noise include military and commercial sonars, airguns used in seismic surveys, and underwater explosions usually associated with construction. These sounds can be extremely intense and may travel large distances underwater. Loud acute noises have the potential to cause a variety of effects in cetaceans, including hearing threshold shifts, production of stress hormones, and tissue damage, as well as a variety of behavioural responses. Although there is no direct evidence of the effects of high intensity sounds on transient killer whales, by inference from other cetacean species, detrimental effects might be expected.

Physical Disturbance

Vessels moving in close proximity have the potential to affect transient killer whales by disrupting behaviours. Although no studies have focused on transients specifically, resident killer whales have been shown to alter their swimming behaviour when approached by boats (Williams et al. 2002). With the increased intensity of whale watching activity in the vicinity of WCT killer whales in some areas, there is a potential for vessels to disrupt hunting behaviour, thereby reducing overall foraging success. Transient attacks on marine mammals are often protracted and involve energetic, high-speed swimming, and vessels in close proximity can cause the whales to abandon their attack, or provide the prey item with a refuge to escape from the attacking whales.

Collision with Vessels

Killer whales are at some risk of injury or mortality as a result of being struck by boats or ships. Although there are no reported cases of transient killer whales being struck by vessels, four such incidents involving resident killer whales have been documented in recent years, two of which were fatal (DFO-CRP unpubl. data). It is not clear whether differences in swimming and diving patterns between the two types of killer whales make transients more or less vulnerable than resident killer whales.

Toxic spills

There is evidence that killer whales do not avoid toxic spills, as indicated by the behaviour of both residents and transients during the Exxon Valdez oil spill in 1989 in Prince William Sound, Alaska (Matkin et al. 1999), and of residents during a recent diesel spill in August, 2007, in Robson Bight, B.C. In the case of the Exxon Valdez incident, exposure to oil was associated with unprecedented mortality of both transient and resident whales, which probably died from inhalation of toxic petroleum vapours (Matkin et al. 1999). Expanded oil tanker traffic or hydrocarbon exploration and extraction off the west coast would increase the risk of toxic spills and potential injury or mortality to WCT whales.

Changes in Prey Availability

Transient killer whales are marine-mammal specialist predators, and potentially could be affected by major changes in prey availability. In western Alaska, there have been sharp declines in abundance of harbour seals, sea lions, and fur seals, and it has been hypothesized that these declines caused transient killer whales in that region to switch to sea otters, a less desirable prey species (Estes et al. 1998). There is no information on abundance or trends of transients in western Alaska to determine whether this reduction in availability of preferred prey had a significant effect on survival or productivity. It is noteworthy that a sudden decline of chinook salmon, the primary prey species of resident killer whales in British Columbia and Washington state, was strongly correlated with a dramatic increase in mortality rates in two separate killer whale populations in the region, northern and southern residents (Ford et al. 2005).

Because of their reliance on highly-specialized foraging strategies that are maintained by social learning and cultural transmission across generations, killer whales likely have a limited ability to shift to efficient hunting of novel prey species. Because WCT killer whales feed on a variety of different marine mammal species, they may not be vulnerable to minor fluctuations in abundance of a particular prey species. However, a wide-scale decline in multiple prey species, as took place in western Alaska, could have significant consequences for WCTs. Marine mammal prey species in the range of WCTs are currently subjected to relatively low levels of human-related mortality, and no significant changes in such mortality rates are anticipated. However, a major change in the marine ecosystem structure off the west coast, possibly resulting from over-harvesting of fish stocks, could affect transients indirectly through effects on their prey.

Maximum sustainable human-induced mortality

Due to the small population size of West Coast Transients, any human-induced mortality may be a cause for concern. In order to estimate the level of human-caused mortality that may be allowable without causing serious population-level consequences or prevent recovery, the U.S. National Marine Fisheries Service has devised a means of calculating the Potential Biological Removal (*PBR*) for marine mammal populations. *PBR* estimates the maximum number of animals, excluding natural mortality, that may be removed per year while still allowing the population to reach or sustain to its ‘optimum sustainable population’ (Wade 1998). *PBR* is calculated as:

$$PBR = N_{\mathrm{MIN}} \frac{1}{2} R_{\mathrm{MAX}} F_R$$

where:

N_{MIN} = the minimum population estimate
$\frac{1}{2} R_{\mathrm{MAX}}$ = one-half the maximum theoretical or estimated net productivity of the stock at a small population size,
F_R = a recovery factor between 0.1 and 1.

To calculate *PBR* for West Coast Transient killer whales, we use the following values:

N_{MIN} = 214, the 20th percentile of estimated population size in 2006
R_{MAX} = 0.04, the default value recommended for cetaceans (Wade 1998), and
F_R = 0.5, the recommended recovery factor for non-endangered cetaceans (Wade 1998; Angliss and Outlaw 2007).

PBR for the West Coast Transient killer whale population was calculated to be 2.14. Killer whales may have a lower maximum productivity potential than most cetaceans (Olesiuk et al. 2005), so an R_{MAX} of 0.03 is more realistic, resulting in a *PBR* = 1.60. It is clear from these calculations that the small WCT population could sustain very little human-induced mortality.

Habitat Quality and Quantity

West Coast Transient killer whales occupy a very extensive range. They travel widely throughout all coastal waters between about 47°N and 58°N latitudes, which corresponds to a straight-line distance of 1600 km along the west coast. Although the extent of their range in offshore waters is unknown, WCT whales have been encountered up to 40 km from shore. Transients spend the majority of their time foraging for marine mammal prey, which likely play a major role in determining movement patterns. WCTs may be found year-round in all parts of their overall range, but they rarely remain in any one location for long. Because their hunting strategy depends on stealth, it is likely more profitable to keep moving once potential prey in an area are alerted to their presence.

The great majority of WCT range is comprised of remote wilderness coastal areas that have received little impact from human activities.

If the recent stabilization in WCT abundance is a result of resource limitation, this implies that the existing habitat is sufficient to support only a population size in the low 200s.

Threats to Habitat

In this discussion, direct threats to individual whales, such as physical disturbance or vessel collisions, and threats to prey abundance, are differentiated from threats to habitat. As such, the main threats to WCT habitat are underwater noise and toxic spills. Chronic noise from shipping, for example, has the potential to interfere with foraging success by masking sounds needed to detect and localize prey, especially in confined waterways such as the Inside Passage. Highly industrialized and therefore noisy areas such as ports and harbours may be avoided by transient killer whales, but these are very small areas relative to the whales' entire range. High intensity acute noise from military sonar or seismic exploration could ensonify large areas of habitat, potentially disturbing whales or displacing them from important foraging areas.

Major toxic spills, including catastrophic oil spills, could cause extensive contamination of habitat, in addition to direct physical harm to transient killer whales. This risk can be anticipated to increase if oil tanker or barge traffic increases or oil exploration and extraction takes place in WCT habitat.

Other forms of pollution that may degrade habitat quality include wastewater effluent near urban areas, which may contain a variety of chemical pollutants, and pesticides and biological pollutants carried in runoff in agricultural areas.

Phase III: Scenarios for Mitigation and Alternatives to Activities

This section of the RPA is intended to provide an inventory of all feasible measures to minimize or mitigate the impacts of human activities on WCT killer whales and their habitat. Much of this is addressed in the draft Recovery Strategy for West Coast Transient Killer Whales (Fisheries and Oceans Canada 2007) and in a recent draft assessment of risk to critical habitat identified for resident killer whales (Lee et al. 2007). The following is extracted from the these draft documents and presented here for information. Note that these generally include only mitigation measures available in Canadian waters.

Noise Mitigation

Military sonar

The Department of National Defence (DND) has established protocols to protect marine mammals from disturbance and/or harm from the use of military active sonar. Maritime Command Order 46-13, for marine mammal mitigation, is to avoid transmission of sonar any time a marine mammal is observed within the defined mitigation avoidance zone, which is established specific to each type of sonar. Ship's personnel receive training in marine mammal identification and detection. All foreign vessels are subject to Canadian regulations while in Canadian waters (D. Freeman, DND, pers. comm.). There remains some concern regarding compliance by foreign vessels with Canadian regulations and the effectiveness of these mitigation protocols.

Seismic air guns

There are currently few industrial or scientific seismic surveys conducted in western Canadian waters. Some projects involving seismic surveying trigger screening under the Canadian Environmental Assessment Act (CEAA), while others are reviewed regionally by DFO. In 2005, DFO developed a draft Statement of Canadian Practice on the Mitigation of Seismic Noise in the Marine Environment (DFO 2005), to address concerns regarding the potential impact of seismic use on marine mammals and other marine life. A process for the revision and further consultation of the draft Statement is underway. In the Pacific Region, each proposed seismic survey is reviewed by DFO marine mammal experts and mitigation measures are developed based on the species of concern in the area of the survey for each project. Seismic mitigation protocols recommended by DFO Pacific Region are designed to prevent exposure of cetaceans to received sound pressure levels in excess of 160 dB re 1 μPa, which is generally the level at which behavioural disturbance can be anticipated. A slow ramp-up of air gun pressure, or a ‘soft start’, is utilized to allow cetaceans to leave the area ensonified with intense sound. A safety zone corresponding to the estimated 160 dB re 1 μPa isopleth is established around the sound source, and a marine mammal observer monitors this zone while air guns are operating. If a cetacean enters the safety zone, air gun use is suspended until it has left the zone.

While many seismic projects are screened prior to commencement, it is not clear that all projects are assessed for impacts to marine mammals prior to initiation of seismic activity. Also, even with a sound exposure mitigation protocol, transient killer whales are often difficult to detect by observers and thus may be unknowingly exposed to intense sound.

Construction noise

Mitigation protocols to prevent exposure of cetaceans to noise associated with construction activities such as dredging and pile driving in the Pacific Region are similar to those for seismic air guns.

Chronic noise

There is currently little mitigation of chronic noise in the marine environment that originates from shipping and other marine vessel traffic. Of particular concern is noise caused by whale-watching vessels that are frequently concentrated in prime transient killer whale foraging habitat (e.g., off Victoria, B.C.). However, whale-watching guidelines developed jointly by DFO and the US National Oceanic and Atmospheric Administration have served to restrict the distance and speeds at which boats can approach killer whales, which serves to reduce the level and extent of noise ensonification in the vicinity of whales.

Toxic Spills

The Transportation of Dangerous Goods Act regulates handling and transport of toxic substances within Canada, and numerous international, federal and provincial measures are in place for the prevention and management of toxic spills (e.g. Canadian/U.S. spill response plans for trans-boundary waters, Oil and Gas Operations

Act, BC EMA). Despite such regulation and preventative measures, spills are frequent along the coast of British Columbia, but most are very small and localized and do not present a major risk to WCT habitat.

Chemical Pollution

There are numerous national and international regulations and agreements that govern the manufacturing and application of many kinds of Persistent Bioaccumulating Toxins (PBTs), particularly the so-called legacy PBTs such as PCBs. The Stockholm Convention on persistent organic pollutants (POPs) and other UN Protocols aim to reduce global levels of legacy PBTs. Manufacture and availability of toxic chemicals in Canada are managed via listing under Schedule 1 of the CEPA and the BC Environmental Management Act (EMA) has regulations in place for management of contaminants in industrial and municipal effluents and outflows. The Fisheries Act (S. 36) prevents discharge of toxic substances into fish habitat(s), mitigating toxic threats to killer whale prey. Environment Canada is revising their proposed Risk Management Strategy for Polybrominated Diphenyl Ethers, under the Canadian Environmental Protection Act (CEPA). This strategy supports the ban of several (but not all) of the forms of PBDEs that are known to bioaccumulate in killer whales.

Regulations on manufacture of chemicals and vectors of contamination (e.g. sewage outflows) manage toxins in runoff in British Columbia. The BC Ministry of Environment’s storm-water planning, as well as non-governmental programs are in place for education on toxic runoff. For agriculture, the Fertilizers Act manages chemicals and the BC EMA Agricultural Waste Control regulation and Best Agricultural Waste Management Plans (BAWMPs) manage industry practices specifically.

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FIGURES

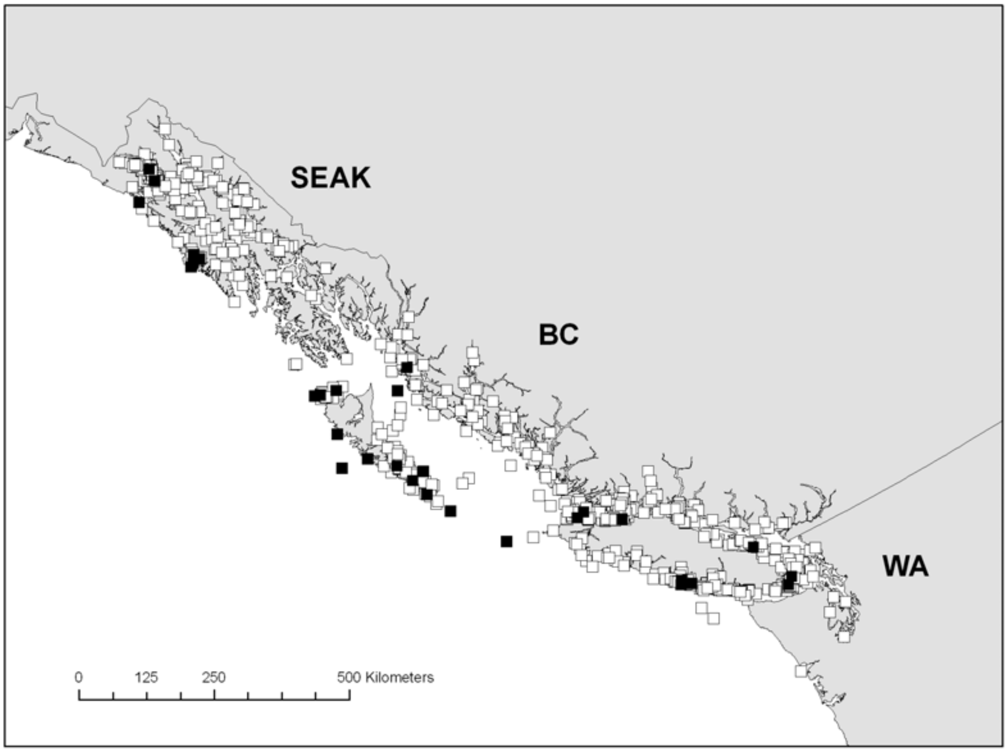

Figure 1. Locations of 1947 photographic encounters with West Coast Transient killer whales (open squares) in the coastal waters of Washington State (WA), British Columbia (BC) and Southeastern Alaska (SEAK). Solid squares show sightings with transient killer whales that did not associate regularly (more than once) with the WCT social network, and were typically on the offshore periphery of the study area.

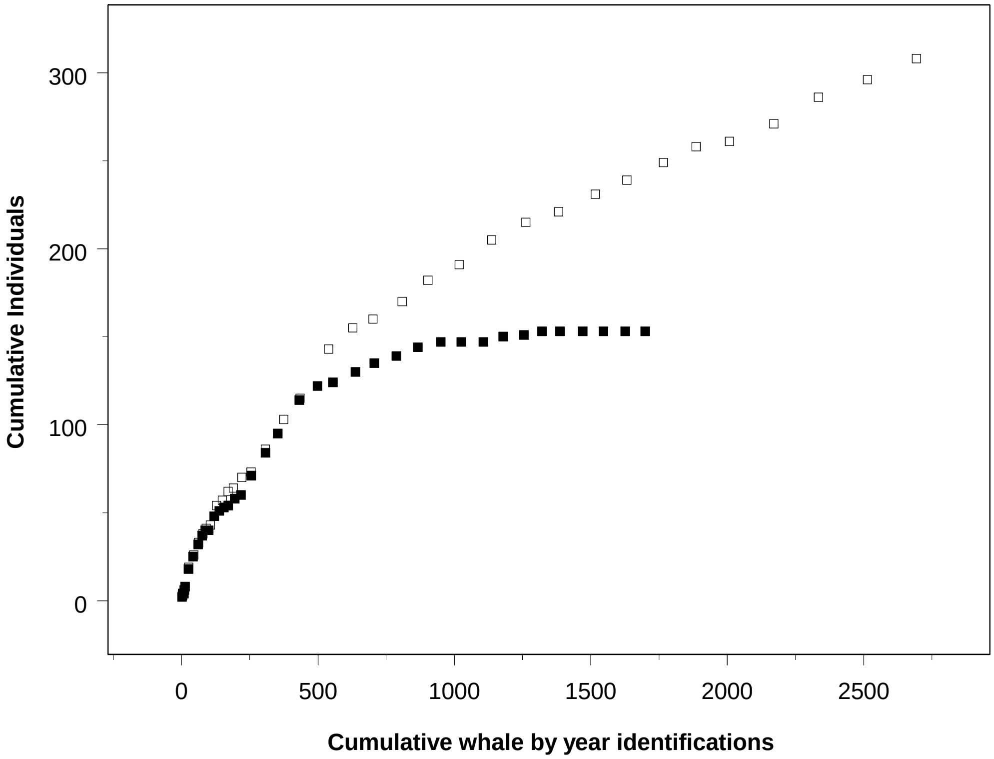

Figure 2. A discovery curve of individual whales in the west coast transient population (open squares), with an asymptote in the discovery of non-calf individuals (closed squares).

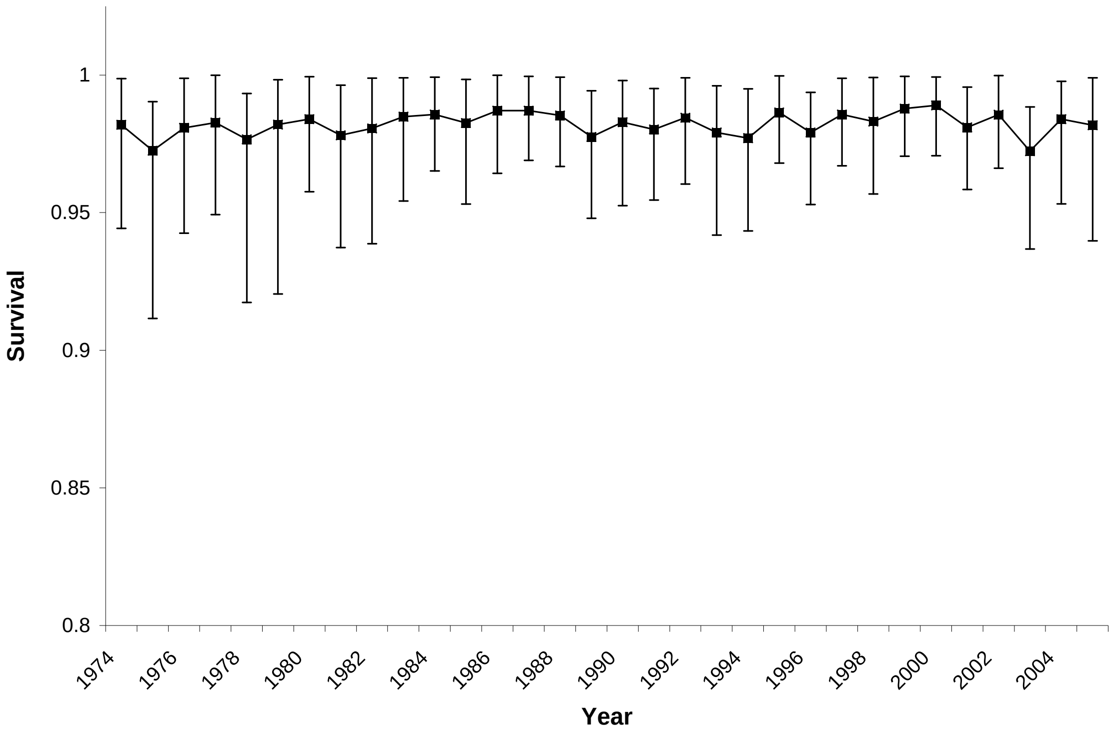

Figure 3**.** Survival probability φ_j, for each year *j* from 1974 to 2005. Estimates are presented as posterior medians (squares) with 95% posterior intervals (bars).

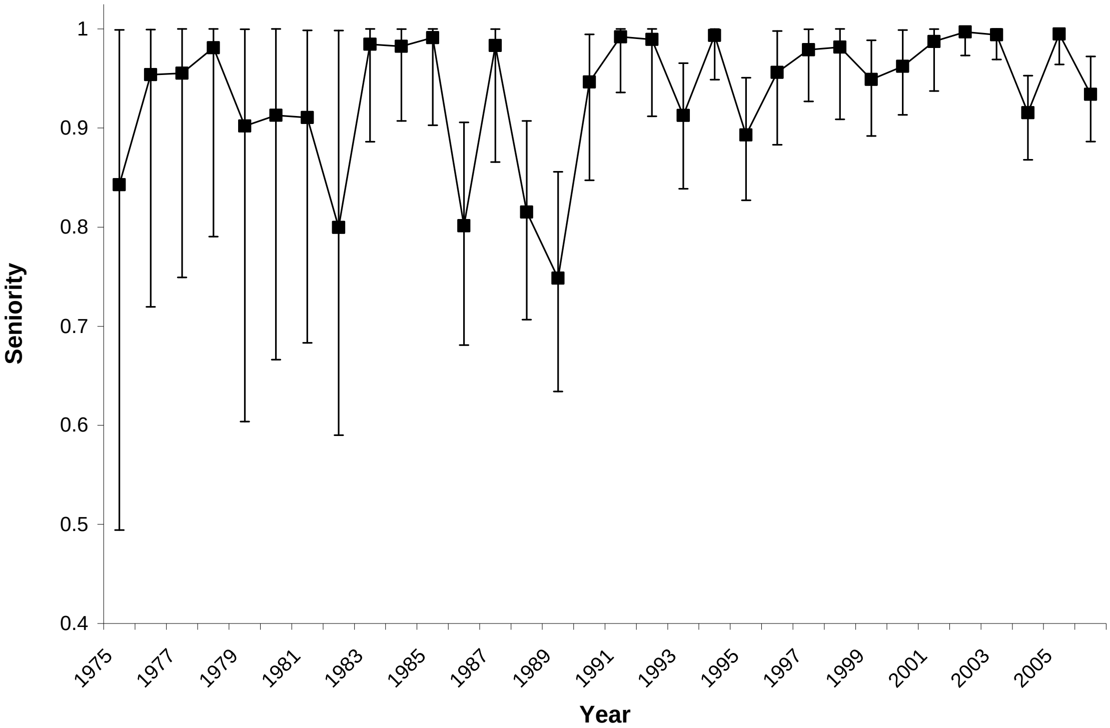

Figure 4. Seniority probability γ_j, for each year j from 1975 to 2006. Estimates are presented as posterior medians (squares) with 95% posterior intervals (bars).

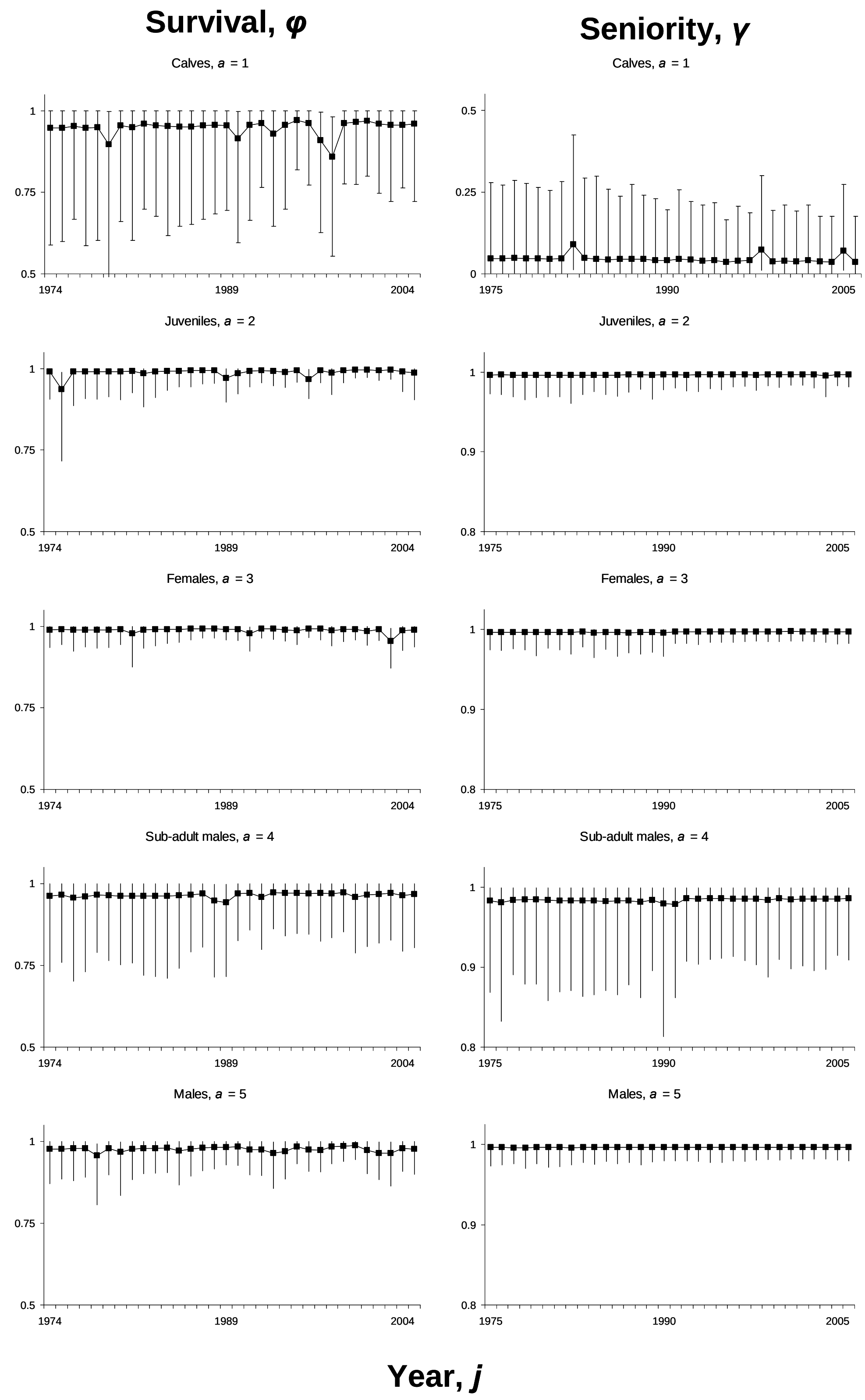

Figure 5. Survival φ_{aj} and seniority probabilities γ_{aj} for each year j for each age class a. Estimates are presented as posterior medians (squares) with 95% posterior intervals (bars).

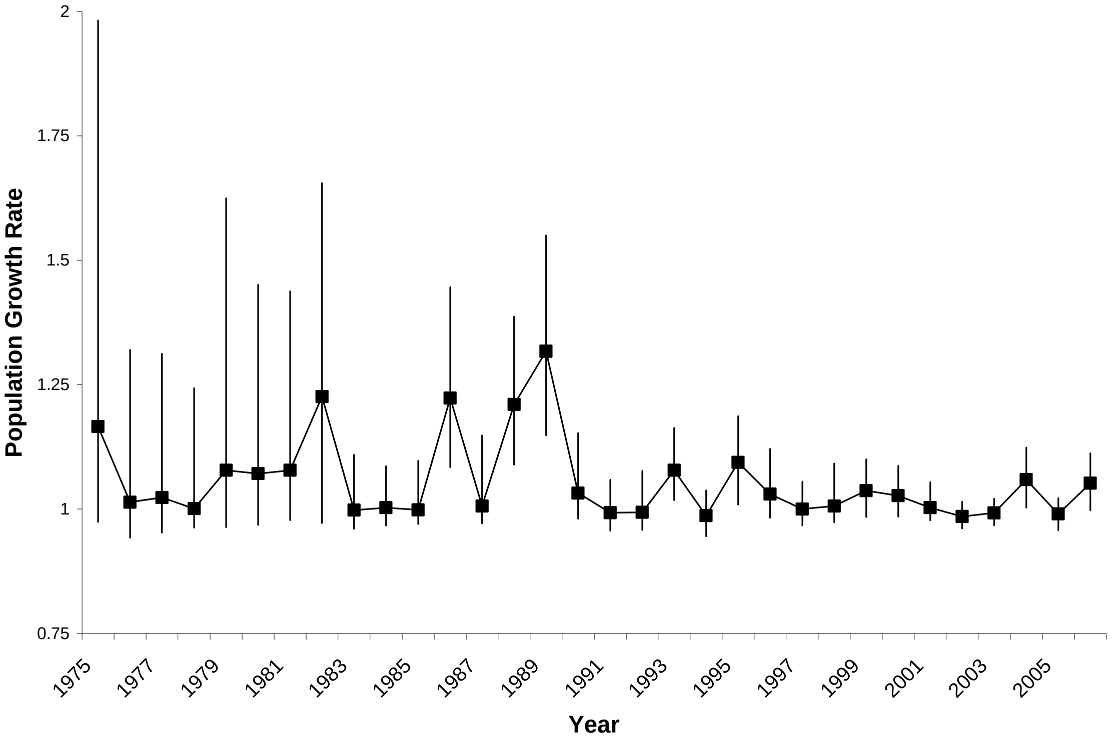

Figure 6. The population growth rate ρ_j, for each year *j* from 1975 to 2006. Estimates are presented as posterior medians (squares) with 95% posterior intervals (bars).

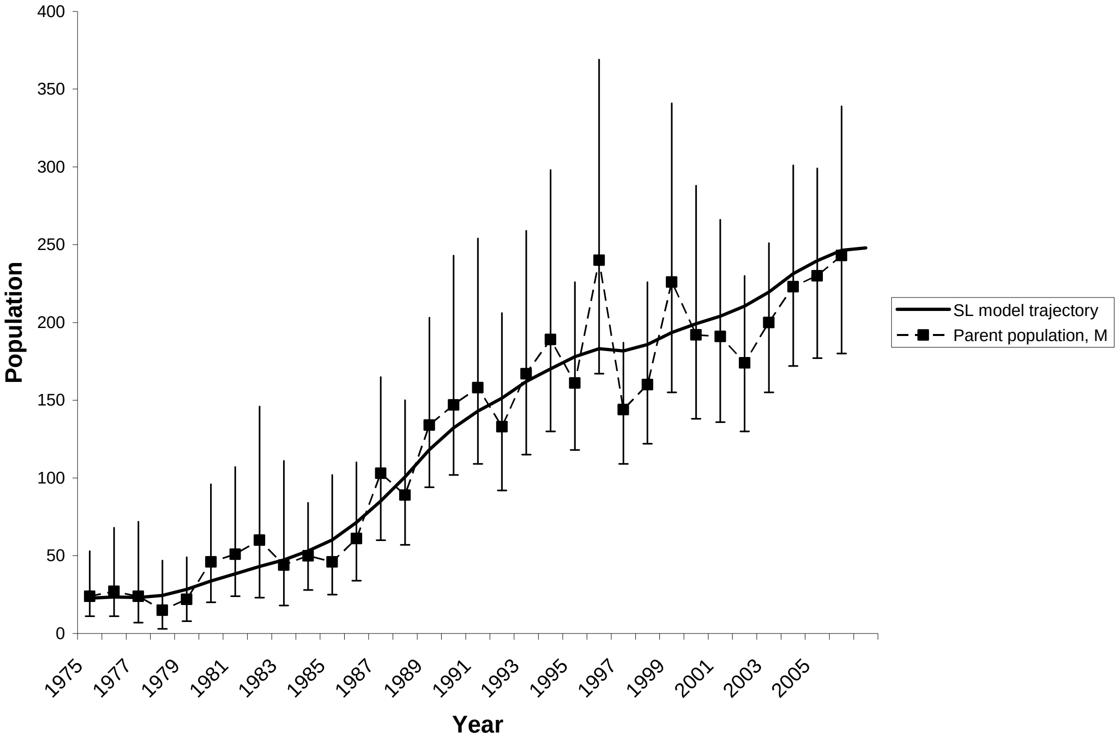

Figure 7. The overall “parent” population size of whales that use (but are not necessarily always in) the study area M_j, for each year j from 1975 to 2006. Estimates are presented as posterior medians (squares) with 95% posterior intervals (bars). The solid line shows the median population trajectory estimated from the stochastic logistic (SL) model.

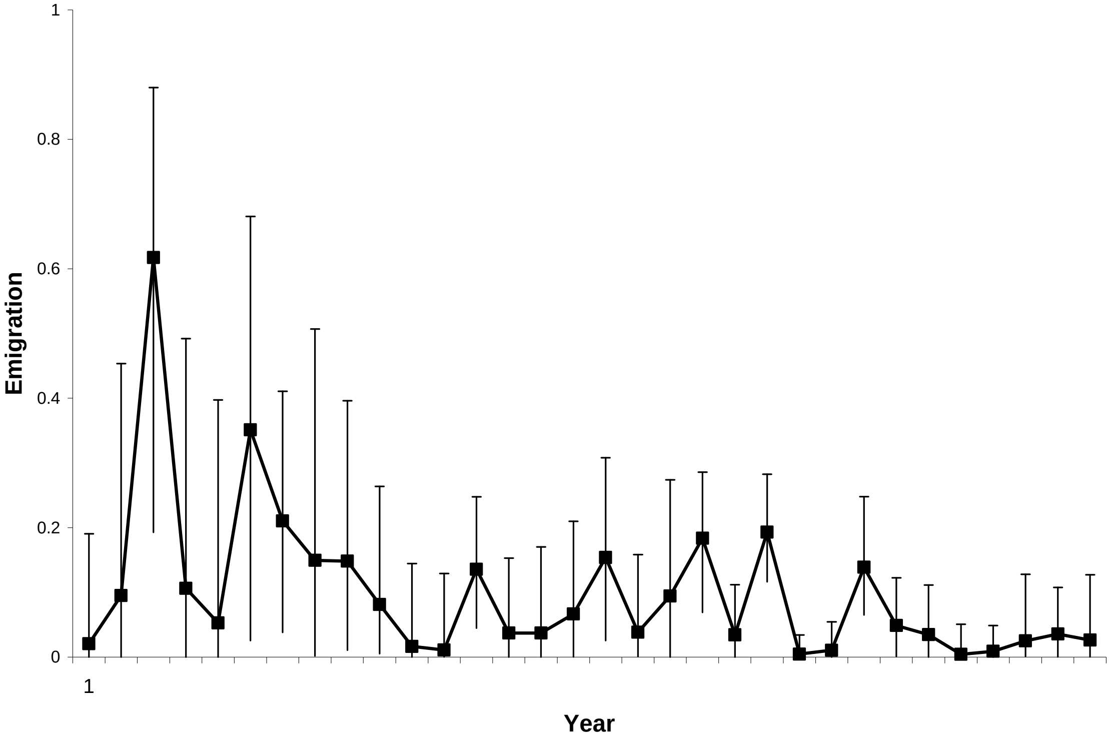

Figure 8. Emigration probability λ_j, for each year j from 1974 to 2005. Estimates are presented as posterior medians (squares) with 95% posterior intervals (bars).

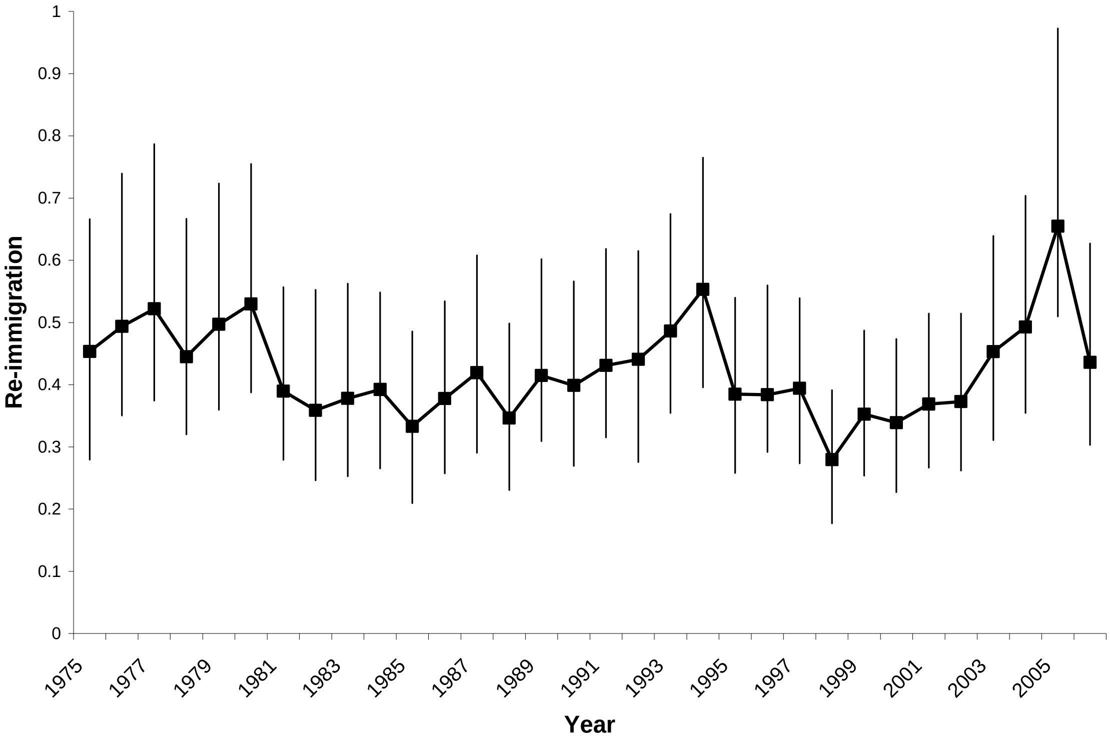

Figure 9. Re-immigration probability μ_j, for each year j from 1975 to 2006. Estimates are presented as posterior medians (squares) with 95% posterior intervals (bars).

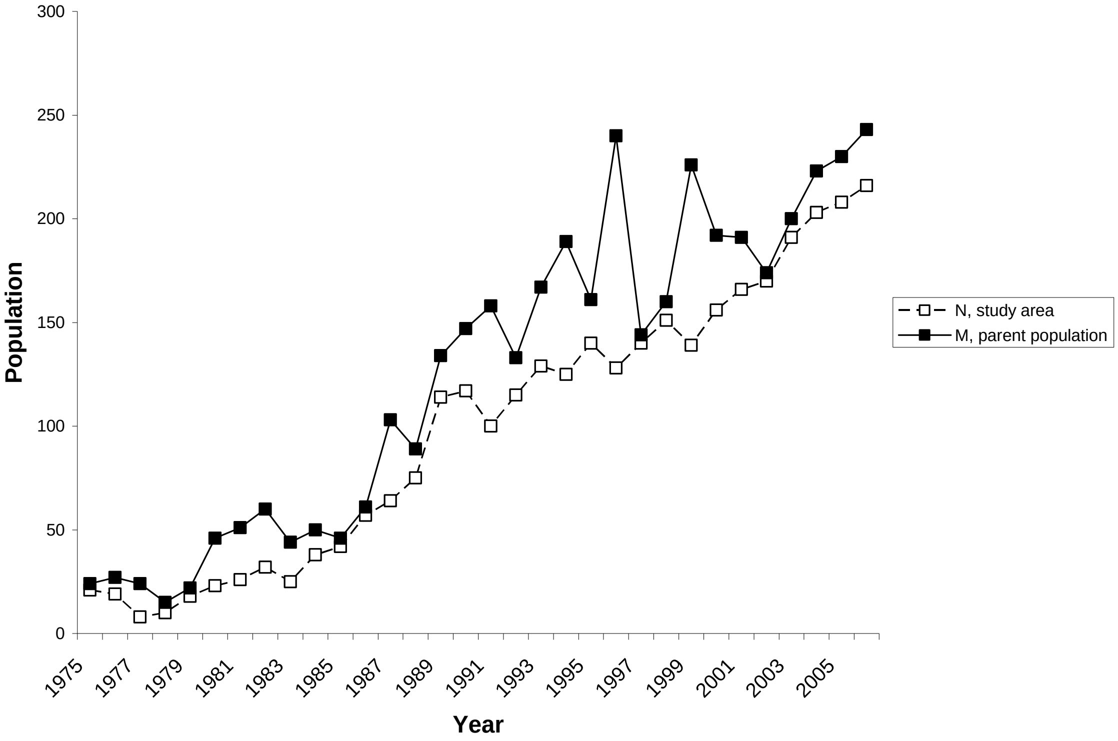

Figure 10. The median of the posterior estimates for the parent population size M_j (closed squares) and the abundance of individuals in the study area N_j (open squares) in each year *j* from 1975-2006.

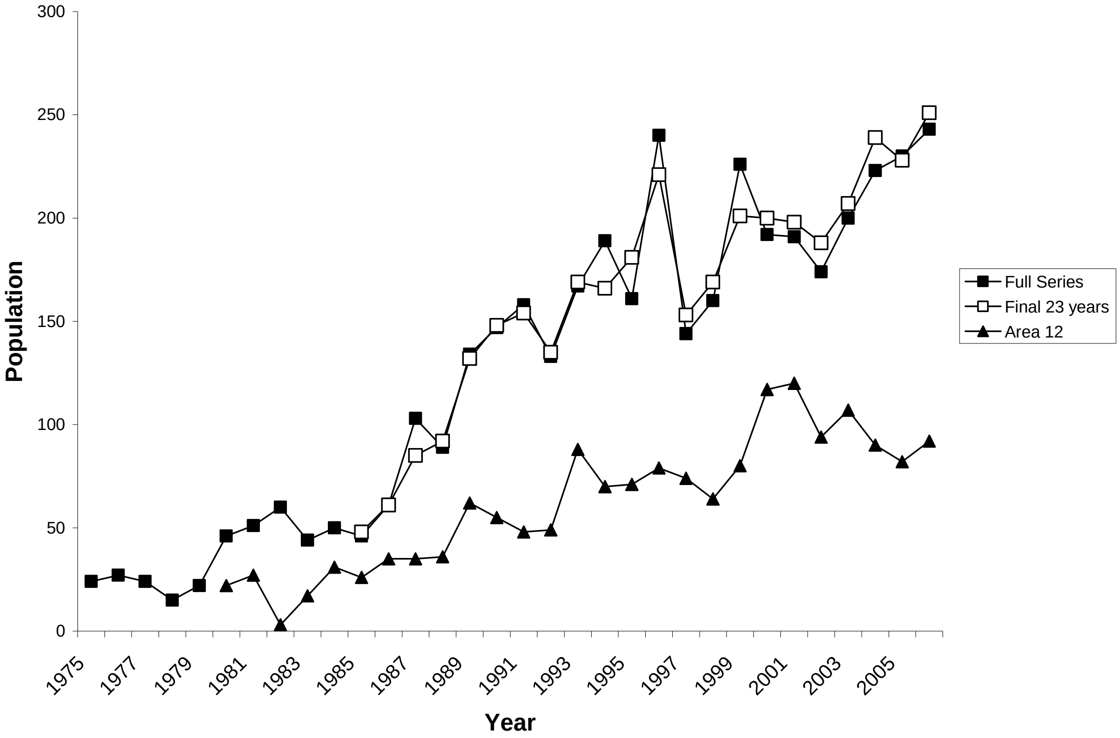

Figure 11. The median of the posterior estimates for the parent population size M_j estimated from the full 1975-2006 dataset (closed squares), alongside estimates of M_j for a reduced dataset using just the final 23 years of the study (open squares) and the a dataset restricted geographically (closed triangles) to just DFO Pacific Fisheries Management Area 12 off northeastern Vancouver Island.

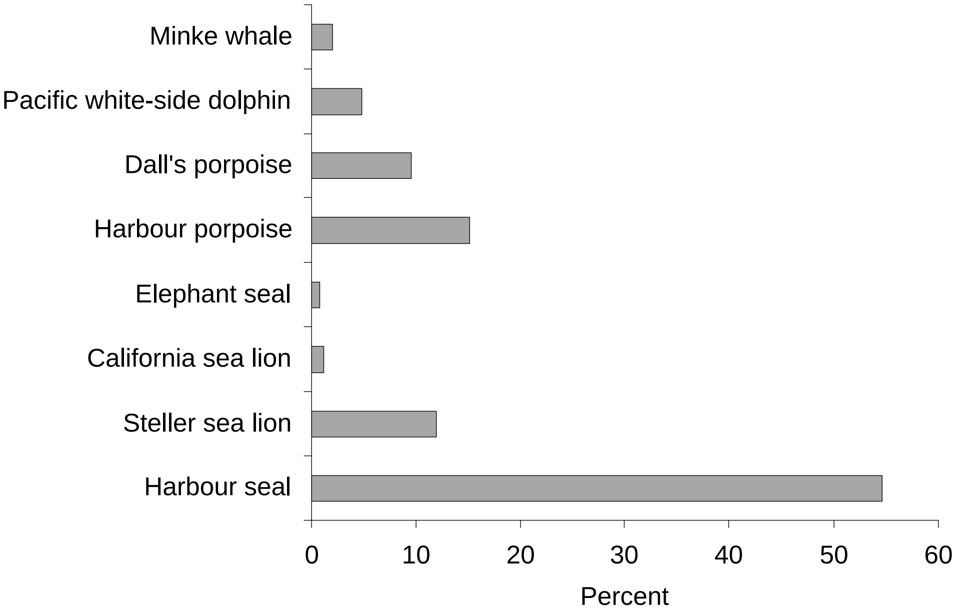

Figure 12. Species composition of 251 marine mammal kills by West Coast transient killer whales, 1975-2006. Data from Ford et al. (1998) and DFO-CRP (unpubl. data).