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Via Email and U.S. Mail

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Re: Comments on proposed modification of Letter of Authorization for Taking Marine Mammals Incidental to Oil and Gas Activities in Cook Inlet, Alaska

On behalf of the Center for Biological Diversity and Cook Inletkeeper, we urge the National Marine Fisheries Service (NMFS) to reconsider its proposal to issue a modified Letter of Authorization (LOA) to Hilcorp Alaska LLC (Hilcorp) to take marine mammals incidental to oil and gas activities in Cook Inlet. The agency should instead revoke the incidental take regulations under which the modified LOA would be issued because the activities to be conducted under the regulations will have more than a negligible impact on critically endangered Cook Inlet beluga whales.¹ In the alternative, if NMFS proposes to allow nighttime seismic surveys without clearing the exclusion zone, its incidental take regulations and associated environmental analysis documents must be revised and re-circulated for public comment in advance of any further activity occurring under the regulations.

The arrival last week in Kachemak Bay of the 273 foot seismic vessel *Polarcus Alima* under contract to Hilcorp makes all the more visceral what is in store for the Cook Inlet ecosystem this month if NMFS allows Hilcorp to proceed. In fact, the vessel has already begun operating, with reports from whale researchers on the water yesterday and evidence on its own 24 Hour Plan that Hilcorp is continuing to survey after dark despite its statement that “[a]fter step 2 Alima will be delayed by darkness.”²

¹ See e.g., 16 U.S.C. § 1371(a)(5)(B)(ii) (requiring NMFS to revoke incidental take authorizations when they are having, or may have, more than a negligible impact on the species concerned).

² Hilcorp, Cook Inlet Seismic Updates Polarcus Alima 24 Hour Plan available at <http://www.hilcorp.com/3dsurvey/>.



Figure 1. The *Polarcus Alima* in Kachemak Bay (September 2019). *Photo credit:* Cook Inletkeeper.

From plankton to whales, the best available science demonstrates that the wildlife of Cook Inlet will suffer. Cook Inlet beluga whales in particular are critically endangered with only 328 individuals remaining and no signs of recovery since they were protected under the Endangered Species Act. Along with catastrophic events like oil spills and cumulative impacts (including noise), noise itself is one of the top three activities currently threatening Cook Inlet belugas.

Through the incidental take regulations for this activity, NMFS has authorized the take of up to 35 Cook Inlet beluga whales per year, or up to 175 whales over the five year project period. This represents 53 percent of the population, which is not a small number by any definition of the term. Now it proposes to modify a mitigation measure pertaining to 3D seismic surveying during Year 1 of Hilcorp's activity that will expose even more beluga whales to harm and further imperil this vulnerable population.

According to the agency it:

published a mitigation measure in error that stated before ramp up of seismic airguns during the 3D seismic survey, the entire exclusion zone (EZ) must be visually cleared by protected species observers (PSOs). This measure is correct for operations beginning in daylight hours. However, visually clearing the entirety of the EZ to ramp up airgun activity at night was not NMFS' intent.³

Instead, NMFS states that its intent was that observers should monitor the EZ to “the greatest extent possible” for 30 minutes prior to ramp-up of nighttime operations, but that Hilcorp would still be allowed to initiate ramp-up at night as long as no marine mammals were seen during this 30 minute time period. If any marine mammal is observed in the EZ, during daylight hours or at night, ramp up would not commence until “either the animal has voluntarily left and been

³ NMFS, Takes of Marine Mammals Incidental to Specified Activities; Taking Marine Mammals Incidental to Oil and Gas Activities in Cook Inlet, Alaska, Notice; request for comments on modification of Letter of Authorization, 84 Fed. Reg. 41,957 (August 16, 2019).

visually confirmed outside the EZ or the required amount of time (15 for porpoises and pinnipeds, 30 minutes for cetaceans) have passed without re-detection of the animal.”

There are several problems with this result, including but not limited to:

- **NMFS continues to fail to take into consideration the critically endangered status of Cook Inlet beluga whales.** As NMFS has elsewhere acknowledged, the removal of even one endangered Cook Inlet beluga whale will impede the recovery of this species.⁴ Cumulative and synergistic impacts (including most notably noise) pose serious threats to these belugas. The Cook Inlet beluga whale Recovery Plan recommended a reassessment of the current system for the allocation of take by harassment to evaluate cumulative impacts comprehensively.⁵ We cannot stress enough that as the Marine Mammal Commission has recommended, Hilcorp’s proposed activities, including its seismic surveys, should not be authorized unless NMFS can ensure that take will not impede the survival and recovery of the Cook Inlet beluga whale population, which it has not done.
- **NMFS has failed to explain why other measures are not practicable to minimize take and to maximize monitoring and enforcement of take limits.** NMFS requires that Hilcorp operations shutdown should the entirety of the exclusion zone not be visible due to inclement weather, such as fog, but now states it will allow nighttime 3-D seismic surveys when the entirety of the exclusion zone is not visible and cannot be cleared. NMFS called ramping up at night “essential” to Hilcorp’s “survey design” and “*most practicable*” (emphasis added) without explaining why other designs were not *practicable*.⁶ Very little is known about the behavior of whales in general at night. This counsels for NMFS using more caution when authorizing activities at night, not less.

Protected species observers can be ineffective even in the best of conditions and particularly when NMFS has identified *monitoring zones that are significantly smaller than the harassment zones* as it did here. Hilcorp is “required to shutdown airguns if any beluga whale is observed within the Level B isopleth,” which is estimated to be 7,330 m from the source. Yet for seismic surveys using full array, NMFS only requires 1,500 m to be monitored. NMFS has not provided a sufficient explanation for why a greater monitoring area consistent with the harassment isopleth is not required nor why other mitigation measures are not employed to monitor the full Level B and Level A harassment isopleths. NMFS has arbitrarily dismissed or ignored other methods and technologies, including additional protected species observers, improved Passive

⁴ National Marine Fisheries Service, Stock Assessment Report: Beluga Whale (*Delphinapterus leucas*) Cook Inlet Stock (December 30, 2018) (“even one take every 2 years may still impede recovery”).

⁵ A team of scientists just implemented such an approach for Southern Resident killer whales and found that a cumulative threats model incorporating all priority threats (in that case Chinook salmon abundance, vessel noise/presence, vessel strike, and PCB contamination) best predicted observed demographic rates. Murray, C.C., Hannah, L.C., Doniol-Valcroze, T., Wright, B., Stredulinsky, E., Locke, A., and R. Lacy. 2019. Cumulative Effects Assessment for Northern and Southern Resident Killer Whale Populations in the Northeast Pacific. DFO Can. Sci. Advis. Sec. Res. Doc. 2019/056. x. + 88 p.; *see also* Francis, C.D. and J. R. Barber. 2013. A Framework for understanding noise impacts on wildlife: an urgent conservation priority, *Front Ecol Environ* 2013; 11(6): 305–313, doi:10.1890/120183.

⁶ 84 Fed. Reg. at 41,958.

Acoustic Monitoring (PAM), and thermal imaging technologies to increase the chances of observation particularly at night.⁷ Furthermore, as stated in previous comments, other time-area closures are warranted during other times and in other areas of high biological significance to Cook Inlet beluga whales.

If monitoring is not effective in the whole harassment zone, then it is unclear how Hilcorp will accurately report take or how NMFS will determine if take limits are met or exceeded. The Cook Inlet beluga whale population is a small population which has continued to decline over the 25 years the agency has actively been monitoring them, and at this rate and using these limited measures, they will monitor them right into extinction.

- **This proposed perpetuates NMFS’ pattern of underestimating take of beluga whales by Hilcorp.**
 - *Ramp up is not considered take.* NMFS states that part of its mitigation involves introducing less than full-power seismic noise into the environment (i.e., a ramp up) to disturb marine mammals to the point they leave the area so they will not be exposed to the full power sound. This is an intentional harassment, yet the animals disturbed to the point of fleeing the area (a clear change of behavior) are not considered as “taken” in NMFS’s analysis and thus the agency’s small numbers and negligible impact determinations are flawed. The agency fails to provide a reasonable and clear explanation why some noises in excess of acoustic thresholds are not considered as take.
 - *24-hour take is elevated over instances of take.* NMFS’s statement that nighttime operations minimize the amount of days that active acoustic sources are emitting sound into the marine environment and thus minimizes exposure is not supported by anything but conclusory statements. For example, NMFS states “that any potential slight increase in the probability of injury (in the form of a small degree of PTS, and not considered at all likely, or authorized, for beluga whales or other mid-frequency specialists) is offset by the reduced behavioral harassment and reduced potential for more serious energetic effects expected to result from the significant reduction in the overall number of days across which the area will be ensonified by the airgun operation.”⁸
 - *NMFS focuses on Level A and disregards Level B harassment.* NMFS states that this proposed change in nighttime mitigation does not change the predicted take numbers or its negligible impact analysis because “the predicted Level A

⁷ See, e.g. Horton, T.W., A. Oline, N. Hauser, T.M. Khan, A. Laute, A. Stoller, K. Tison, and P. Zawar-Reza. 2017. Thermal Imaging and Biometrical Thermography of Humpback Whales. *Front. Mar. Sci.* 4:424. doi: 10.3389/fmars.2017.00424; Zitterbart D.P., L. Kindermann, E. Burkhardt, O. Boebel. 2013. Automatic Round-the-Clock Detection of Whales for Mitigation from Underwater Noise Impacts. *PLoS ONE* 8(8): e71217. doi:10.1371/journal.pone.0071217; Press, R. 2015. Automatic Whale Detector, version 1.0, provided by NOAA, retrieved 16 September 2019 from <https://phys.org/news/2015-02-automatic-whale-detector-version.html> (describing thermal imaging cameras in use by the National Marine Fisheries Service to track gray whales throughout the day and night).

⁸ *Id.*

harassment (injury) numbers conservatively do not include any sort of an adjustment to account for the effectiveness of any of the measures.”⁹ NMFS says nothing about the impact of this change on Level B numbers. Level B harassment still constitutes take and should be independently reviewed and authorized.

- *NMFS continues to use outdated science that underestimates exposure and inaccurately represents density.* A new publication by Tyack and Thomas (2019) provides additional support for the conclusion that the calculation of a single-threshold approach (such as used by NMFS for Level B acoustic harassment) underestimates the number of animals affected.¹⁰ Current exposure criteria also fail to reflect best available science or the cumulative effects of seismic surveys and other noise pollution and stressors on marine mammals.¹¹ As noted in previous comments, NMFS has failed to use the best scientific and commercial information available to estimate Cook Inlet beluga whale density. There is more recent data from protected species observers, beluga researchers, aerial surveys, and opportunistic sightings that should be included.
- **This nighttime exception is not in NMFS’ final incidental take regulations.** The final incidental take regulations require that the exclusion zone “is clear of marine mammals,” a determination that “must be made during a period of good visibility (*i.e.* the entire EZ must be visible to the naked eye).”¹² Nothing is said in the regulations to distinguish nighttime activities. The word “night” appears nowhere in the regulations. These regulations must be revised to reflect NMFS actual intention if different from the language promulgated.

In conclusion, NMFS has identified noise as a high threat to the Cook Inlet beluga whales and recognizes it could result in population-level consequences, but it appears unwilling to take action to reduce the amount of anthropogenic noise introduced into beluga habitat, as evidenced by the continued permitting of extremely loud sources of noise like this project. While NMFS sponsors events like the upcoming *Third Annual Belugas Count!* on September 21 to engage the public in conserving the endangered Cook Inlet beluga whale, we urge the agency to do more to protect the whales from clear and cumulative threats so that there are whales left to count in the future.

A first step would be to revoke the previously issued incidental take authorizations and to stop authorizing any take of this critically endangered species incidental to oil and gas activity in the Inlet unless and until the agency has a better understanding of why the Cook Inlet beluga has failed to recover and continues to decline and has a reasonable basis for determining that authorizing additional takes will have no more than a negligible impact on the species. Short of

⁹ *Id.*

¹⁰ Tyack, P. and L. Thomas, Using dose–response functions to improve calculations of the impact of anthropogenic noise Aquatic Conserv: Mar Freshw Ecosyst. 2019; 29(S1):242–253.

¹¹ Nowacek, D.P., C.W. Clark, D. Mann, P. J.O. Miller, H.C. Rosenbaum, J.S. Golden, M. Jasny, J. Kraska, and B.L. Southall. 2015. Marine seismic surveys and ocean noise: time for coordinated and prudent planning, *Front Ecol Environ* 2015; 13(7): 378–386, doi:10.1890/130286.

¹² 50 C.F.R. § 217.164(g)(3).

that, the agency must revise this LOA and the incidental take regulations to ensure the LOA and regulations are accurate and based on best available science, as mandated by law.

Sincerely,

/s/ Julie Teel Simmonds

Senior Attorney, Oceans Program

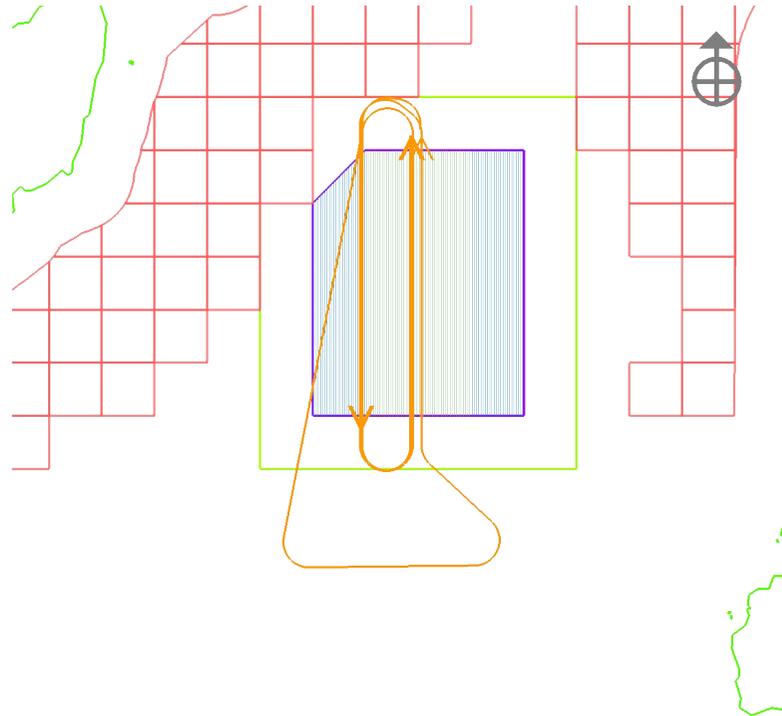
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ATTACHMENTS



Polarcus Alima 24 Hour Plan



After step 2 Alima will be delayed by darkness

Plan Step	Preplot Name	Azimuth	Duration	SOL	EOL	Local Time SOL	Local Time EOL	Line Change Time
1	1360	180.0°	1:14	Sep 16 01:54	Sep 16 03:09	Sep 15 17:54	Sep 15 19:09	0 minutes
2	1728	360.0°	3:04	Sep 16 04:27	Sep 16 07:31	Sep 15 20:27	Sep 15 23:31	1:18
3	1792	0.0°	3:04	Sep 16 17:14	Sep 16 20:19	Sep 16 09:14	Sep 16 12:19	9:42
4	1344	180.0°	3:01	Sep 16 21:38	Sep 17 00:39	Sep 16 13:38	Sep 16 16:39	1:19
5	1712	0.0°	3:04	Sep 17 01:58	Sep 17 05:02	Sep 16 17:58	Sep 16 21:02	1:18

A framework for understanding noise impacts on wildlife: an urgent conservation priority

Clinton D Francis^{1*} and Jesse R Barber^{2†}

Anthropogenic noise is an important environmental stressor that is rapidly gaining attention among biologists, resource managers, and policy makers. Here we review a substantial literature detailing the impacts of noise on wildlife and provide a conceptual framework to guide future research. We discuss how several likely impacts of noise exposure have yet to be rigorously studied and outline how behavioral responses to noise are linked to the nature of the noise stimulus. Chronic and frequent noise interferes with animals' abilities to detect important sounds, whereas intermittent and unpredictable noise is often perceived as a threat. Importantly, these effects can lead to fitness costs, either directly or indirectly. Future research should consider the range of behavioral and physiological responses to this burgeoning pollutant and pair measured responses with metrics that appropriately characterize noise stimuli. This will provide a greater understanding of the mechanisms that govern wildlife responses to noise and help in identifying practical noise limits to inform policy and regulation.

Front Ecol Environ 2013; 11(6): 305–313, doi:10.1890/120183 (published online 15 Jul 2013)

An emerging aim in applied ecology and conservation biology is to understand how human-generated noise affects taxonomically diverse organisms in both marine (eg Slabbekoorn *et al.* 2010; Ellison *et al.* 2012) and terrestrial (eg Patricelli and Blickley 2006; Barber *et al.* 2010; Kight and Swaddle 2011) environments. Noise is a spatially extensive pollutant and there is growing evidence to suggest that it may have highly detrimental impacts on natural communities; yet efforts to address this issue of emerging conservation concern lack a common framework for understanding the ecological consequences of noise. A conceptual scaffold is critical to scientific progress and to

its ability to inform conservation policy. As more attention and resources are invested in understanding the full ecological effects of noise, it is important that investigators design research questions and protocols in light of the many possible costs associated with noise exposure and also that they properly link responses to several relevant features of noise, such as intensity, frequency, or timing, that could explain wildlife responses (Panel 1).

Here we introduce a framework using a mechanistic approach for how noise exposure can impact fitness at the level of the individual organism as a result of changes in behavior, and identify several acoustic characteristics that are relevant to noise exposure and ecological integrity. We provide representative examples of noise impacts, primarily from terrestrial systems; however, these issues are equally applicable to organisms in aquatic environments. We stress that various responses to noise exposure are less obvious than those that have typically been studied to date, such as signal modifications (eg changes in vocal frequency, amplitude, or vocalization timing) and decreases in site occupancy (eg Bayne *et al.* 2008; Francis *et al.* 2011b). Importantly, probable behavioral responses to noise that merit further scientific study might be detrimental to individual fitness and may have severe population-level consequences. As we show below, the presence of a species in a noisy area cannot be interpreted as an indication that it is not being impacted by elevated sound levels, because there are many potential costs associated with noise exposure that have not been rigorously studied.

In a nutshell:

- Noise is an intense, widespread pollutant, relevant to conservation efforts worldwide
- Using the number of animals present in environments exposed to anthropogenic noise as the sole metric of noise impacts can be deceiving because there are many hidden costs of noise exposure (eg compromising predator/prey detection or mating signals, altering temporal or movement patterns, increasing physiological stress)
- To ensure that conservation initiatives (and efforts to establish regulatory limits) are relevant, investigators must properly characterize a suite of noise features
- Reducing noise exposure and incorporating sound measurement into environmental planning will quickly benefit ecological systems

■ Variation in responses to the same noise stimulus

Species differ in their sensitivities to noise exposure (Bayne *et al.* 2008; Francis *et al.* 2009, 2011a); however,

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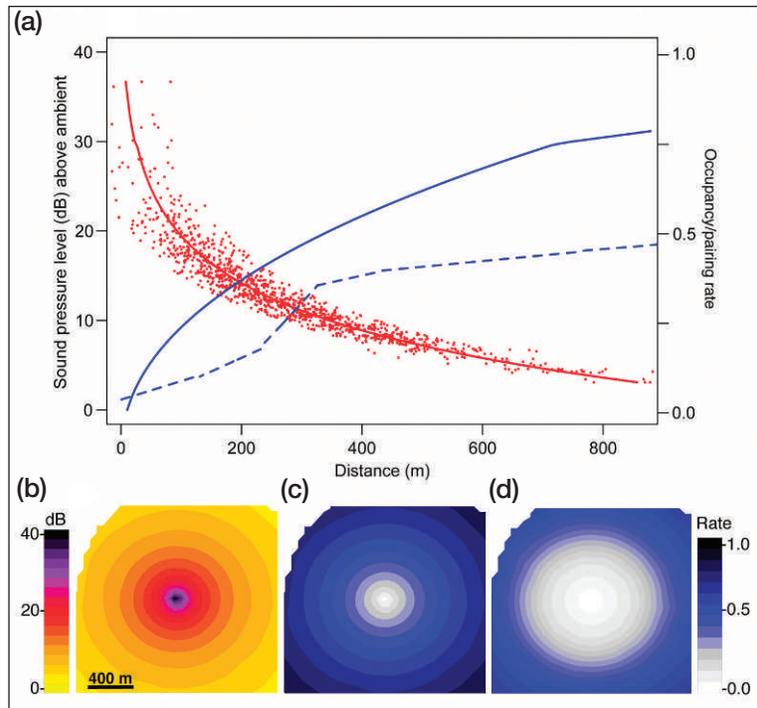


Figure 1. Responses to the same noise stimulus can take a variety of shapes. (a) The sound pressure level (SPL) of noise (red) decreases with increasing distance from the source but may not reach “baseline” ambient levels until ~1 km away (this distance will vary depending on noise source and the environment). Response curves for species occupancy (blue solid line) and pairing rates (blue dashed line) in response to noise may have unique shapes, as might other measures of species responses to noise stimuli. The relationship between SPL and distance is from Francis et al. (2011c) and Francis (unpublished data) with noise generated from gas well compressors. Behavioral responses are hypothetical but based on responses in Francis et al. (2011c). (b) Spatial propagation of elevated noise levels from a point source (such as a single car or an oil/gas compressor station), which decays at a spreading loss of 6 dB or more per doubling of distance, due to the geometry of the spherical wave front. It is important to note that line sources (such as a busy highway; not shown) lose only 3 dB per doubling of distance due to their cylindrical wave front. Clearly, knowledge of the geometry of anthropogenic noise stimuli is essential to understanding the scale of exposure. (c and d) Spatial representation of (c) species occupancy and (d) pairing success surrounding a point source of noise.

the degree to which individuals vary in sensitivity to noise during each life-history stage or due to behavioral context has been underappreciated. For example, ovenbird (*Seiurus aurocapilla*) habitat occupancy appears uninfluenced by noise exposure (Habib et al. 2007; Bayne et al. 2008; Goodwin and Shriver 2011), yet males defending noisy territories are less successful in attracting mates (Habib et al. 2007). Reed buntings (*Emberiza schoeniclus*) also show reduced pairing success in noisy areas (Gross et al. 2010). Such examples should serve as a warning to biologists, land managers, and policy makers: the same noise stimulus can affect various response metrics in different ways. An organism might show little to no response to noise in terms of habitat occupancy or foraging rate, for example, but may experience strong negative

impacts in terms of pairing success, number of offspring, physiological stress, or other measures of fitness (Figure 1). Because the various responses may range from linear to threshold functions of noise exposure, investigators should take an integrative approach that incorporates several different metrics (eg density, pairing success, number of offspring), rather than using a single metric to describe how noise influences their study organism. But which alterations in behavior are most likely to occur and which are the most detrimental? These are important questions because funding and logistical constraints ensure that measuring all of the potential impacts of noise is impossible. Fortunately, the nature of sound stimuli can guide investigators toward likely behavioral changes that may influence fitness.

■ Characterizing noise and the disturbance–interference continuum

Determining whether a particular noise stimulus is within an organism’s sensory capabilities is foremost in importance; if a sound consists of frequencies that are outside of an organism’s hearing range, it will not have a direct effect (Panel 1; Figure 2). Provided that an organism can hear the noise stimulus, its acoustic energy could cause permanent or temporary hearing loss, but this might only occur when the animal is extremely close to the source of the noise (Dooling and Popper 2007).

Instead, sounds may have their greatest influence on behavior, which then translates into fitness costs, but how and why noise elicits a response can vary greatly (Figures 2 and 3). At one extreme, noise stimuli that startle animals are perceived as threats and generate self-preservation responses (eg fleeing, hiding), which are similar to responses to real predation risk or non-lethal human disturbance (ie the risk–disturbance hypothesis, which posits that animal responses to human activities are analogous to their responses to real predation risk; Frid and Dill 2002). Noise stimuli at this end of the continuum are often infrequent, but are abrupt and unpredictable. At the other end of the continuum, noise can impair sensory capabilities by masking biologically relevant sounds used for communication, detection of threats or prey, and spatial navigation. These noise stimuli tend to be frequent or chronic and their spectral (ie frequency) content overlaps with biologically relevant sounds. Increases in noise intensity (loudness or amplitude) will increase the severity of the impacts, regardless of whether it is perceived as a threat or masks biologically relevant sounds. An important supplement to this dichotomy is

that limited stimulus processing capacity could be responsible for some detrimental effects. Noise stimuli of various kinds might act as a distraction, drawing the animal's attention to a sound source and thereby impairing its ability to process information perceived through other sensory modalities (Chan *et al.* 2010). Alternatively, noise may reduce auditory awareness, trigger increased visual surveillance, and compromise visually mediated tasks. The mechanistic details and ecological importance of such distractions still need to be fully explored. Regardless, the conservation implications of understanding the importance of noise as a distractor are not trivial; if distraction is a fundamental route for noise impacts, our concern might spread beyond those frequencies that overlap with biologically relevant signals.

■ Behavioral changes

Although a limited number of laboratory studies have suggested that noise may affect gene expression, physiological stress, and immune function directly (Figure 3a; Kight and Swaddle 2011), most noise-related impacts appear to involve behavioral responses across four categories: (1) changes in temporal patterns, (2) alterations

in spatial distributions or movements, (3) decreases in foraging or provisioning efficiency coupled with increased vigilance and anti-predator behavior, and (4) changes in mate attraction and territorial defense (Figure 3). As demonstrated below, these disturbance-, distraction-, and masking-mediated behavioral changes could directly impact individual survival and fitness or lead to physiological stress that may then compromise fitness.

Changes in temporal patterns

Sound stimuli that are perceived as threats can alter temporal patterns; for example, red foxes (*Vulpes vulpes*) cross busy roads when traffic rates are lower, suggesting noise cues might be affecting the timing of their movements (Figure 3b; Baker *et al.* 2007). Similarly, noise from boat traffic disrupts the timing of foraging by West Indian manatees (*Trichechus manatus*), potentially influencing foraging efficiency and energy budgets (Figure 3m; Miksis-Olds *et al.* 2007). Noise can also change behavior due to interference with cue detection. European robins (*Erithacus rubecula*) avoid acoustic interference from urban noise by singing at night, when noise levels are lower than during daylight hours (Figure 3c; Fuller *et al.* 2007). Although this example may appear to be an

Panel 1. Sound features relevant to noise-impact studies

In the main text we discuss how the spectral (frequency) composition of noise is related to an organism's hearing range and its ability to detect relevant sounds. For these reasons, it is critical that researchers collect sound-level data with an appropriate frequency-weighting filter. For instance, the "A" filter on many sound-level meters is based on equal loudness contours for human hearing; this filter provides a conservative estimate of bird hearing and is the best readily-available weighting for bird studies (Dooling and Popper 2007). However, whether working with birds or other taxa, it is best to simultaneously record and measure the noise using a "flat" frequency filter, then truncate the resulting spectral output to the most relevant frequency range for each species of interest (see below).

Investigators should also avoid the temptation to characterize a noise stimulus as a single decibel value, whether weighted or not, as other metrics that describe the noise are equally important (Figure 2). Time-averaged values, such as equivalent continuous sound level (L_{eq}), can be extremely informative to describe sounds that are chronic or frequent; however, these integration times do not properly characterize sounds that occur once, infrequently, or more regularly. Instead, measurements integrated over several hours will mischaracterize short, abrupt sounds that could be viewed as disturbances, such as noise events created by infrequent and loud military jet overflights that alter the behavior and time budgets of harlequin ducks (*Histrionicus histrionicus*; WebFigure 1; Goudie 2006). For disturbance sounds, exposure metrics that capture each sound event's maximum power (L_{max} ; WebFigure 1a) and the rate at which power rises from the lowest detectable level to its maximum are important (ie onset; Figure 2). L_{max} values are often reported without stating the frequency weighting; in these cases, A-weighting (a human-centric curve) is

assumed, which may be inappropriate for many animals.

In contrast, quantification of chronic noise can best be served with time-averaged values such as L_{eq} (WebFigure 1b). L_{eq} is typically calculated over 24 hours; however, many studies fail to report over what time period L_{eq} values were integrated and a 24-hr integration is assumed, which may not be appropriate for many ecological questions. For example, for a species that is sensitive to traffic noise, such as the white-breasted nuthatch (*Sitta carolinensis*; WebFigure 1b; Goodwin and Shriver 2011), it may be best to truncate the time interval to the hours of biological interest, such as during dawn chorus. Limiting frequency analyses to the hearing or vocal range of the target species or community may also be beneficial (eg Halfwerk *et al.* 2011b). Future studies should aim to use biologically relevant integration times and report these details.

Best practices will include simultaneous acquisition of high-quality audio recordings along with multiple sound level measurements to offer unconstrained opportunities to investigate alternative spectral filtering, time integration, and additional measurements, such as order statistics indicating the percentage of time above a certain decibel level or metrics reflective of the sound event's predictability (Figure 2). Carefully considering how these temporal, intensity, and frequency features (Figure 2b) interact will help investigators identify where along the disturbance–interference continuum (Figure 2a) the stimulus is most likely to fall and will help identify the most likely behavioral responses (Figure 3).

Above all, to maximize interpretability of results, facilitate comparisons among studies, and provide meaningful data for conservation measures, it is critical to explicitly report the acoustic metrics used in each study to describe species responses. Additional sound metric and terminology details can be found in Barber *et al.* (2011) and Pater *et al.* (2009).

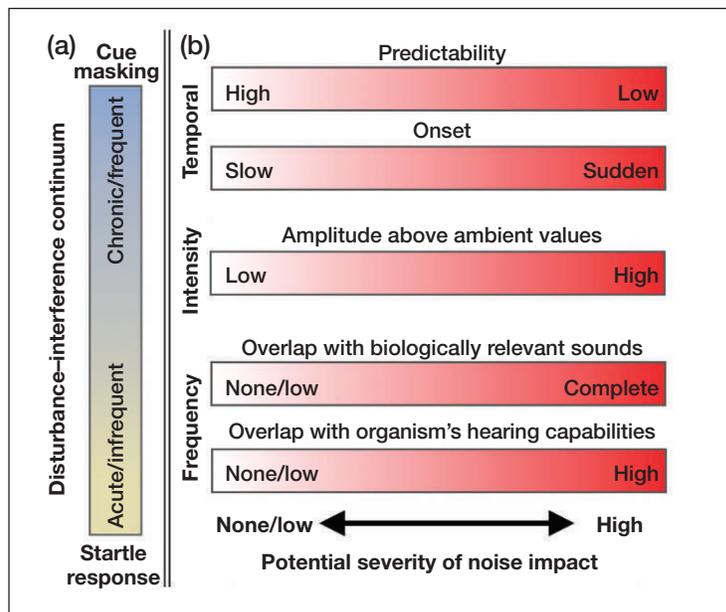


Figure 2. (a) The disturbance–interference continuum can range from acute or infrequent noise stimuli that will likely trigger startle or hide responses to frequent or chronic noises that interfere with cue detection. (b) The severity of an impact from a noise stimulus will depend on the temporal, intensity, and frequency features of the stimulus.

important behavioral adaptation that permits this species to overcome unfavorable acoustic conditions, the consequences of shifting the timing of song delivery are unknown. The effects of signal timing on mate attraction or territorial defense may be just as important to fitness as other signal features (eg frequency, syntax). Changes in the timing of song delivery of less than one hour can break down signaler–receiver coordination so that conspecific males do not recognize species-specific signals (Luther 2008). If signaler–receiver coordination is disrupted between singing males and responsive females, the behavioral flexibility that permits shifts in signal timing in response to noise may possibly be maladaptive.

Sleep is an important factor and follows a strong temporal profile. Although a substantial body of research has investigated the impact of noise on sleep in humans, scant information is available regarding its effects in other animals (reviewed in Kight and Swaddle 2011). Understanding the importance of sleep disruption on overall fitness is critical as we might expect detrimental influences even for species not typically described as dependent upon hearing (eg visually oriented predators such as raptors).

Alterations in spatial distributions or movements

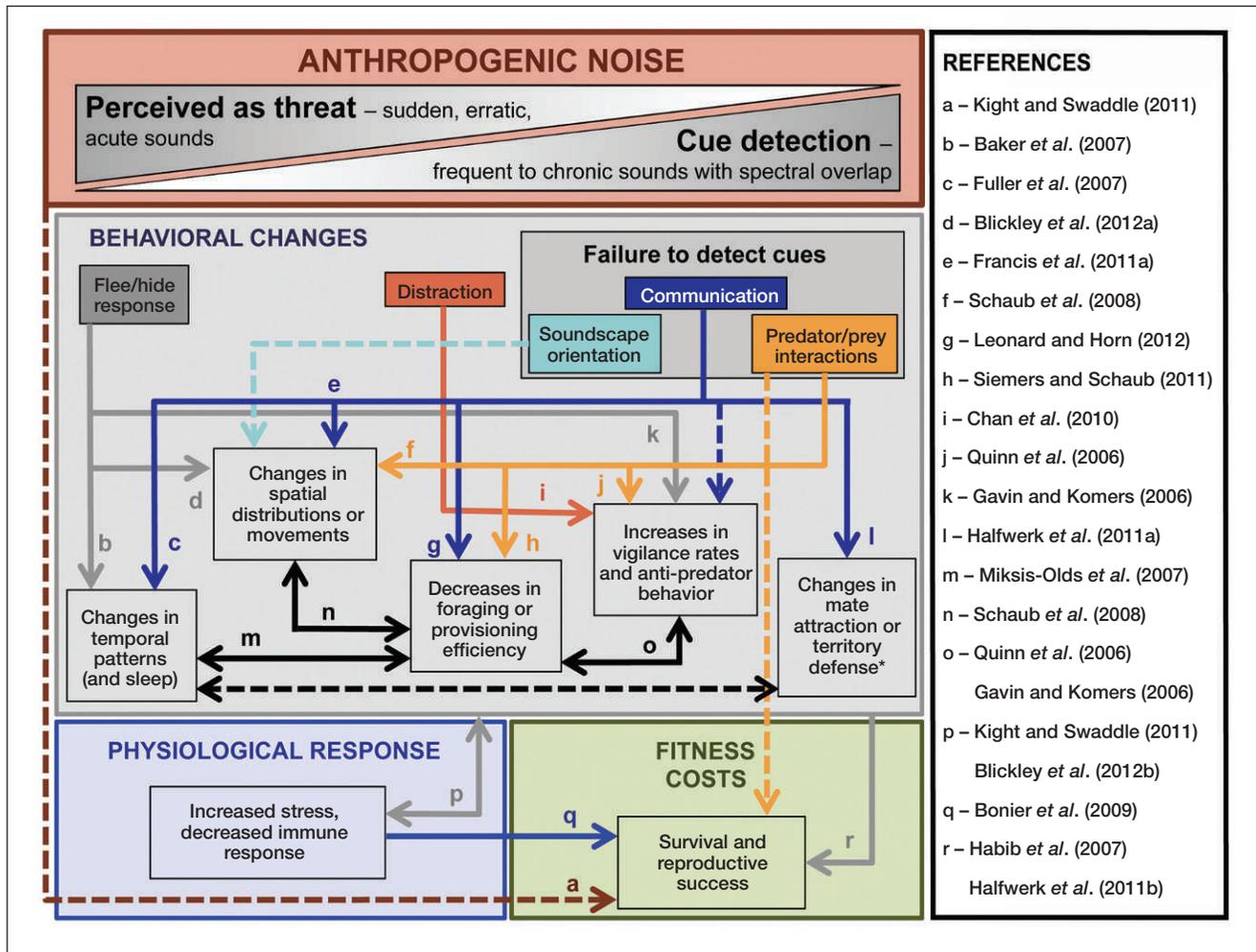
Among the most obvious responses to noise are site abandonment and decreases in spatial abundance. These metrics may also be easiest and least costly to quantify, which perhaps explains why there are many such examples in the literature (eg Bayne *et al.* 2008; Eigenbrod *et al.* 2008; Francis *et al.* 2009). However, noise itself can affect an

investigator's ability to measure responses to noise. For example, increases in continuous noise of 5–10 decibels (dB, A-weighted; Panel 1) above baseline can reduce bird numbers during standard bird surveys by one-half, greatly biasing measures of site occupancy and abundance (Ortega and Francis 2012). If not carefully considered, this detection problem could bias subsequent interpretations and management efforts.

Despite the known effects of noise on population sizes, there is still considerable evidence to suggest that animals may abandon areas when frequent or chronic noise stimuli interfere with cue detection or when more variable sounds are perceived as threats (Bayne *et al.* 2008; Goodwin and Shriver 2011; Blickley *et al.* 2012a). Birds with low-frequency vocalizations experience more acoustic interference from chronic low-frequency anthropogenic noise and therefore exhibit stronger negative responses to noise in their habitat use than birds with high-frequency vocalizations that experience less acoustic interference (Figure 3e; Francis *et al.* 2011a). These masking effects can be spatially extensive, potentially impairing communication at distances ranging from 0.5 to 1.0 km or farther from the noise source (Blickley and Patricelli 2012). Furthermore, changes in spatial distributions due to noise's effect on cue detection are not restricted to intraspecific communication; for instance, greater mouse-eared bats (*Myotis myotis*), which locate terrestrial prey based on sounds they generate when walking, also avoid hunting in noisy areas (Figure 3f; Schaub *et al.* 2008). In addition to disrupting cue detection at the intra- and interspecific level, ambient noise may also interfere with cue detection used for movement at larger spatial scales. Some frog species use conspecific calls to locate appropriate breeding habitat, while some newt species use heterospecific calls for the same purpose (reviewed in Slabbekoorn and Bouton 2008). Whether noise exposure impedes animals from using such acoustic beacons to locate critical resources (eg water, food, habitat) is unknown and should be a focus of future research.

Site abandonment or decreases in population numbers can also occur in response to unpredictable, erratic, or sudden sounds, which are perceived as threats (Figure 3d). For example, greater sage grouse (*Centrocercus urophasianus*) lek attendance declines at a higher rate in response to experimentally introduced intermittent road noise than to continuous noise (Blickley *et al.* 2012a), suggesting that sage grouse site occupancy may depend more on perceived risk than on masking of acoustic cues. Nevertheless, masking of communication may have other consequences (Figure 1).

Species undoubtedly differ in their sensitivities to disruptive sounds, but individuals within a population also show such differences (Bejder *et al.* 2006). Individuals can



REFERENCES

- a – Kight and Swaddle (2011)
- b – Baker *et al.* (2007)
- c – Fuller *et al.* (2007)
- d – Blickley *et al.* (2012a)
- e – Francis *et al.* (2011a)
- f – Schaub *et al.* (2008)
- g – Leonard and Horn (2012)
- h – Siemers and Schaub (2011)
- i – Chan *et al.* (2010)
- j – Quinn *et al.* (2006)
- k – Gavin and Komers (2006)
- l – Halfwerk *et al.* (2011a)
- m – Miksis-Olds *et al.* (2007)
- n – Schaub *et al.* (2008)
- o – Quinn *et al.* (2006)
- Gavin and Komers (2006)
- p – Kight and Swaddle (2011)
- Blickley *et al.* (2012b)
- q – Bonier *et al.* (2009)
- r – Habib *et al.* (2007)
- Halfwerk *et al.* (2011b)

Figure 3. Conceptual framework for understanding how noise stimuli – perceived as a threat or interfering with cue detection (the disturbance–interference continuum) – can elicit behavioral responses that have direct consequences for fitness or via a physiological stress response, which can also feed back to behavioral changes. Startle/hide responses are more likely to occur in response to noise stimuli that are perceived as a threat (acute, erratic, or sudden sounds). Problems arising from a failure to detect cues are more likely to occur when noise stimuli are chronic and overlap with biologically relevant cues used for communication, orientation, and predator/prey detection. Problems arising from distraction may occur as a result of sounds with features ranging from those that interfere with cue detection to those that are perceived as threats. Lowercase letters indicate studies (listed on the right) providing evidence for the link made for each arrow. Dashed arrows signify a link that we predict as important but for which no current evidence exists. The asterisk denotes that which could result from a change in behavior or a failure to change behavior in response to noise.

vary greatly in their behavioral responses to stimuli, which may explain the variations in their ability to cope with environmental change (Sih *et al.* 2004). The redistribution of sensitive and tolerant individuals across the landscape may not appear to be a problem. However, in the case of social animals, where group living provides protection from predation, the loss of sensitive individuals from the group through site abandonment could increase predation risk for the group as a whole through the removal of the most vigilant group members. These sensitive individuals, who are now isolated from the group, lose the benefit of safety in numbers. Depending on population structure and the scale at which these individuals are displaced by noise, genetic diversity may be reduced because traits that govern risk-averse (shy/sensi-

tive) and risk-prone (bold) behaviors can be heritable (Dingemanse *et al.* 2002).

Site abandonment and changes in abundance provide only a limited understanding of how noise can impact wildlife populations and communities. Importantly, abundance can also be misleading because areas where individuals are abundant do not always translate into high fitness for those individuals (eg Johnson and Temple 1986). Using such evidence to conclude that noise has no impact is problematic; individuals may not have alternative areas to occupy or other responses (survival, mating success, reproductive output) may be negatively affected by noise even when abundance is high (Figure 1a). These possibilities are especially likely when a noise stimulus is new and demographic processes

have not had time to impact population size or when the population in an area that is exposed to noise is supplemented by individuals from elsewhere (ie source-sink dynamics).

Decreases in foraging or provisioning efficiency and increased vigilance and anti-predator behavior

Noise can impair foraging and provisioning rates directly (Figure 3, g and h) or indirectly as a consequence of increased vigilance and anti-predator behavior (Figure 3, i–k, o). When noise is perceived as a threat, an organism may miss foraging opportunities (“missed opportunity cost”; Brown 1999) while hiding or as a result of maintaining increased vigilance (Figure 3k; Gavin and Komers 2006). Missed opportunities can also occur when noise interferes with cue detection. For instance, nestling tree swallows (*Tachycineta bicolor*) exposed to noise beg less in response to recorded playbacks of parents arriving at nests (eg calls, movement, sounds) than nestlings in quiet conditions, presumably because the ambient noise masks parent-arrival sounds (Figure 3g; Leonard and Horn 2012). Unfortunately, this study did not determine whether missed provisioning opportunities translated into costs, such as reduced nestling mass or fledging success.

Noise that interferes with cue detection can also hamper predators’ hunting abilities. For example, among greater mouse-eared bats, search time for prey was shown to increase and hunting success to decrease with exposure to experimental traffic noise (Figure 3h; Siemers and Schaub 2011). This decrease in foraging success may explain why some predators avoid noisy areas (Figure 3n; eg Schaub *et al.* 2008; Francis *et al.* 2009). Noise also impairs foraging in three-spined sticklebacks (*Gasterosteus aculeatus*), resulting in more unsuccessful hunting attempts (Purser and Radford 2011). Noise also possibly interferes with the ability of prey species to hear approaching predators, which could impact fitness directly. Although likely, elevated predation risk due to noise has yet to be demonstrated, but some evidence does suggest that animals exposed to noise behave as though they are at greater risk of predation. For example, in the chaffinch (*Fringilla coelebs*), continuous noise impairs auditory surveillance, triggering increased visual surveillance, as a result of which the birds spend less time foraging (Figure 3j; Quinn *et al.* 2006). Noise that serves as a distraction may also lead to an increased latency in predator-escape response (Figure 3i; Chan *et al.* 2010), potentially compromising survival. Both distraction and elevated vigilance could also cause a decrease in foraging rates and success (ie a trade-off; Figure 3o; Gavin and Komers 2006; Quinn *et al.* 2006). Collectively, these studies suggest that both interference noise and noise perceived as a threat decrease the rate and frequency at which organisms obtain food. Studies aimed at understanding the extent to which these behavioral shifts

represent a metabolic expense (relevant to survival and reproductive success) will help to reveal the hidden costs of noise exposure.

Changes in mate attraction and territorial defense

The most direct way in which noise may alter an individual’s ability to attract mates or defend its territory is through energetic masking, in which potential receivers are simply unable to hear another individual’s acoustic signals through noise that is frequent or continuous during important temporal signaling windows. Changes made to acoustic signals appear to be an adaptive behavioral adjustment that permits individuals to communicate under noisy conditions (eg Fuller *et al.* 2007; Gross *et al.* 2010; Francis *et al.* 2011b), yet these shifts could also incur a cost. In noisy areas, female great tits (*Parus major*) more readily detect male songs sung at higher frequencies than females typically prefer (Halfwerk *et al.* 2011a). However, males who sing predominately at higher frequencies experience higher rates of cuckoldry (Figure 3l). Great tits breeding in noisy areas also have smaller clutches and fewer fledglings (Halfwerk *et al.* 2011b); similarly, eastern bluebirds (*Sialia sialis*) experience decreased productivity when nesting in areas with elevated noise levels (Kight *et al.* 2012). Paired with patterns of decreased pairing success in noisy areas (Habib *et al.* 2007; Gross *et al.* 2010), these studies suggest that short-term signal adjustments in response to anthropogenic noise might function as evolutionary traps (eg Schlaepfer *et al.* 2002) in which behavioral responses to novel acoustic stimuli could be maladaptive. That is, behavioral shifts to be heard in noisy areas may come with the cost of compromising the attractiveness of the signal to potential mates. This possibility remains to be tested against other potential explanations for declines in pairing or reproductive success, but emphasizes why investigators should measure aspects of fitness in noise-impact studies rather than simply documenting changes in site occupancy or abundance.

Finally, although the list of species known to shift their signals in response to noise is growing, there is at least one frog species and some bird species that do not alter their vocalizations in response to noise (eg Hu and Cardoso 2010; Love and Bee 2010; Francis *et al.* 2011b). More work is needed to provide a thorough understanding of the phylogenetic distribution of noise-dependent vocal change and researchers should strive to publish negative results, as knowledge of the apparent absence of these behavioral modifications is just as important as knowledge of their presence.

■ Linking behavioral changes, physiological responses, and fitness costs

The behavioral changes mentioned above can have direct consequences for fitness (Figure 3r), such as reduced pairing success (Habib *et al.* 2007) or reduced

reproductive success (Halfwerk *et al.* 2011b). However, behavior can influence, and be influenced by, physiological responses (Figure 3p; Kight and Swaddle 2011), which in turn can affect fitness (Figure 3q; Bonier *et al.* 2009). Kight and Swaddle (2011) reviewed many links between noise, physiological stress, and behavioral change, so we only briefly mention them here.

It is well known that increased physiological stress affects fitness (Figure 3q); yet, to our knowledge, a direct link between increased physiological stress due to noise and decreased survival or reproductive success has not been shown in wild animals. The best evidence for this potential link comes from two studies. In one, Blickley *et al.* (2012b) found that greater sage grouse on leks exposed to experimental playback of continuous natural gas drilling noise or intermittent road noise had higher fecal glucocorticoid metabolites (fGMs) than individuals on control leks. The authors suggested that masking of cues likely resulted in elevated stress levels, inhibiting social interactions or leading to a heightened perception of predation risk. In the other, Hayward *et al.* (2011) showed that experimental exposure to motorcycle traffic and motorcycle noise increased fGMs in northern spotted owls (*Strix occidentalis caurina*). In an observational component of the same study, spotted owls nesting in areas with higher levels of traffic noise fledged fewer offspring, even though they did not have elevated fGMs, suggesting that the effects of road noise may have been offset by greater prey availability in noisy areas. These two studies demonstrate that noise may lead to decreased fitness in sage grouse and spotted owls, and also clearly indicate that more research is needed to determine how noise exposure, physiological stress, and fitness are linked in wild populations.

■ Scaling up behavioral responses

Here, we have focused on effects of noise exposure at the level of the individual; however, studies that integrate individual behavior, population responses among multiple species, and species interactions are critical to understanding the cumulative, community-level consequences of noise. Measures of species richness are a good starting point, but may be misleading because species may respond negatively, positively, or not at all to sound stimuli (Bayne *et al.* 2008; Francis *et al.* 2009), individuals within a single species may respond differently to the same stimulus (Sih *et al.* 2004), and individuals that remain in noisy areas may suffer from one or more of the fitness costs discussed above. This variation within and among species in response to noise guarantees that communities in noisy areas will not always be subsets of the species that make up communities in comparable quiet areas. Researchers should couple standard measures of richness and alpha (local) diversity with beta-diversity metrics that reflect variations in the composition of species within communities and among sites.

Nevertheless, additional investigations will be needed to understand why species respond to sound stimuli as they do. Settlement patterns may not hinge on the intensity of noise, but are perhaps due to the presence or absence of cues indicating the presence of predators and heterospecific competitors (Francis *et al.* 2009). These other species (ie predators or competitors) may have unique settlement patterns in response to noise and will complicate efforts to measure how noise directly affects the species of interest. Disentangling these interactions will also be essential to understanding the consequences of noise exposure for organisms that are not directly impacted by noise, such as plants that depend on noise-sensitive faunal taxa (Francis *et al.* 2012) or animals whose hearing range is not tuned to a particular frequency that makes up a sound stimulus.

■ Conclusions

Both policy and scientific literature have often oversimplified the effects of noise on wild animals, typically suggesting that species either are sensitive and abandon noisy areas or are not and remain. In our experience with stakeholders, habituation is an oft-cited reason for persistence and an absence of noise impacts, yet research on other stressors indicates that acclimation to a stressor might not release an organism from costs to fitness (Romero *et al.* 2009). Additionally, we have shown how behavioral modifications among individuals confronted with noise – even those individuals that outwardly appear to habituate – can lead to decreased fitness. Challenging the assumption that habituation to noise equals “no impact” will be difficult, but it will also be a critical component in revealing how a range of behavioral mechanisms link noise exposure to fitness costs. Ideally, we need to predict which combination of noise characteristics and behavioral contexts are most detrimental and under what circumstances behavioral changes affect fitness directly or indirectly. This will require an array of experimental and observational approaches and frameworks that complement the conceptual structure presented here (Figure 3). Other promising frameworks include the risk–disturbance hypothesis (Frid and Dill 2002), which provides an avenue for understanding energetic costs associated with wildlife responses to noise disturbances that are perceived as threats. Studies evaluating aspects of habitat selection and acoustic communication in response to noise may find it useful to frame questions in terms of ecological and evolutionary traps (Schlaepfer *et al.* 2002). Furthermore, investigators should strive to measure responses along a range of noise exposure levels to reveal the shape of response curves (eg threshold, linear) because these details will be indispensable to resource managers and policy makers when establishing and modifying regulatory limits that reflect the ecological effects of noise exposure.

An increase in anthropogenic noise levels is only one of many threats to biodiversity on which ecologists and policy makers should focus their attention. However, rel-

ative to other conservation problems, noise may also offer readily available solutions, which, if implemented, could lead to major, measurable improvements for both wildlife and people. For example, use of noise-attenuating walls could reduce the area of a landscape exposed to elevated noise levels from natural gas extraction activities by as much as 70% (Francis *et al.* 2011c) and similar solutions exist for mitigating noise from roadways and cities (Code of Federal Regulations 2010). These mitigation efforts could come with drawbacks; for instance, noise-attenuating walls near roads could restrict the movement of wildlife and impede gene flow. Nevertheless, as we develop a better understanding of the ecological effects of noise, implementation of mitigation efforts can begin in many well-studied and high-priority systems (eg oil and gas developments in natural areas, transportation networks in national parks), where benefits outweigh the potential costs. In addition to protecting contiguous natural habitat, reducing noise exposure in and around developed areas will not only benefit wildlife populations and diversity, but will also provide adjacent human populations with the suite of physiological benefits afforded by living in a quieter community.

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Thermal Imaging and Biometrical Thermography of Humpback Whales

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Determining species' distributions through time and space remains a primary challenge in cetacean science and conservation. For example, many whales migrate thousands of kilometers every year between remote seasonal habitats along migratory corridors that cross major shipping lanes and intensively harvested fisheries, creating a dynamic spatial and temporal context that conservation decisions must take into account. Technological advances enabling automated whale detection have the potential to dramatically improve our knowledge of when and where whales are located, presenting opportunities to help minimize adverse human-whale interactions. Using thermographic data we show that near-horizontal (i.e., high zenith angle) infrared images of humpback whale (*Megaptera novaeangliae*) blows, dorsal fins, flukes and rostrums record similar magnitude brightness temperature anomalies relative to the adjacent ocean surface. Our results demonstrate that these anomalies are similar in both low latitude and high latitude environments despite a $\sim 16^{\circ}\text{C}$ difference in ocean surface temperature between study areas. We show that these similarities occur in both environments due to emissivity effects associated with oblique target imaging, rather than differences in cetacean thermoregulation. The consistent and reproducible brightness temperature anomalies we report provide important quantitative constraints that will help facilitate the development of transient temperature anomaly detection algorithms in diverse marine environments. Thermographic videography coupled with laser range finding further enables calculation of whale blow velocity, demonstrating that biometrical measurements are possible for near-horizontal datasets that otherwise suffer from emissivity effects. The thermographic research we present creates a platform for the delivery of three important contributions to cetacean conservation: (1) non-invasive species-level identifications based on whale blow shapes and velocities recorded by infrared videography; (2) reduced ship-strike rates through automated thermographic cetacean detection systems deployed in high traffic areas; (3) monitoring the spatial and temporal distributions of endangered animals in remote habitats.

Keywords: humpback whale, infrared, biometrical thermography, emissivity, blow velocity

INTRODUCTION

Since the International Whaling Commission (IWC) banned commercial whaling in 1986, many baleen whale species have shown signs of recovery (Thomas et al., 2016). However, all species listed as Least Concern under the International Union for Conservation of Nature and Natural Resources (IUCN) Red List of Threatened Species framework also include threatened subpopulations classified as Vulnerable, Endangered, or Critically Endangered (Thomas et al., 2016). The conservation status, recovery and health of whale populations is very much site and context specific: modern human threats to whales, including ship strikes and entanglement in fishing gear, are not evenly distributed with respect to both space and time (Thomas et al., 2016).

Despite the spatially and temporally dynamic challenges associated with cetacean conservation and protection in the post-whaling era, progress has been made. For example, revisions to shipping lane positions, vessel traffic management plans and mandatory maximum vessel speeds along the eastern coast of North America correlate with significant reductions in North Atlantic right whale (*Eubalaena glacialis*) deaths due to ship strikes (Laist et al., 2014; Thomas et al., 2016). However, the risks associated with ship strikes remain high elsewhere. Necropsies performed on stranded whales demonstrate that at least one humpback whale (*Megaptera novaeangliae*), one fin whale (*Balaenoptera physalus*), and two blue whales (*Balaenoptera musculus*) are killed by ship strikes off the California coast every year (Redfern et al., 2013). Similar analyses suggest that, on average, one Bryde's whale (*Balaenoptera edeni*) is killed every year by ship strikes in New Zealand's Hauraki Gulf (Constantine et al., 2015). True cetacean mortality rates due to human activities at sea are almost certainly higher (Kraus, 1990), however, and the annual loss of even a single individual can be significant for smaller populations of long-lived species with low recruitment rates (Laist et al., 2001).

In an effort to reduce the risks ships pose to large whales, the IWC has developed a 3-year (2017–2020) Strategic Plan that seeks to increase the development and use of whale avoidance technologies (Cates et al., 2017). Acoustic and infrared automated cetacean detection systems are attractive and emerging tools for enhanced cetacean conservation (Zitterbart et al., 2013; Nowacek et al., 2016). The ability to detect whale blows, as far away as 5 km using around-the-clock 360° infrared scanners outfitted with rigorous detection algorithms (Zitterbart et al., 2013), will benefit many, including marine mammal observers onboard large vessels and land-based scientists studying whale movement behavior (e.g., Perryman et al., 1999) and human-whale interactions along rapidly changing coastlines (e.g., Graber, 2011). Infrared thermography can also facilitate the non-invasive collection and monitoring of fundamental biometrical information, including thermal physiology, injury diagnoses and population surveys (McCafferty, 2007).

Infrared cetacean detection systems also create opportunities for conservation biologists and cetacean ecologists to document the spatial and temporal distribution of animals utilizing

remote or inaccessible environments. For example, the Oceania subpopulation of humpback whales, the only migratory humpback whales in danger of going extinct (Childerhouse et al., 2008), seasonally inhabit ~10 million km² of the tropical South Pacific Ocean. Yet, only a handful of scientists, spread across an area of ocean the size of China, actively study these whales. Automated detection systems have the potential to dramatically improve our knowledge of when and where these endangered whales are utilizing highly understudied breeding/calving ground habitats.

However, thermal imaging also has several important limitations. Infrared imaging systems are not inexpensive, particularly so for current high sensitivity models with cryogenically cooled detectors or large focal lengths capable of long-range applications. Infrared detectors also require a direct line of site to the target, yet they can also lose functionality through interaction with sea-spray. The data streams generated by infrared imaging systems are large, creating challenges with data handling, analysis and signal processing. Thermal cameras are also highly inaccurate when imaging scenes from near-horizontal positions due to emissivity effects (Masuda et al., 1988; Cuyler et al., 1992; see Nomenclature).

The effects of emissivity on the brightness temperatures recorded by a thermal imaging device are extremely relevant to cetacean thermography. For example, as a whale exhales, its breath pushes sea water present in the near-surface water column, or nasal depression, or both, into the overlying atmosphere. From observation points at or near sea-level, this spouting of water droplets immediately and drastically changes the angle at which the whale's blow is being measured by the thermal camera. For example, a 2 m high whale blow will be measured perpendicularly (i.e., measured at a 0° zenith angle) from an observation point located 100 m distant and 2 m above sea level. In contrast, the adjacent flat ocean's surface will be measured sub-horizontally at an 89° zenith angle. Similar to blows, emergent dorsal fins, flukes or rostrums will also be measured at a relatively low zenith angles in relation to the adjacent ocean's surface. These rapid changes in the angle at which the object is being imaged will have large effects on the surface brightness temperatures estimated by the thermal imaging device due to the effect zenith angle of observed radiation has on sea water emissivity (Masuda et al., 1988).

The research we present was driven by three primary objectives, all aligned to the IWC's strategic goal of developing large whale avoidance technologies. We sought to: (1) quantify infrared image brightness temperature and brightness temperature anomaly (BTA) values for humpback whale blows, dorsal fins, flukes, and rostrums in both tropical breeding/calving ground and sub-polar feeding ground habitats; (2) calculate humpback whale blow height and blow velocity through coupling of infrared videography with laser-range finding; (3) evaluate the effects of emissivity on thermal imaging data collected from high zenith angle (i.e., oblique to target) positions. Achievement of these objectives creates a platform from which a variety of cetacean conservation tools can be further developed and delivered.

METHODS

Thermal images of humpback whale surfacing features were collected using a Forward Looking Infrared camera (FLIR A615, FLIR Systems, Inc.) and analyzed using FLIR Tools+ software (FLIR Systems, Inc.). The FLIR A615 we used had a focal length of 24.6 mm, $25^\circ \times 19^\circ$ field of view, F-number of 1.0, infrared resolution of 480×640 pixels and a detector pixel pitch of $0.017 \text{ mm pixel}^{-1}$. The camera's detector comprised an uncooled Vanadium Oxide (VoX) long-wavelength (i.e., 7.5–14 μm) microbolometer (see Nomenclature) with a thermal sensitivity of $<0.05^\circ\text{C}$. Infrared images were captured every 0.04 s (i.e., 25 Hz) but frame rates as high as 200 Hz can be achieved with the A615's high-speed windowing option. The A615 was powered by a small 12-volt battery externally strapped to the camera's casing. The A615 was also connected to a FZ-G1 ToughPad tablet computer (Panasonic Corporation) via a high-speed USB cable. A GoPro Hero4 camera (GoPro, Inc.) was affixed to the top of the A615 for contemporaneous visible wavelength image collection. This study was carried out in accordance with the recommendations of the Cook Islands Government. The protocol was approved by the Office of the Prime Minister, Cook Islands Government.

In Rarotonga, infrared and visible wavelength images were collected either $\sim 2 \text{ m}$ above the ocean surface while onboard a Cook Islands Whale Research vessel, or from shore-based positions $\sim 5\text{--}10 \text{ m}$ above sea level on the island's northwest coast (**Figure 1**). In Sitka Sound, all images were recorded $\sim 4 \text{ m}$ above the ocean surface while onboard a commercial whale watching cruise arranged by the Sitka Sound Science Center as part of the annual Sitka Whale Fest (e.g., **Figures 1D,E**). Despite these variable imaging heights, our entire dataset was collected at $>85^\circ$ zenith angles (i.e., $<5^\circ$ off horizontal) due to the range in distances at which whales were imaged. A Nikon Forestry Pro laser rangefinder was used to determine whale distances whenever possible. All measurements were made during Beaufort wind force scale numbers 2–4 and similarly ranked World Meteorological Organization (WMO) Sea State codes.

Brightness temperatures were extracted from individual thermal images using the line measurement tool available in FLIR Tools+. Two lines for temperature data extraction were drawn across each image: the first line was drawn vertically through the background scene immediately adjacent to the targeted whale feature (i.e., blow, dorsal fin, fluke, rostrum, **Figure 1**), and the second line was drawn vertically such that it passed through the maximum brightness temperature included within the targeted whale feature. Thermal benchmarks included within each image, such as the steep thermal gradient across the ocean–atmosphere boundary, were used to align the pixels included in each line's thermal profile (**Figure 2**). Once aligned, the brightness temperatures recorded by each line were subtracted from each other in order to calculate BTA-values at the individual pixel scale for each whale feature analyzed (**Figure 2**).

Because the A615's pixel pitch and focal length were known, independent measurement of whale distances by laser ranging allowed us to estimate blow height from thermal image

pixel measurements by combining the optical lens equation,

$$\left(\frac{1}{\text{blow distance (m)}} \right) + \left(\frac{1}{\text{image distance (m)}} \right) = \frac{1}{\text{focal length (m)}} \quad (1)$$

with the magnification equation,

$$\left(\frac{\text{image height (m)}}{\text{blow height (m)}} \right) = \left(\frac{\text{image distance (m)}}{\text{blow distance (m)}} \right) \quad (2)$$

and the camera's pixel pitch,

$$\text{pixel pitch} \left(\frac{\text{m}}{\text{pixel}} \right) = \frac{\text{image height (m)}}{\text{image height (pixel)}} \quad (3)$$

Equation (1) can be rearranged to,

$$\frac{1}{\text{image distance (m)}} = \left(\frac{1}{\text{focal length (m)}} \right) - \left(\frac{1}{\text{blow distance (m)}} \right) \quad (4)$$

Equation (2) can be arranged to,

$$\begin{aligned} \text{blow height (m)} &= \left(\frac{1}{\left(\frac{\text{image height (m)}}{\text{blow height (m)}} \right)} \right) \times \left[\left(\frac{\text{image height (m)}}{\text{blow height (m)}} \right) \right. \\ &\quad \left. \times \left(\frac{\text{blow distance (m)}}{\text{image distance (m)}} \right) \right] \end{aligned} \quad (5)$$

and equation (3) can be rearranged to,

$$\text{image height (m)} = \text{pixel pitch} \left(\frac{\text{m}}{\text{pixel}} \right) \times \text{image height (pixel)}. \quad (6)$$

Substituting equations (4) and (6) into equation (5) gives,

$$\begin{aligned} \text{blow height (m)} &= \left[\left(\frac{1}{\left(\frac{\text{focal length (m)}}{\text{image distance (m)}} \right)} \right) - \left(\frac{1}{\text{blow distance (m)}} \right) \right] \\ &\quad \times \left[\left(\frac{\text{pixel pitch} \left(\frac{\text{m}}{\text{pixel}} \right) \times \text{image height (pixel)}}{\text{blow height (m)}} \right) \right. \\ &\quad \left. \times \left(\frac{\text{blow distance (m)}}{\text{image distance (m)}} \right) \right] \end{aligned} \quad (7)$$

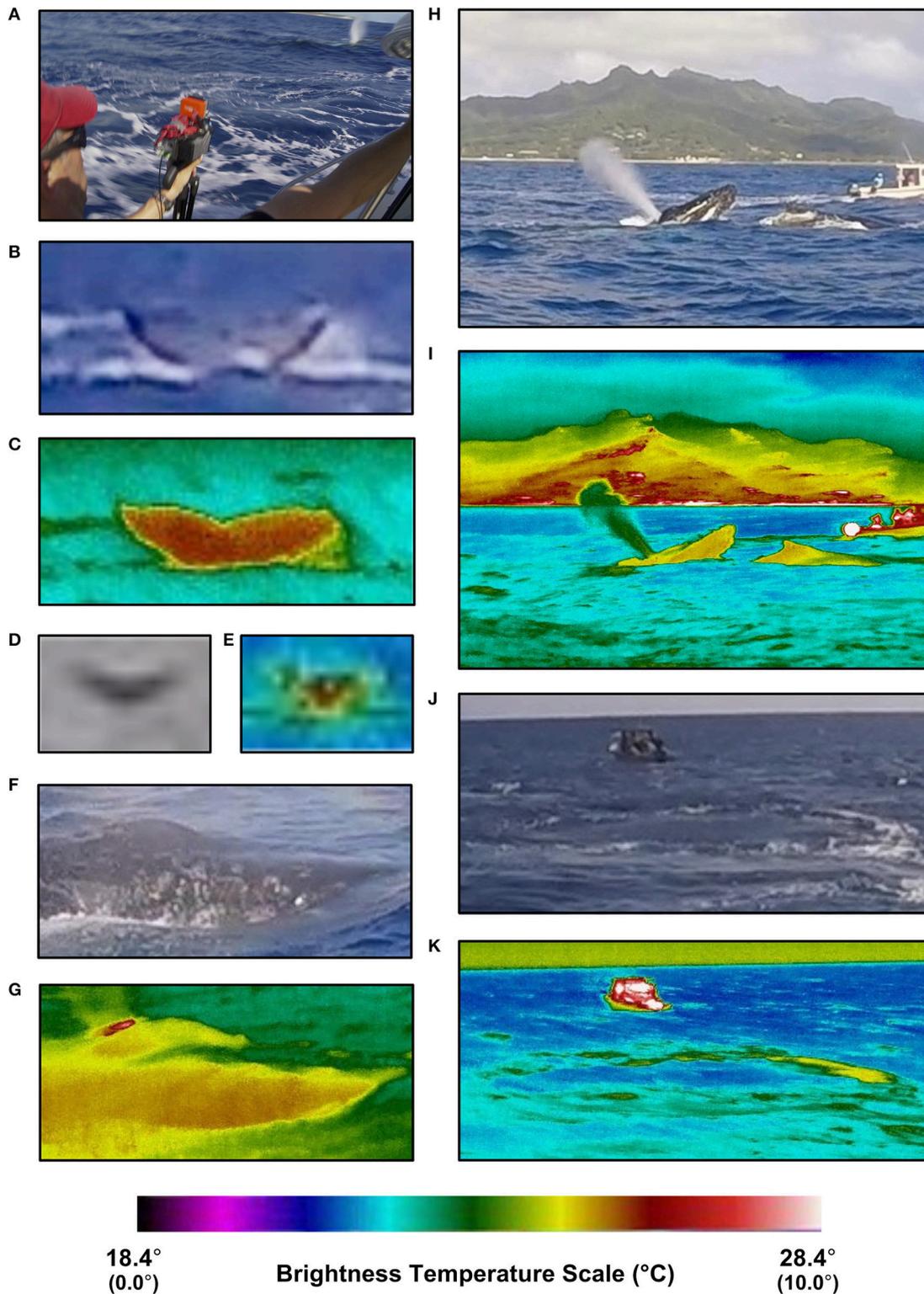


FIGURE 1 | Visible and infrared spectrum images of various humpback whale surfacing features. Visible and thermal camera set-up (A); visible (B) and infrared (C) images of a fluke at 100m distance in tropical waters; visible (D) and infrared (E) images of a fluke at ~350m distance in sub-polar waters; visible (F) and infrared (G) images of a nostril and adjacent rostrum at ~10m distance in tropical waters; visible (H) and infrared (I) images of a blow, rostrum and dorsal fin at 40m distance in tropical waters; visible (J) and infrared (K) images of a footprint at ~30m distance and 50s following fluke in tropical waters. Temperature scale numbers in parentheses (i.e., 0°–10°C) correspond with sub-polar thermal image brightness temperatures shown in (E).

which simplifies to,

$$\text{blow height (m)} = \frac{\left[\left(\frac{\text{pixel}}{\text{pitch}} \left(\frac{\text{m}}{\text{pixel}} \right) \times \text{image height (pixel)} \times \frac{\text{blow (m)}}{\text{distance (m)}} \right) \right]}{\text{focal length (m)}} - \left(\frac{\text{pixel}}{\text{pitch}} \left(\frac{\text{m}}{\text{pixel}} \right) \times \frac{\text{image height (pixel)}}{\text{height (m)}} \right). \quad (8)$$

Blow heights were estimated using Equation (8) every 0.04 s following blow initiation. Image pixel heights were measured using FLIR Tools+ and blow distances were measured by laser range-finding as described above.

RESULTS

In total, we determined BTA profiles for 174 humpback whale blows, 20 dorsal fins, 9 flukes, and 20 rostrums. An equivalent number of whale features were analyzed from each of the two study areas, with the exception of flukes, for which 6 were imaged in Alaska and only 3 were imaged in Rarotonga. Of the 87 blows analyzed in each study area, 32 Rarotonga blows and 16 Alaska blows were imaged at distances <150 m. Of these, only 10 blows from each study area were recorded in the 100–150 m range.

Average BTA profiles demonstrate that humpback whale blows, dorsal fins, flukes and rostrums appear as thermal anomalies of similar magnitude relative to adjacent ocean water (Figure 3). For example, 100–150 m distant blows in Rarotonga and Alaska appear as 20–30 pixel-wide thermal anomalies that are ~3°C warmer than the adjacent ocean (Figure 3A). Similarly, dorsal fins and flukes in both areas exhibited maximum BTA values ca. 3–4°C (Figures 3B,C), whereas rostrums from both populations were ~2–3°C warmer than the adjacent ocean (Figure 3D). Ocean water temperatures were measured by perpendicular thermography and satellite observations in both study areas. These measurements indicate surface ocean water temperature was ~24°C in Rarotonga, and ~8°C in Sitka Sound, Alaska, at the time thermal images were recorded.

The shapes of the average dorsal fin, fluke and rostrum BTA profiles differ because these features were recorded across a large range of distances in each study area. Because the Rarotonga whales were generally imaged at closer ranges, the dorsal fin, fluke and rostrum BTA profiles are spread across a larger number of image pixels than the Sitka BTA profiles (Figures 3B,D). In other words, the Rarotonga whale features fill a larger portion of the 640 × 480 pixel thermal images because these images were recorded at closer distances. Despite these distance-related differences in BTA profile shape between the study areas, the maximum BTA values for humpback whale blows, dorsal fins, flukes and rostrums (indicated by arrows in Figure 3D) we recorded are not significantly different ($p \gg 0.05$, two-tailed t -test, Figure 3).

Laser range-finding enabled quantification of the relationship between the pixel-length of individual blows and blow distance for the FLIR A615. As expected, blow pixel-lengths are larger for images recorded at closer range, and blow pixel-length decreases

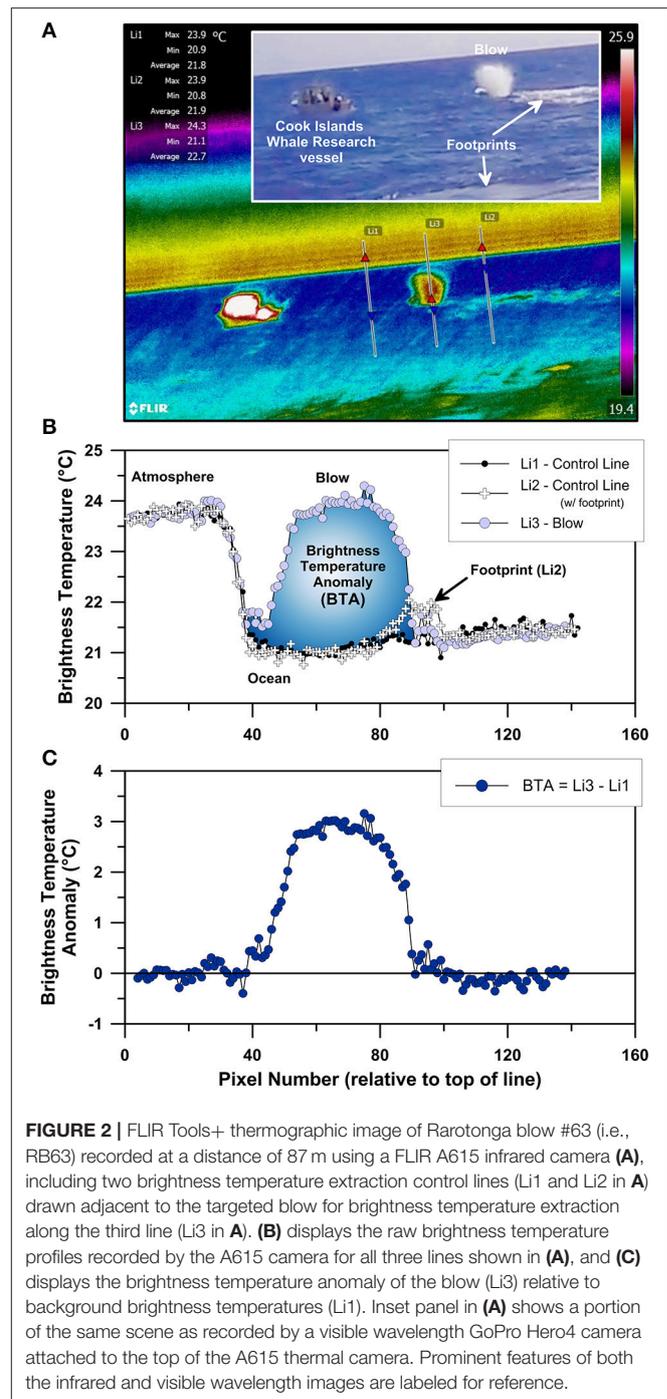


FIGURE 2 | FLIR Tools+ thermographic image of Rarotonga blow #63 (i.e., RB63) recorded at a distance of 87 m using a FLIR A615 infrared camera (A), including two brightness temperature extraction control lines (Li1 and Li2 in A) drawn adjacent to the targeted blow for brightness temperature extraction along the third line (Li3 in A). (B) displays the raw brightness temperature profiles recorded by the A615 camera for all three lines shown in (A), and (C) displays the brightness temperature anomaly of the blow (Li3) relative to background brightness temperatures (Li1). Inset panel in (A) shows a portion of the same scene as recorded by a visible wavelength GoPro Hero4 camera attached to the top of the A615 thermal camera. Prominent features of both the infrared and visible wavelength images are labeled for reference.

with blow distance according to an inverse power relationship (Figure 4). Although blows imaged at <200 m range were easily recognizable with the A615 (Figures 4A,B,D), a blow imaged at ~400 m range appeared as an 8 pixel tall ~0.4°C BTA (Figure 4E). Higher sensitivity cooled detector thermal imaging devices and/or devices with longer focal lengths would no doubt extend the range at which whale blows might be detectable (e.g., Zitterbart et al., 2013). However, these larger systems are

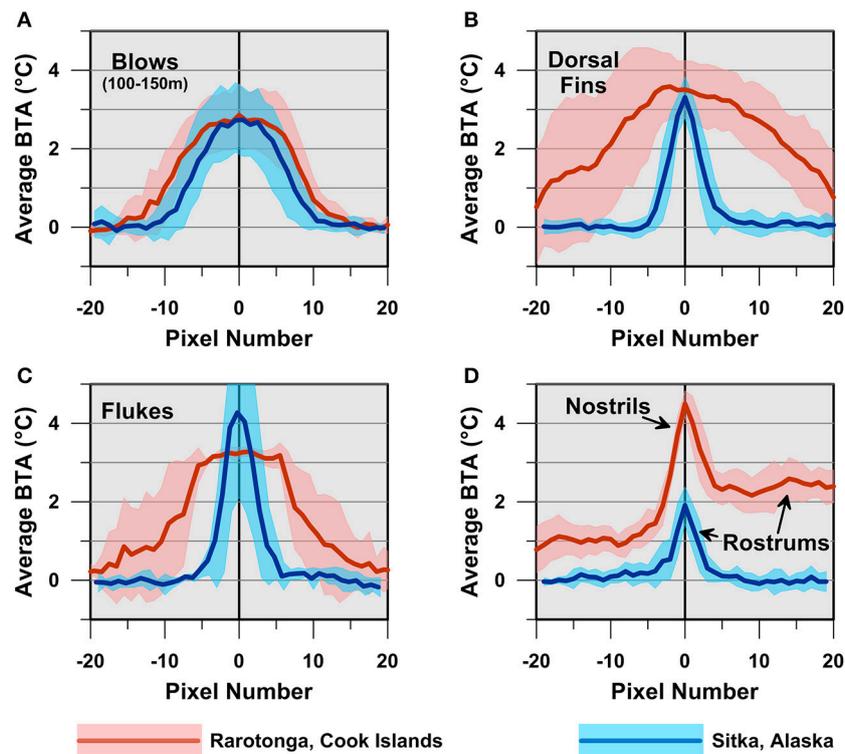


FIGURE 3 | Thermographic profiles of emergent humpback whale blows and body parts in both tropical (Rarotonga, Cook Islands) and sub-polar (Sitka Sound, Alaska, U.S.A.) waters. Each plot presents the average brightness temperature difference relative to background ocean water brightness temperatures for blows (A), dorsal fins (B), flukes (C), nostrils and rostrums (D). Only rostrums were imaged in Alaska, whereas both nostrils and adjacent rostrums were imaged in Rarotonga. Ten individual profiles from each of the study areas were used to construct the average brightness temperature anomaly (BTA) profiles shown, with the exception of flukes, where only 6 fluke profiles were recorded in Alaska and 3 fluke profiles in Rarotonga. Shaded regions correspond with ± 1 SD brightness temperature variation. Individual profiles were aligned such that the image pixel with the largest brightness temperature difference relative to seawater was assigned pixel number zero. Negative pixel numbers correspond with pixels that are skyward of the maximum brightness temperature difference pixel. Positive pixel numbers correspond with pixels that are seaward of the maximum brightness temperature difference pixel. The parabolic shape of each average profile reflects the fact that the individual datasets used to determine the average profiles shown were imaged at different distances with correspondingly different image pixel widths/lengths. For example, because the Rarotonga rostrums were imaged at closer range than the Sitka rostrums, the Rarotonga rostrums span a much larger number of pixels and include positive thermal anomalies across the nostrils that were not captured in any of the Sitka images (D).

currently much more expensive and less maneuverable than the FLIR A615 we used here.

Regardless of the device used or its imaging range, whale blow heights will also vary in response to a number of uncontrollable factors, including: wind shear, the volume of sea water in the nasal depression at exhalation, and the whale's position relative to the ocean surface at which exhalation is initiated. In an effort to partially overcome these complicating factors, we calculated blow heights 0.4 s after blow initiation, the minimum observed period for a blow to achieve its maximum height, for 32 humpback whale blows across an 18–140 m range in distances (mean = 71 ± 38 m, \pm SD, Figure 4). The pixel height (range = 24–230 pixels, mean = 63 pixels \pm 46 pixels, \pm SD, Figure 4) of each imaged and laser ranged blow was measured using FLIR Tools+. Estimated blow heights at 0.4 s ranged between 1.0 and 3.3 m (mean = 2.2 ± 0.5 m, \pm SD, $n = 32$). In addition to wind, water volume, and whale position, blow heights are also likely to vary with the volume of air being expelled in a specific exhalation. Although untested, focal follows incorporating thermal imaging

techniques have the potential to reveal the breathing behaviors of individual whales of different size, maturity, sex and physiological condition.

Utilization of the 25 frames per second videography option enabled us to also estimate humpback whale blow velocity (Figure 5). All blows analyzed reached maximum blow height in <1.2 s and the maximum blow height measured was 4.7 m at 0.8 s following blow initiation equating to a 21 km h^{-1} velocity for this blow (Figures 5G,J,K). Notably, some blows were unambiguously initiated while the nostrils/blowholes were still submerged. Blows of this type exhibited a relatively slow initial acceleration (e.g., Figures 5G–I) as the exhaled air pushed into the overlying water column. Individual blows exhibited maximum blow velocities that ranged between 40 and 55 km h^{-1} (mean = $13\text{--}23 \text{ km h}^{-1} \pm 12\text{--}18 \text{ km h}^{-1}$, \pm SD). Maximum blow heights ranged between 2.7 and 4.7 m and occurred 0.76–1.16 s following blow initiation. At 0.4 s following exhalation initiation, the humpback whale blows we recorded were 1.4–3.3 m tall. It is important to acknowledge that these estimates are derived

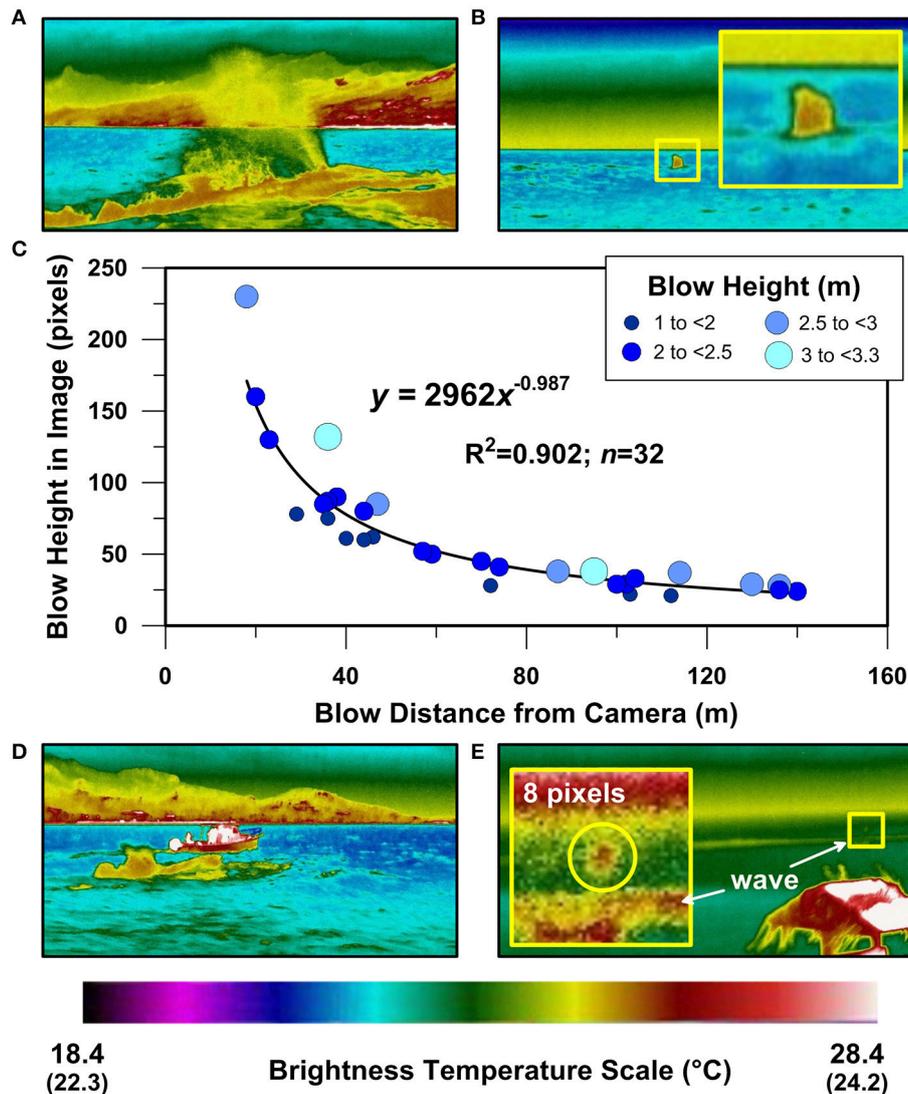


FIGURE 4 | Rarotonga humpback whale blow thermography at different distances. Infrared thermographic images of humpback whale blows 0.4 s after blow initiation for whales at 18 m (A), 136 m (B), 44 m (D), ~400 m (E). (C) displays the relationship between blow pixel length as determined from individual thermographic images and blow distance as measured using a laser rangefinder in the field. Each blow has been symbolized according to its estimated blow height at 0.4 s after blow initiation as indicated in the legend. Inset panel in (B) displays a magnified perspective of a humpback whale blow imaged from a distance of 137 m. Inset panel in (E) displays a magnified perspective of an 8 pixel high blow imaged at dusk from a distance of ~400 m at Tuoro/Black Rock on the northwest coast of Rarotonga. Temperature scale numbers in parentheses (i.e., 22.3°–24.2°C) correspond with the thermal image brightness temperatures shown in the magnified inset panel in (E).

from the thermal anomalies associated with water droplets that are blasted out of the ocean's surface or nasal depression by exhaled air. Thus, the velocities we calculated must be considered minimum estimates of the true gaseous exhalation velocities achieved by humpback whales.

Our results demonstrate that humpback whale blows, dorsal fins, flukes and rostrums present as similar magnitude brightness temperature anomalies (BTA) in both tropical (Rarotonga, Cook Islands) and sub-polar (Sitka Sound, Alaska, U.S.A.) environments despite an ~16°C difference in ocean surface temperature between the two study areas. This occurs

due to emissivity effects associated with the oblique near-horizontal imaging angles used in the current study. Thus, absolute temperatures determined from oblique (i.e., sub-parallel to target) measurement angles do not represent accurate quantifications of whale blow or skin temperatures. Our results also demonstrate how to calculate blow heights and blow velocities by combining target BTA pixel size with target distance as measured by a laser range finder. Although blow acceleration varied both within and between individual blows, our results indicate that humpback whale blows have average instantaneous velocities of ~4.6 m s⁻¹.

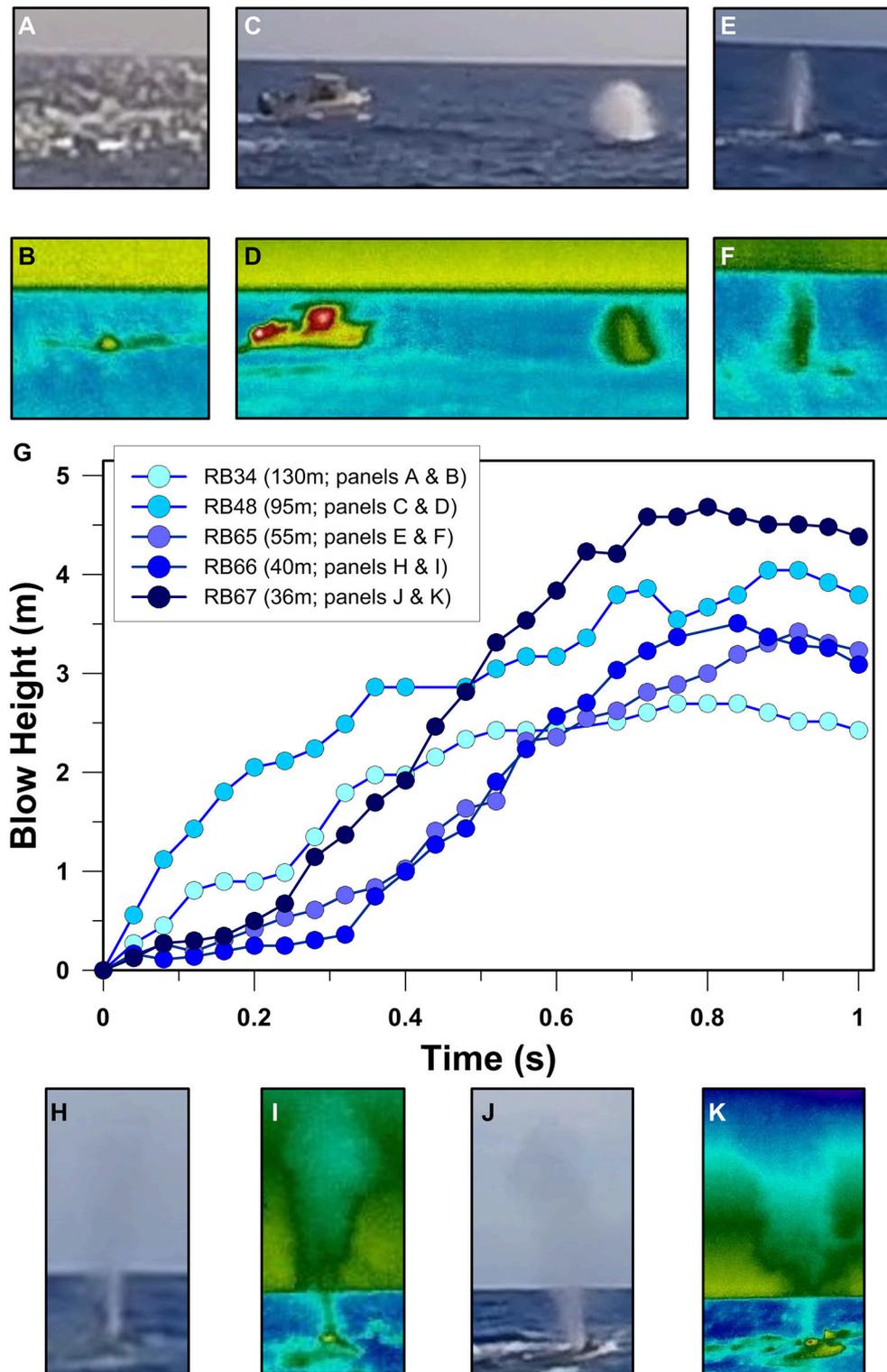


FIGURE 5 | Humpback whale blow evolution through time. Visible (A) and infrared (B) images of whale blow RB34 0.16 s after blow initiation at a distance of 130 m. Visible (C) and infrared (D) images of whale blow RB48 0.36 s after blow initiation at a distance of 95 m. Visible (E) and infrared (F) images of whale blow RB65 0.48 s after blow initiation at a distance of 55 m. Visible (H) and infrared (I) images of whale blow RB66 0.64 s after blow initiation at a distance of 40 m. Visible (J) and infrared (K) images of whale blow RB67 0.80 s after blow initiation at a distance of 36 m. (G) displays blow height vs. time (i.e., velocity profiles) for the blows indicated in the legend.

DISCUSSION

The infrared radiation emitted by a surface is a function of both the surface's temperature and its spectral emissivity (see Nomenclature). Thermal imaging systems estimate surface temperatures by assigning emissivity values to the imaged scene. However, sea surface emissivity, the ratio of the energy radiated from the ocean's surface relative to a blackbody, further depends on the ocean's surface roughness, refractive index, and the zenith angle from which the surface is being observed (Masuda et al., 1988). Thus, quantitative analyses and accurate interpretations of thermographic datasets collected at sea depend on a large number of variables.

Of these variables, the angle from which the surface is being observed has the largest effect on emissivity and, as a consequence, thermographic temperature estimates (Masuda et al., 1988). For example, the emissivity of perfectly planar sea water at a 0° zenith angle (i.e., perpendicular to the sea surface) is ~0.98 (Masuda et al., 1988). At a 60° zenith angle this same surface will have an emissivity of ~0.92 and at 85° (i.e., 5° above the horizontal) the emissivity drops to ~0.36 (Masuda et al., 1988). Using human targets included in our thermographic image dataset, we found that a decrease in surface emissivity of 0.98–0.36 resulted in a 12.2°C increase in the human skin surface temperature reported by the camera. Similar tests on 25°C ocean water revealed that a similar magnitude change in emissivity resulted in a 3.5°C change in sea surface temperature at ~100 m distance. As suggested by Cuyler et al. (1992), our findings confirm it is inappropriate to assume relatively high emissivity values (i.e., >0.95) in thermographic cetacean research when imaging is performed at high zenith angles.

Thus, the data we report suffers from extreme emissivity effects due to the fact that our thermal images were collected at sub-horizontal observation angles (i.e., zenith angles of ~85°–89°). However, the A615 infrared camera we used includes a high sensitivity microbolometer (<0.05°C); thus, the brightness temperature measurements we report can be considered precise but not accurate. Although the loss of thermographic accuracy due to emissivity effects associated with oblique-angle imaging is problematic for biometrical estimates of cetacean thermoregulation, it is a benefit to cetacean detection.

Brightness temperature anomalies of ~2–4°C, like those we report for humpback whale blows, dorsal fins, flukes and rostrums (Figure 3), are the consequence of rapid changes in emissivity as the whale feature emerges from the ocean's surface and immediately changes the observation point zenith angle. However, the higher BTA values we report for humpback whale nostrils (ca. 4.5°C, Figure 3D) likely reflect a more accurate approximation of humpback whale skin temperatures due to the closer range at which nostrils were imaged (i.e., at lower zenith angle) and the observed ~2°C difference between nostril/blowhole temperatures and adjacent (wet) rostrums (Figure 2D). The potential utility of thermographic imaging of cetacean nostrils/blowholes for biometrical research purposes should be more deeply explored using aerial drones mounted with high resolution and high frame-rate thermal imaging systems.

One of the primary challenges in cetacean ecology and conservation is determining when and where whales are located. Although our results do not include accurate determinations of whale surface temperatures, they conclusively demonstrate that whale blows and emergent body parts appear as similar magnitude thermal anomalies, ca. 2–4°C, relative to surface waters in both tropical and sub-polar environments at distance ranging between 100 and 150 m. These thermal anomalies are largely due to emissivity effects associated with thermographic imaging from sub-horizontal positions. Thus, our findings represent an important quantification of the magnitude of the thermal signal from which thermographic cetacean detection algorithms can be developed and refined.

Quantitative constraints on the magnitude, size and duration of whale-derived thermal anomalies can also be used to help restrict the number of false positives and false negatives produced by automated cetacean detection systems that use transient thermal contrast algorithms based on average brightness temperatures (e.g., Zitterbart et al., 2013). Improving automated detection systems in this way should assist applications in windy conditions or large swells, when ocean surface roughness has the potential to produce thermal anomalies of similar magnitude as whale blows due to emissivity effects (e.g., Figure 4E). Differentiating cetacean induced anomalies from non-cetacean induced anomalies will also benefit from quantifications of thermal anomaly shapes and their evolution through time. For example, our results demonstrate that the water spouts produced by humpback whale exhalations move at ~4.6 m s⁻¹ and accelerate at ~100–300 m s⁻². Such biometrical measurements not only provide additional quantifications for the development of automated cetacean detection systems, but also create a platform for species-level identifications using measurable differences in blow geometry and velocity.

AUTHOR CONTRIBUTIONS

TH and PZ-R conceived of the study. All authors contributed to the fieldwork and infrared imaging in Rarotonga, Cook Islands. TH performed the fieldwork and infrared imaging in Sitka Sound, Alaska, U.S.A. TH and AO performed all of the thermographic image processing and analysis. TH and AO wrote the initial manuscript and all authors contributed to the revision of the manuscript.

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Conflict of Interest Statement: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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NOMENCLATURE

Brightness temperature—temperature measured by the thermal imaging device.

Brightness temperature anomaly (BTA)—difference between the brightness temperature of the targeted object and the brightness temperature of the background scene.

Emissivity (spectral) – the ratio of the energy radiated from a surface to the energy radiated from a blackbody at the same temperature, wavelength and environmental conditions.

Microbolometer – the detector in a thermal imaging device (for further details see: Ostrower, 2006).

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Cumulative Effects Assessment for Northern and Southern Resident Killer Whale Populations in the Northeast Pacific

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Foreword

This series documents the scientific basis for the evaluation of aquatic resources and ecosystems in Canada. As such, it addresses the issues of the day in the time frames required and the documents it contains are not intended as definitive statements on the subjects addressed but rather as progress reports on ongoing investigations.

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ABSTRACT

The Northern and Southern Resident Killer Whale populations (NRKW and SRKW) that inhabit the waters of the Canadian Pacific coast are listed as Threatened (NRKW) and Endangered (SRKW) under the Species at Risk Act (SARA). The SARA recovery plan developed for these populations identified the assessment of the cumulative effects of anthropogenic threats impacting these populations as a high priority. To address this, a cumulative effects assessment framework was developed and applied comprising two components: a Pathways of Effects (PoE) conceptual model and a subsequent Population Viability Analysis (PVA) model. The PoE model summarises the current understanding of each priority threat (prey availability, disturbance and contaminants) and describes the structure of the threats in the assessment, including threat interactions and potential impacts to population parameters (fecundity and mortality). The PoE model forms the basis for the subsequent PVA model, which utilises the most recent available threat data to quantify the way threats impact population parameters and, together with demographic data, explore patterns of population growth and decline in different threat scenarios. The impacts of individual and cumulative threat scenarios on modelled SRKW and NRKW populations were compared to the observed population trajectories (2000-2017) in order to define a model that best captured the real world dynamics of the two populations. Of the various individual and combined threat models tested, the cumulative threats model, which incorporated all priority threats (Chinook salmon abundance, vessel noise/presence, vessel strike, and PCB contamination), predicted demographic rates closest to that observed for both populations. Population dynamics predicted by the model closely followed the observed demographics for NRKW and though it was the closest model to the observed population size for SRKW, it did not include the observed values within the bounds of uncertainty. However, when historical Chinook salmon model data were included in the model prediction, rather than a randomly chosen Chinook salmon index value, the fit improved for SRKW and the uncertainty bounds of both models included the observed values, suggesting that the cumulative model is a useful representation of the system.

The findings of this cumulative effects assessment highlight the importance of considering threats collectively. Specifically, within the cumulative effects PVA assessment, Chinook salmon abundance and its interactions with vessel noise/presence and PCBs strongly influenced modelled killer whale population dynamics. The cumulative effects PVA model was also used to project population trajectories for NRKW and SKRW into the future. The model outputs indicate that the mean modelled NRKW population trajectory increased to the carrying capacity set in the model within 25 years. In contrast, the mean modelled SRKW population trajectory declined, with a 26% probability of population extinction (defined in the model as only one sex remaining), and in those projections, extinction was estimated to occur after 86 (\pm 11) years. The cumulative effects assessment framework developed, that combines a PoE with a PVA model, is a novel approach that explicitly identifies and quantifies threat linkage pathways, and associated uncertainties. The framework is a potentially useful tool for managers and scientists and has been refined and tested with the latest threat information for these populations but could also be applied to other populations and species. It is cautioned that as model outputs are only as good as the model inputs, changes in exposure to natural and anthropogenic threats can affect the model's accuracy. An iterative approach should be used so that model inputs and structure are regularly reviewed and updated to include new information about existing threats and the addition of new threats as knowledge is increased on these populations.

1 INTRODUCTION

1.1 BACKGROUND

Three genetically and acoustically distinct killer whale (*Orcinus orca*) ecotypes inhabit the waters of the Northeast Pacific coast of North America: offshore, Bigg's (or transient), and residents (Ford et al. 1998). The resident fish-eating ecotype is further divided into the Northern and Southern Resident Killer Whale (NRKW and SRKW) and the Southern Alaskan Resident Killer Whale populations (SARKW) (Ford et al. 2000; Matkin et al. 1999; 2014). Though all populations of Resident Killer Whales are fish-eating cetaceans, feeding primarily on Chinook (*Oncorhynchus tshawytscha*) and Chum salmon (*O. keta*), and overlap to some extent in habitat and diet, they do not interact with one another socially and are distinct in terms of their culture, acoustics, and genetics (DFO 2017a).

The NRKW and SRKW populations were listed as Threatened (NRKW) and Endangered (SRKW) in Schedule 1 under the Species at Risk Act (SARA) in 2003. The NRKW range includes the coastal waters from Glacier Bay (Alaska, USA) to Gray's Harbor (Washington State, USA), and the SRKW range extends from southeastern Alaska to central California (Ford et al. 2000, 2006).

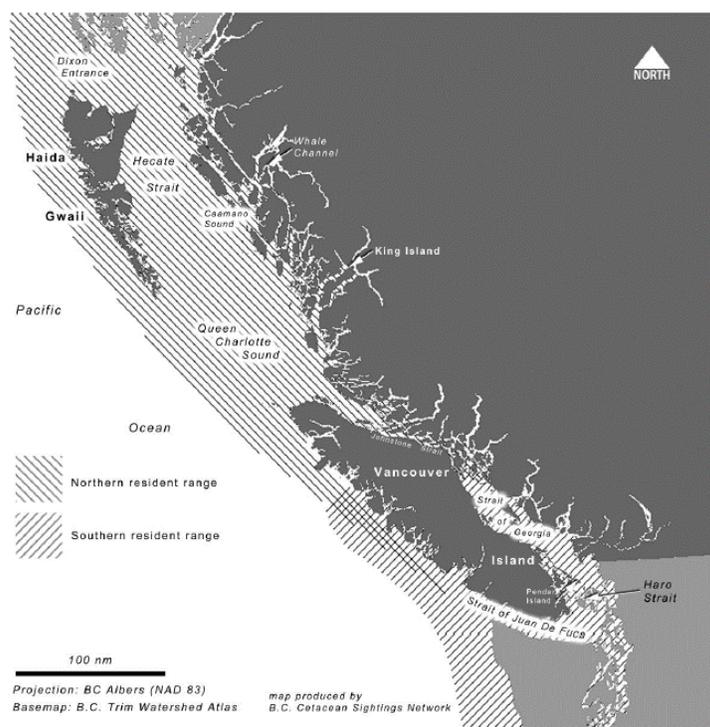


Figure 1 - Overlapping ranges of Northern Resident and Southern Resident Killer whales centered in Canadian waters (after DFO 2018a).

In summer, the movements and habitat use by Resident Killer Whale populations often reflects the timing and locations of inbound Pacific salmon migrations. The SRKW population tends to concentrate in the waters of southern Vancouver Island and northern Washington State while a portion of the NRKW population are often found frequenting Johnstone Strait and Queen Charlotte Strait (Figure 1; DFO 2018a). The SRKW range has a higher overlap with major coastal population centres (Vancouver and Seattle) than the NRKW population and

consequently, is more likely to be exposed to, and potentially impacted by anthropogenic threats such as contaminants and vessel related threats (noise and physical disturbance) (Krahn et al. 2004; Wiles 2004; Figure 1).

1.1.1 Population Trends

Long-term photo-identification census surveys for both populations were initiated by Michael Bigg in the 1970s and continue to the present day (DFO Cetacean Research Program; Center for Whale Research, CWR). The SRKW census (begun in 1976) is considered to be more accurate than the NRKW census (begun in 1973), as not all members of the Northern population are seen each year (DFO 2018a).

Population trends based on the census data indicate that the SRKW population has experienced an overall negative population growth rate (-0.002; 1979-2017), but experienced particularly sharp declines between 1995 and 2001 (Figure 2). Since then, the population has shown little recovery, having 77 members in December 2017. In contrast, the NRKW population has experienced a steady increase over the census period (population growth rate = 0.02; 1979-2017), except for a decline between 1997 and 2001 (Figure 2). The population has since increased from 219 members in 2004, to 308 members in 2017 (41% increase).

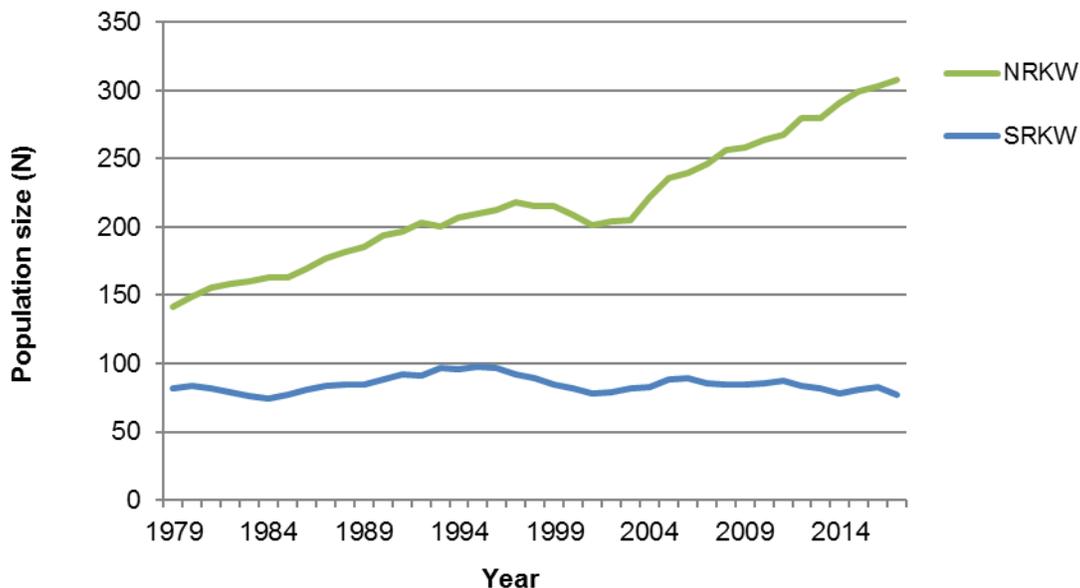


Figure 2 – Resident killer whale population time series (data shown 1979-2017).

1.1.2 Goal of the Assessment

The Southern and Northern Resident populations were listed as Endangered and Threatened, respectively, under SARA in 2003. Under SARA, the federal government has a commitment to prevent wildlife species from being extirpated or becoming extinct, to provide for the recovery of wildlife species that are Extirpated, Endangered or Threatened as a result of human activity and to manage species of Special Concern to prevent them from becoming Endangered or Threatened. The Minister of Fisheries and Oceans Canada (DFO) is the competent minister for the recovery of aquatic species at risk.

The three primary stressors (from hereon referred to as threats) to NRKW and SRKW identified by the DFO Species at Risk Program (COSEWIC 2009; DFO 2011; DFO 2017b) are:

-
1. Reduced prey availability,
 2. Acoustic and physical disturbance, and
 3. Environmental contaminants

There is a legal requirement to assess cumulative effects within the Canadian SARA action plan for NRKW and SRKW (DFO 2017b). Three of the 98 Recovery Measures (RMs) in the action plan relate (directly or indirectly) to cumulative effects (RM 6, 11, 17), and the focus of this assessment is to address RM 11 (Table 1).

Table 1 - SARA Recovery Measures for Resident Killer Whales related to cumulative effects.

#	Recovery Measure	Priority
6	Take into account both the seasonal (acute) as well as the cumulative (chronic) effects of poor returns for Chinook and other important prey species on Resident Killer Whales when managing fisheries.	High
11	Assess cumulative effects of potential anthropogenic impacts on Resident Killer Whales using an appropriate impact assessment framework for aquatic species.	High
17	Review and assess project impacts on Resident Killer Whales and their habitat, and provide advice on avoidance and mitigation measures as required.	High

1.2 WHY A CUMULATIVE EFFECTS ASSESSMENT IS REQUIRED

DFO's Species at Risk Program has requested that the Science Branch provide an assessment of the cumulative effects of the three primary anthropogenic threats on NRKW and SRKW populations. To date, most research on threats to killer whales has studied these threats in isolation, for instance focusing solely on prey availability or acoustic disturbance. Assessment of cumulative effects, however, involves examining the combined, incremental impacts that threats from multiple human activities can have on individuals, populations, communities and ecosystems through space and time. Cumulative effects assessments evaluate the effects of multiple threats by converting impacts into a single currency or metric, thereby allowing for comparisons among threats and their combined long-term impact.

This study provides an opportunity to incorporate the best available scientific information into a single assessment that includes all three threats, the interactions between these threats, and the resulting long-term impacts on the population. Previous cumulative effects assessments (CEAs) fall into three categories: risk assessment, statistical analysis, and population viability analysis (Lawson and Lesage 2012; O et al. 2015; Williams et al. 2017; Lacy et al. 2017). An example of risk assessment is the general framework to evaluate the relative additive cumulative risk for a range of activities and ecosystem components that was developed by O et al. (2015). Statistical models have been used to evaluate the impact of single threats on mortality and fecundity of Resident Killer Whales (Ward et al. 2009; Vélez-Espino et al. 2014b). Williams et al. (2017) developed a cumulative population viability analysis (PVA) model to quantify factors limiting the recovery of the St. Lawrence Estuary Beluga population and Lacy et al. (2017) evaluated the cumulative effects of anthropogenic threats on SRKW using a PVA.

A DFO framework has been developed to quantify and cumulate risks of impacts for marine mammal populations, the Cumulative Effects Risk Assessment Framework (CERAF) (Lawson and Lesage 2012; DFO 2017c; Figure 3). While the current work does not explicitly utilise the CERAF, for comparison, the current work fits within the CERAF steps (Lawson and Lesage 2012; DFO 2017c). The scoping and relative risk phases (Box A, B, C) were conducted previously, either explicitly or implicitly in the various recovery documents that identified the most important threats to SRKW and NRKW (Figure 3). The focus of the current work corresponds to the last step in the CERAF (Box D), i.e., taking the highest risk threats and assessing them together in a viability analysis to investigate the cumulative effects on the long-term persistence of the population.

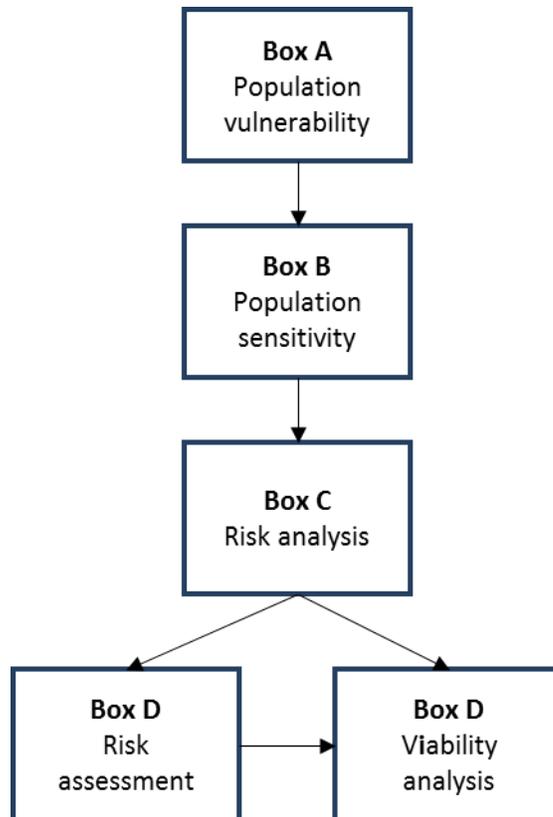


Figure 3 - Structure of the Cumulative Ecological Risk Assessment Framework (CERAF), adapted from DFO 2017c.

The PVA method was selected as the most appropriate assessment to use to address the recovery measure (RM 11) as it incorporates the required threats and has been applied to a number of species, including SRKW. Relatively minor modifications were necessary to adapt the PVA to also assess NRKW. In addition, the software used to carry out the PVA (Vortex) is open access and available for use (Lacy and Pollak 2014).

1.3 CUMULATIVE EFFECTS ASSESSMENT FOR RESIDENT KILLER WHALES

The current cumulative effects assessment consists of two phases. First, a Pathways of Effects (PoE) conceptual model describes the impacts of threats on the mortality and fecundity of the species. As threats can interact over space and time, altering their respective intensities and consequent effects on individuals and populations, this study will also assess potential

interactions between threats to more accurately represent the natural system. The outputs of the PoE conceptual model were used to design and refine the PVA model in the next phase.

In the second phase, impacts are parameterised (e.g., effect size for each threat and its impact on vital rates) and a quantitative PVA is conducted to assess the cumulative effects, building upon the methods and results of previous work (Taylor and Plater 2001; Ward et al. 2009; Vélez-Espino et al. 2014b; Williams et al. 2017; Lacy et al. 2017; DFO 2018a). Existing literature and data are used to parameterise the impact of each threat on killer whale vital rates and previously published relationships are updated with recent data and re-analysed. These quantitative values and relationships specific to each population (SRKW and NRKW) are used to define the inputs to a population model describing the combined impact on population persistence through time. The model structure builds upon an existing PVA model developed for SRKW by Lacy et al. (2017). To capture the unique population structure and threat exposure, a PVA model is run for each population separately (SRKW and NRKW).

An overview of the steps used in the current work to assess cumulative effects on NRKW and SRKW is outlined in Figure 4.

1.3.1 Assumptions for this Assessment

- The mechanisms and consequences of threats on individuals are assumed to be the same for both Resident Killer Whale populations, whereas the level of exposure to threats is assumed to be population-specific.
- Impacts from threats to population vital rates (mortality and fecundity), based on the best available information at the time of the assessment, are assumed to be accurately described in the Pathways of Effects model
- The analysis assumes impacts only from the focal threats examined (reduced prey availability, disturbance, and contaminants), and does not consider other threats and the effects of broader impacts such as changing climate conditions and increasing human populations.
- The way that impacts are parameterised in the PVA model is assumed to represent the impacts of the entire threat (e.g. impacts of PCBs represents the Contaminants threat). Specific limitations and assumptions for each threat are described in detail in section 5.3.
- Information used to quantify threat impacts to vital rates was assumed to represent threat conditions throughout the range and throughout the year, despite primarily being obtained in the Salish Sea area in the summer/fall period.
- The assessment assumes no spatial or temporal variation at the sub-population level, even though variation between matrilineal groups could affect their exposure to threats.
- Projections of the RKW population dynamics into the future assume that current threat levels remain the same. It is also assumed that that no threat mitigation measures and management actions are taken.
- The population model chosen to be used for predictions is assumed to be an effective surrogate for the real population dynamics for SRKW and NRKW populations.

Cumulative Effects Assessment Framework Steps

1. Scoping

- a. The goal of the assessment (e.g., population recovery)
- b. The range of the population(s) to be assessed
- c. The threats (stressors) to the population to be considered
- d. The existing knowledge of the population(s)
- e. Desired endpoint measures (e.g., population size, birth rate, etc.)

Scoping

2. Develop a Pathways of Effects (PoE) Conceptual Model

Create a Pathways of Effects model linking the threats to the endpoint measures. Provide supporting information/data sources for each linkage pathway in the model, including expert elicitation where published information is unavailable. Document and incorporate levels of uncertainty. If possible, determine potential interaction types among threats (e.g. synergistic, additive etc.). Assume only additive if not.

Pathways of Effects Model (PoE)

3. Population Model and Scenario Testing

Quantify threat-species relationships identified in conceptual model through literature review, expert elicitation, and quantitative models. Obtain demographic data on population. Input data into population model and test the outcomes of different threat combinations on endpoint measures (to look at differing levels of population recovery, allowable harm).

Population model scenarios

Figure 4 – Steps in the current cumulative effects assessment framework.

1.4 OBJECTIVES

The current working paper has four major objectives:

1. Develop a Pathways of Effects (PoE) conceptual model to visually represent threat-impact pathways, limited to the primary threats identified by the Species at Risk Program (DFO 2017b). Provide supporting text to accompany the PoE model diagram to describe and justify the linkage pathways presented and explain how threats act on population parameters based on evidence in the literature and elsewhere. The PoE conceptual model will be generic to include both SRKW and NRKW populations, as the mechanisms of impact

are expected to be similar in both populations but the levels of exposure to each threat are different.

2. Quantify threat linkage pathways identified in the PoE model by determining the best available and most recent data or information from data mining, literature review and expert elicitation. This information will be used to develop and parameterise a quantitative PVA model.
3. Assess the cumulative effects acting on Resident Killer Whales by running single and cumulative PVA model scenarios to evaluate and compare the effects of each scenario to the observed population dynamics.
4. Identify uncertainties in data and methods and highlight knowledge gaps

1.5 SCOPE OF THIS STUDY

The aim of the current assessment is to evaluate the cumulative effects of anthropogenic threats on Resident Killer Whales. The study is limited to considering the primary threats identified in the (SARA action plan for NRKW and SRKW (DFO 2017b, 2018a). The effects of low probability but high impact events, such as catastrophic oil spills, are not included in the current assessment. Future changes in anthropogenic activities are not included or assessed. Potential mitigation measures and management actions will not be evaluated, but this assessment can be used as a tool to evaluate future changes and mitigation measures once the cumulative effects model has been reviewed.

2 PATHWAYS OF EFFECTS CONCEPTUAL MODEL

2.1 DEFINITION, STRUCTURE AND OUTPUTS

PoE models are widely used conceptual modelling tools that can guide assessments by providing a science-based foundation for decision-making (Government of Canada 2012). They can be useful for scoping different types of cumulative effects assessments (activity, threat, species, and area) and they help identify the threats and clarify links between human activities and potential impacts on aquatic ecosystem components. The Government of Canada has developed national guidelines for the format of these models (Government of Canada 2012). PoEs can range from small scale, simple impact links, suitable for a species-specific habitat, to more complex, large scale networks, suitable for a bioregion (Government of Canada 2012). PoEs have typically been used to describe activities, such as aquaculture, but can also be used to illustrate the linkage pathways between anthropogenic activities, threats and population parameters, such as changes in mortality and fecundity.

This assessment uses a species-based PoE model to elucidate the linkage pathways between threats and their impacts on a particular species, which then informs a cumulative effects assessment.

PoE models provide useful outputs for the scoping phases of cumulative effects assessments as they can identify all linkage pathways (including interactions between threats), and the literature available to guide further investigations into quantifying the linkage pathways. The outputs of a PoE conceptual model consist of a visual representation of the threat linkage pathways, with supporting justification text. This can be in the form of a table or linked text presenting the evidence that is available to describe and quantify each linkage pathway shown on the diagram.

2.2 RESIDENT KILLER WHALE SPECIFIC POE MODEL

The first step in developing a PoE model is to scope the threats and endpoints (in this case fecundity and mortality) that the model will examine. In the present case this is not required, as the identification of the primary threats to the RKW populations has already been completed by DFO Species at Risk (DFO 2017b, 2018a). This PoE model does not explicitly include the source activities of threats as is traditionally found in such models because the focus is on specific threat impacts well defined by SARA, and we do not have sufficient knowledge or a time series of the activities to be able to include these.

The proposed PoE conceptual model (Figure 5) outlines how the potential impacts to RKW from these primary threats might manifest. In addition to describing the direct linkage pathways from threats to vital rates, as in a standard PoE model, this relatively data rich model allowed a novel approach whereby known and potential threat interactions linkage pathways to vital rates were also included. The overall PoE conceptual model (Figure 5a) identifies the important conceptual connections between threats and RKW populations, based on literature review and expert opinion. The diagrams consists of two to three rows of boxes; grey boxes represent threats of interest, dashed line boxes represent any interactions between threats, and black boxes indicate the population parameters (vital rates) affected by the impacts. Each linkage pathway is tagged with a numerical value that links it to the subsequent text to justify and provide supporting evidence for that specific linkage pathway. For clarity, direct and interaction effects are presented separately below the main diagram to simplify their interpretation and to aid the identification of the correct numbered linkage pathways, Figure 5b illustrates the direct linkage pathways between threats and vital rates, and Figure 5c the (indirect) interaction linkage pathways between threats and vital rates.

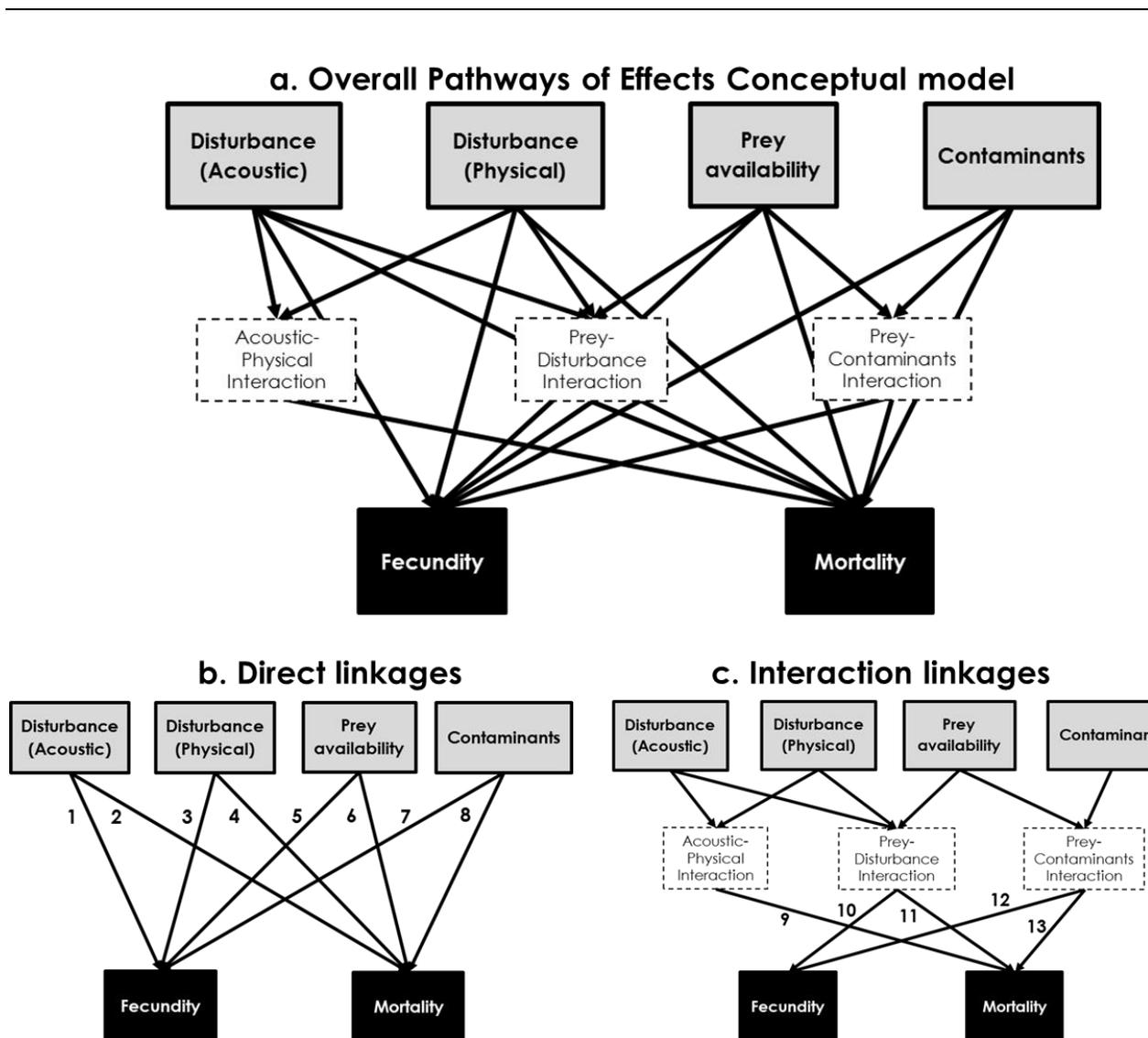


Figure 5 - a. Overall Resident Killer Whale Pathways of Effects (PoE) conceptual model, including priority threats, interactions, and impacts on Resident Killer Whale fecundity and mortality. The main diagram (a) is broken up to clearly illustrate linkage pathways and numbering for direct linkage pathways (b) and interaction linkage pathways (c). Numbers next to each linkage pathway refer to descriptions in the text.

This single PoE model (Figure 5a) represents both populations of Resident Killer Whale (NRKW and SRKW) as the mechanisms by which threats affect individuals in the two populations are assumed to be the same in this assessment; it is in the details and quantification of the linkage pathways where differences between populations may occur. Population differences are explored in the PVA section (Sections 3 and 4 - Population Viability Analysis Model and Results) where the same linkage pathways will be assessed for each population to identify where differences occur and will be captured in accompanying tables of evidence. The components making up the PoE model outlined above are explained and justified in detail in the following sections (2.4 - 2.7), the numbered links on the PoE diagram connect to sections of text through the numbers assigned to each in this format [1]; [2]; etc.

The subsequent PVA section (Sections 3 and 4) will contain a more specific model and will be presented based on the actual analysis that was done with linkage pathways containing the values and data used to represent each component.

2.3 AQUARIUM REMOVALS (A HISTORIC THREAT)

2.3.1 Background

The removal of killer whales from the wild for display in aquaria around the world (a 'live capture' fishery) was a significant historical threat to Resident Killer Whale populations and could still have residual effects on current populations. This historic threat is not included in the PoE conceptual model but will be investigated in the PVA section.

In Canada, the absence of laws to guide interactions with killer whales or regulate their capture prior to 1970 meant that killer whales were classed as 'wildlife', and permits were issued that allowed them to be removed and held in captivity. These permits had no catch quotas for Canadian netters until 1970 when initial protective legislation was introduced. In the USA, there were no permits or limits introduced until 1971 (Bigg and Wolman 1975; Baird 2001). In 1982 the capture of killer whales was no longer permitted in BC, as the provincial Wildlife Act was amended to exclude killer whales as 'wildlife', and the Canadian federal government included the protection of cetaceans under the Fisheries Act. Specific marine mammal regulations protecting killer whales in Canada were fully in place by 1994 (Baird 2001). In the USA, the Marine Mammal Protection Act prohibits the capture of killer whales without a permit, and no permits have been issued since 1989 (Tierney 2010).

In addition to live captures, there is evidence that in the years before protective regulations, killer whales regularly suffered gunshot injuries and likely deaths due to fisheries conflicts. Approximately 25% of SRKW whales from the live capture fishery in Puget Sound had evidence of bullet wounds. The shooting of killer whales in the NE Pacific is estimated to have begun around 1929, and became illegal in Canada in 1970 under the Fisheries Act and in the US in 1972 under the Marine Mammal Protection Act (Hoyt 1990; Olesiuk et al 1990; Krahn et al. 2002). However, in the US, fishermen were still allowed to shoot marine mammals to 'protect their catch and gear' until the Act was amended in 1988, shootings are still thought to occur in Alaska (Fraker 2013). The current status of gunshot wounds in the NRKW and SRKW populations is uncertain, though data from the NRKW survey indicate that observations of injuries suspected to be a result of gunshots mostly occurred prior to the year 2000, and in the last 10 years there has been only one observation of an injury potentially consistent with a gunshot (DFO Cetacean Research Program, unpublished data). Gun shot wound mortality at this time is assumed to be zero for both populations.

2.3.2 The Live Capture Fishery

The live capture fishery removed 68 killer whales from BC and Washington State waters from 1962-1977. Of these, an estimated 48 were from the SRKW population (removed between 1962-1977), 15 from the NRKW population (removed between 1965-1969), and five were Bigg's killer whales (removed between 1970-1975) (Asper and Cornell 1977; Olesiuk et al. 1990, Table 2). These numbers represent all whales removed from the Canadian Pacific killer whale populations, including those that died during a capture, or since. The majority of individuals removed from the SRKW population were physically immature (Table 2: 30/48, 63%). More males were removed than females; of those identified to sex, there were 26 males and 18 females removed from the SRKW population (Table 2). Fewer individuals were removed from the NRKW population: nine juveniles and six adults, of which eight were males and seven were females (Table 3).

Table 2 - Southern Resident Killer Whales (SRKW) captured or killed by pod, year, length (m) and sex during the live capture fishery in BC and WA (after Olesiuk et al. 1990). Animals that were presumed to belong to the SRKW population based on their location of capture are indicated by the superscript¹.

Pod/Area	Year caught	N	Physically immature						Mature			
			≤3.5m			3.5-4.5m			4.5-6m	≥ 4.5m	≥6m	
			M	F	U	M	F	U	M	F	M	U
South Vancouver ¹	1962	2	-	-	-	-	-	-	-	-	2	-
J01, K01 or L01	1964	1	-	-	-	-	-	-	1	-	-	-
J01, K01 or L01	1965	2	-	-	-	-	1	-	-	1	-	-
South Vancouver ¹	1966	1	1	-	-	-	-	-	-	-	-	-
K01	1967	8	1	2	-	2	1	-	-	1	1	-
J01, L01	1968	5	-	-	-	3	-	-	2	-	-	-
South Vancouver ¹	1968	2	-	-	-	1	-	-	-	1	-	-
South Vancouver ¹	1969	3	-	-	-	-	-	-	-	1	1	1
South Vancouver ¹	1970	2	-	1	-	-	-	1	-	-	-	-
J01, K01, or L01	1970	11	2	-	2	3	2	-	1	1	-	-
Washington ¹	1971	2	-	-	-	2	-	-	-	-	-	-
L01	1971	3	-	1	-	1	1	-	-	-	-	-
J01	1972	1	1	-	-	-	-	-	-	-	-	-
Washington ¹	1973	1	-	-	-	-	-	-	-	1	-	-
K01	1973	1	-	-	-	-	-	-	-	1	-	-
L01	1973	2	-	-	-	-	-	-	1	1	-	-
South Vancouver ¹	1977	1	-	1	-	-	-	-	-	-	-	-
Total	1962-77	48	5	5	2	12	5	1	5	8	4	1

¹. Presumed SRKW based on area captured

Table 3 - Northern Resident Killer Whales (NRKW) captured or killed by pod, year, length (m) and sex during the live capture fishery in BC and WA (after Olesiuk et al. 1990). Animals presumed to belong to the NRKW population based on their location of capture are indicated by the superscript².

Pod/Area	Year caught	N	Physically immature						Mature			
			≤3.5m			3.5-4.5m			4.5-6m	≥ 4.5m	≥6m	
			M	F	U	M	F	U	M	F	M	U
C01	1965	1	-	-	-	-	-	-	-	-	1	-
I11	1967	1	-	-	-	1	-	-	-	-	-	-
NE Vancouver ²	1968	1	-	-	-	1	-	-	-	-	-	-
A (A05)	1968	6	1	-	-	-	1	-	1	3	-	-
A05	1969	6	-	2	-	2	1	-	1	-	-	-
Total	1965-69	15	1	2	0	4	2	0	2	3	1	0

²Presumed NRKW based on area captured

2.3.3 Effects of Removals

Because of the proximity of SRKW habitat to population centres, the majority of whales (48, 71%) removed for aquaria display were from this population. This removal had a disproportionate impact on the smaller SRKW population, as small populations are more vulnerable to extinction because of stochastic events (DFO 2018a). Not only did the removals significantly reduce SRKW population size, they also skewed SRKW population structure, as removals were predominantly comprised of juveniles and young males (Vélez-Espino et al. 2014a). After most live captures ended, there was a period of growth in the SRKW population of 19% until 1980, followed by a decline of 11% that was attributed to the skewing of the population structure from preferential captures (Olesiuk et al. 1990; Giles 2014). Removals are expected to have impacted the NRKW population less significantly than the SRKW population, as fewer individuals (15) were removed from a larger population. There are no mitigation actions for this historic threat but the long-term effects should be acknowledged in any assessment of cumulative effects.

2.4 REDUCED PREY AVAILABILITY [5,6,10,11,12,13]

Prey availability is made up of two components, prey abundance and prey access. A full characterisation of prey availability would involve considering the components that influence both whether sufficient prey is present and whether it can be accessed for consumption, which relates to factors such as timing and ability to forage. The current assessment captures this threat by examining changes in prey abundance, using a time series of Chinook salmon data. The prey access component is partially captured in interactions between prey abundance and vessel acoustic and physical disturbance.

2.4.1 Background [5,6]

Resident Killer Whales are fish-eating marine mammals with a distinct preference for Chinook salmon (*O. tshawytscha*), due to their large size, high lipid content and year round availability

(Ford et al. 2005; Ford and Ellis 2006). Analysis of prey remains indicates that Chinook salmon can comprise up to 90% of the summer diet of SRKW (Ford and Ellis 2006; Ford et al. 1998; Hanson et al. 2010). Changes in RKW population parameters have been directly linked to fluctuations in Chinook salmon stocks [5,6]. Inter-annual variability in Chinook salmon is related to inter-annual variability in RKW mortality (Ford et al. 2010; Vélez-Espino et al. 2014b) [6] and fecundity (Ward et al. 2009; Vélez-Espino et al. 2014b) [6]. There is a positive correlation between RKW calving probability and Chinook salmon abundance (Ward et al. 2009) [5] and a negative relationship between RKW mortality and Chinook salmon abundance (Ford et al. 2009; 2010; Vélez-Espino et al. 2014b) [6]. These findings strongly indicate that Chinook salmon abundance plays an important role in RKW population dynamics.

Aerial photogrammetry has provided information on the link between mortality rates and body condition, or the fat stores, in individual whales (Durban et al. 2015). Declines in the Eye Patch Ratio (EPR, measured as the proportional head width) have been linked to short-term mortality. In 2008 and 2013, 43 individuals from the SRKW population were measured and eleven had significant reduction in EPR, indicating depletion of fat stores (Durban et al. 2015). Animals that were not pregnant or nursing (life stages where body metric changes are expected Kriete 1995; Kastelein et al. 2003), with reduced EPR died shortly after being photographed in this condition.

Though the majority of the summer diet of RKW consists of Chinook salmon, they also consume other species of salmonids and non-salmonids. It is estimated that overall, 96% of the RKW diet comprises salmonids, and within this, 71.5% is Chinook, 24% chum, and 0.5% other salmonids such as Coho salmon (*O. kisutch*). The non-salmonid fish in the diet are Pacific herring (*Clupea pallasii*), sablefish (*Anoplopoma fimbria*), yelloweye rockfish (*Sebastes ruberrimus*), quillback rockfish (*Sebastes maliger*), and Pacific halibut (*Hippoglossus stenolepis*) (Alava et al. 2012; Ford et al. 2006; 2009). As chum salmon can comprise 24% of the salmonids in RKW diet, its availability and abundance may also be a contributor to RKW population growth. Chum salmon become more important in the RKW diet in autumn, surpassing the contribution of Chinook salmon at that time (Ford and Ellis 2006; Ford et al. 2010). However, the two studies that have examined the role of chum and other salmon species in killer whale demography did not find any statistical evidence for a relationship between these fish stocks and RKW with mortality or fecundity (Vélez-Espino et al. 2014b; Ward et al. 2009).

2.4.2 Important Salmonid Stocks

There is evidence from genetic analysis of prey samples that the two Resident Killer Whale populations may exploit different combinations of Chinook salmon stocks. The dominant Chinook salmon stocks found in SRKW diet from late spring to early autumn are the Fraser River and Puget Sound Chinook salmon (Vélez-Espino et al. 2014b; Hanson et al., 2010). Over the season (late spring to early autumn), the proportion of Fraser Chinook salmon in Juan de Fuca Strait increases in relation to Puget Sound stocks as populations travel through the area on their return migrations to the Fraser river (DFO 2018b¹). Chum salmon stocks consumed by SRKW are assumed to be from Puget Sound stocks (Vélez-Espino et al., 2014b).

For NRKW, the dominant Chinook salmon prey stocks are mainly Fraser River, but also Northern and Central BC, west coast of Vancouver Island, Georgia Strait, Puget Sound and the upper Columbia stocks, in the same season (late spring to early autumn) (Ford and Ellis 2006;

¹ DFO. 2018b. Discussion Paper: February 15, 2018. Proposed 2018 Salmon Fishery Management Measures to Support Chinook Salmon Prey Availability for Southern Resident Killer Whales (internal).

Ford et al. 2009; Vélez-Espino 2014b). Chum salmon stocks consumed by NRKW are Fraser, East coast Vancouver Island (ECVI), and South BC Coast (Vélez-Espino et al. 2014b).

Some salmon stocks may be consumed by more than one population of RKW, such as those salmon stocks with a more northerly distribution that may encounter killer whale populations throughout their migrations. For example, Fraser Summer (age-4 ocean type) Chinook salmon migrate to the Fraser River in August through Johnstone Strait and Strait of Juan de Fuca (DFO 2018b).

2.4.3 Trends in Chinook Abundance

Chinook salmon production mainly happens in major river systems such as the Fraser and Yukon rivers, with some in smaller streams (Healey, 2003). Chinook salmon stocks have experienced widespread population declines, especially in the 1990s (Figure 6; Riddell et al. 2013). Since the 1980's, Chinook salmon productivity is estimated to have declined by 25-40% for many BC stocks (DFO 2018b¹). Chinook salmon populations have also shown a trend towards smaller body sizes (Wiles 2016; Ohlberger et al. 2016). The productivity of wild chum salmon stocks has also undergone widespread declines in Washington (WA) and British Columbia (BC), with 81% of stocks having recent declines in productivity (Malick and Cox 2016).

2.4.4 Threshold Effects/Nonlinearities [10,11,12,13]

The ability of killer whales to successfully catch and consume prey (access to prey) may be affected by vessel disturbance, as will be discussed in Section 2.5.6. The prey-disturbance interaction [10,11] effectively reduces killer whales access to prey, which can have effects at the individual and population levels. The interaction between prey abundance and physical and acoustic vessel disturbance may also potentially include reduced access to foraging habitat, in addition to reduced foraging. It is not well understood whether prey distribution becomes increasingly patchy with reduced abundance, and whether patchy prey distribution might impact prey access / foraging for Resident Killer Whales.

The consumption of prey items contaminated with persistent, bioaccumulative and toxic chemicals (PBTs) can also affect killer whale vital rates. Salmonids have been found to contain a range of contaminants that are also found in killer whale tissues, which is further discussed in section 2.7.1. The prey-contaminants interaction [12,13] potentially affects the mortality and fecundity of these populations.

In addition, there may be other factors and non-linearities that are masking or confounding the detection of stronger interactions between RKW vital rates and prey abundance (Vélez Espino et al. 2014b). For example, large-scale climate changes, genetic factors, and other environmental and anthropogenic variables are also changing in this system and make the isolation of single threat impacts more difficult.

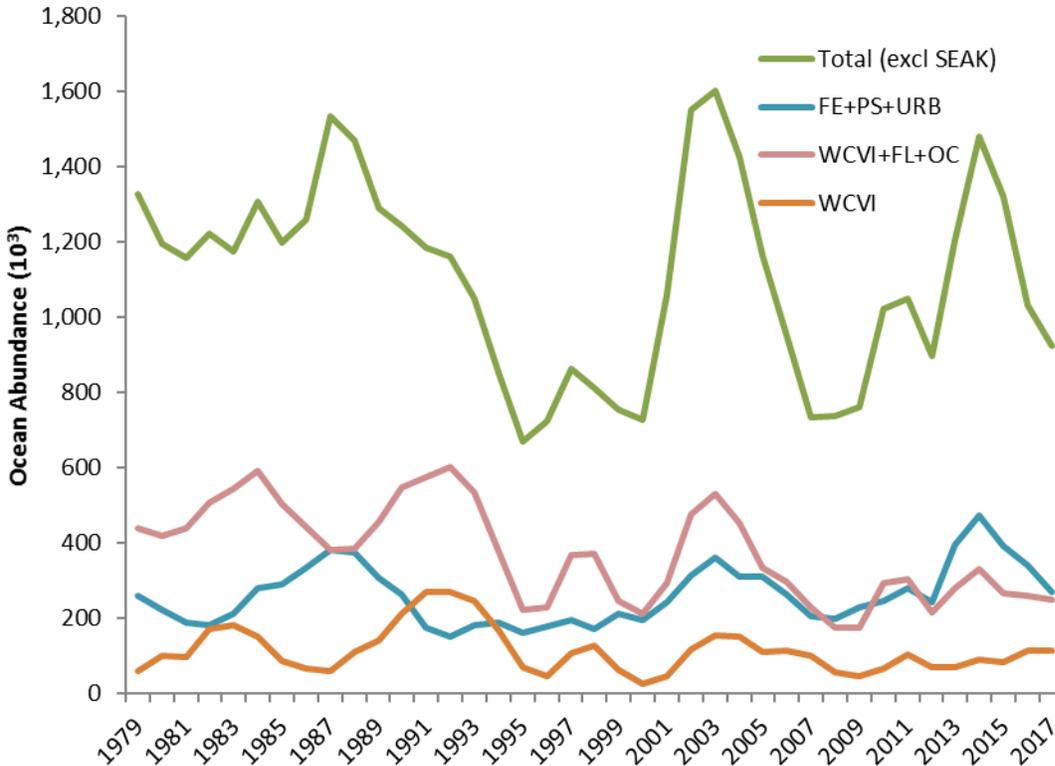


Figure 6 – Time series of modelled Chinook salmon ocean abundance (thousands) for stocks of interest to RKW. Total is the coastwide index (excluding southeast Alaska stocks), FE+PS+URB is the Fraser Early, Puget Sound and Upper River Basin, WCVI+FL+OC is the West Coast Vancouver Island, Fraser Late, and Oregon Coast stocks; WCVI is the West Coast Vancouver Island stock.

2.5 DISTURBANCE (ACOUSTIC) [1,2,9,10,11]

Increasing amounts of vessel traffic, industrial activities and other anthropogenic activities are affecting the physical and acoustic habitat of killer whales, as well as their behaviour. The impact of disturbance on cetaceans at both the individual and population level is not well understood (Nowacek et al. 2007; DFO 2011). Vessel disturbance has been identified as a principal threat to the two Resident Killer Whale populations in the DFO SARA Recovery Strategy (DFO 2011, 2018a). As it is as yet not possible to separate the impacts of vessel noise from the impacts from vessel physical presence, this threat captures both of these aspects.

2.5.1 Background

Vessels are one source of underwater noise; commercial ships, recreational vessels, whale watching vessels, and military vessels are active in the range of the two Resident Killer Whale populations. Noise can also come from military and research activities (e.g., sonar, explosions), aircraft overflights, construction (e.g., pile driving, blasting), and from dredging. Different types of noise can potentially result in various levels and types of disturbance, a level of complexity out of scope for this work. The impact of noise on killer whales can be acute or chronic, with the effects dependent on frequency range, source level and signal structure of the sound (Richardson et al. 1995; National Research Council 2003; Nowacek et al. 2007). Acute impacts are intermittent and less predictable in the environment. The impacts of acute sound exposure (e.g., blasting) can include direct mortality if whales are in the affected area at the time of noise emittance, and are difficult to predict in space and time [2].

Though there are many potential sources of noise disturbance that could impact Resident Killer Whales, this assessment is limited in scope to considering the impacts of acoustic disturbance from vessels, the main source of chronic underwater noise in the assessment area. Vessel-related disturbance has been specifically identified as a risk to recovery by DFO (DFO 2017b). How to discern or measure the impacts of such acoustic disturbance can be challenging and the acoustic component of vessel disturbance on RKW can cause impacts via a number of mechanisms: behavioural changes, auditory masking, fitness reduction, and resultant population consequences. Another potential impact caused by vessel disturbance is due to the physical presence of a vessel, and it is difficult to separate the impact of the physical presence of a vessel from the impact of the noise it produces. For dolphins, boat presence alone can disturb behaviour (Pirrotta et al. 2015), and impact their ability to rest and reach deep sleep (Tyne 2015).

2.5.2 Behavioural Change [10,11]

There can be energetic costs to any disturbance that causes an animal to switch behavioural states (e.g. from resting to travelling) or results in more time spent performing energetically costly activities such as evasive or surface active behaviours (Williams et al. 2009). These costs increase with the severity and duration of the response (Erbe 2002; Williams et al. 2006; Lusseau et al. 2009; Ayres 2012; Williams et al. 2014). Coping mechanisms can range from short-term avoidance to long-term habitat abandonment (Kruse 1991; Williams et al. 2002a,b; Lusseau and Bejder 2007). In nearshore core areas in the summer and fall, Resident Killer Whales spend 40-67% of their time engaged in foraging behaviours (Ford 2006; Noren et al. 2009). The onset of behavioural changes (affecting 50% of observed killer whales) has been observed to begin at received levels of ~ 130 dB re: $1 \mu\text{Pa}$ (broadband, root-mean-square) (Williams et al. 2014). Noise or vessel disturbance that causes an animal to reduce foraging time can have impacts on mortality and fecundity similar to that of lowered prey abundance [10,11]. Increased swimming velocity (associated with vessel avoidance behaviours) can result in an estimated 20% increase in energetic expenditures (Kriete 1995, 2002), meaning less energy is available for other vital functions and increasing caloric requirements.

2.5.3 Auditory Masking [10,11]

Vessel noise overlaps with the sound frequency range used by killer whales (Watkins et al. 1987; Berchok et al. 2006; Mouy et al. 2009; Tervo et al. 2011; Hatch et al. 2012), and can mask the receiving of acoustic signals used for foraging, navigation, communication and social interaction (Erbe 1997, 2016; Weilgart 2007; Clark et al. 2009; Castellote et al. 2012). Masking can interfere with echolocation of prey and the effectiveness of foraging activities [10,11]. SRKW have been shown to increase call duration and amplitude in the presence of boats, which has been suggested to be an adaptation to masking effects but one that may come with energetic costs (Foote et al. 2004; Holt et al. 2008). There is an additional energetic cost if masking prevents or inhibits successful foraging events, causing increased energy to be spent on foraging activities and less available for other life history processes, such as mating and fecundity [10,11].

2.5.4 Fitness Reduction due to Stress [1,9]

When individuals cannot reduce or avoid proximity to disturbance, stress levels may increase causing a reduction in fitness that may be manifested in reduced reproductive success (Lusseau and Bejder 2007) [1]. Individuals with higher levels of stress may be susceptible to other causes of mortality, such as disease, parasites, and vessel strikes (Fair and Becker 2000) [9]. Humpback whales have been shown to be more susceptible to entanglement after acoustic

trauma (Todd et al 1996; Ketten et al 1993). However, Ayres et al (2012) concluded that elevated stress in SRKW, measured by a combination of hormone levels, is linked to prey abundance, rather than vessel disturbance.

2.5.5 Population Consequences of Disturbance [1,2,9,10,11]

Population effects of noise can manifest through behavioural or physiological changes, which can have impacts on health and vital rates [1,2]. A Population Consequences of Disturbance (PCoD) model (NRC 2005; Tollitt et al. 2017) has been used to quantify the chronic and acute impacts of noise disturbance on killer whales (Figure 7). A noise exposure model combined with the PCoD model has been used to estimate lost foraging time for SRKW as a combination of behavioural response and masking (Tollitt et al. 2017). In the Salish Sea’s busy traffic areas, individual noise disturbance events can combine to have potentially substantial impacts (Tollitt et al. 2017). The SMRU (2017) model predicted that in the Salish Sea, SRKW foraging time was decreased by 20-23% of each whale day (i.e. days when SRKW pods were predicted to be in the study area). Two-thirds of those effects were estimated to be from commercial vessels and one-third from whale watching vessels. Prey detection range was decreased as a result of masking of echolocation clicks by vessel noise. The combined effect of both vessel types was estimated to reduce the range of prey detection by 12-37%. Reductions in foraging time and efficiency can result in the same impacts to vital rates as that of reduced prey abundance (increased mortality and reduced fecundity) [10,11].

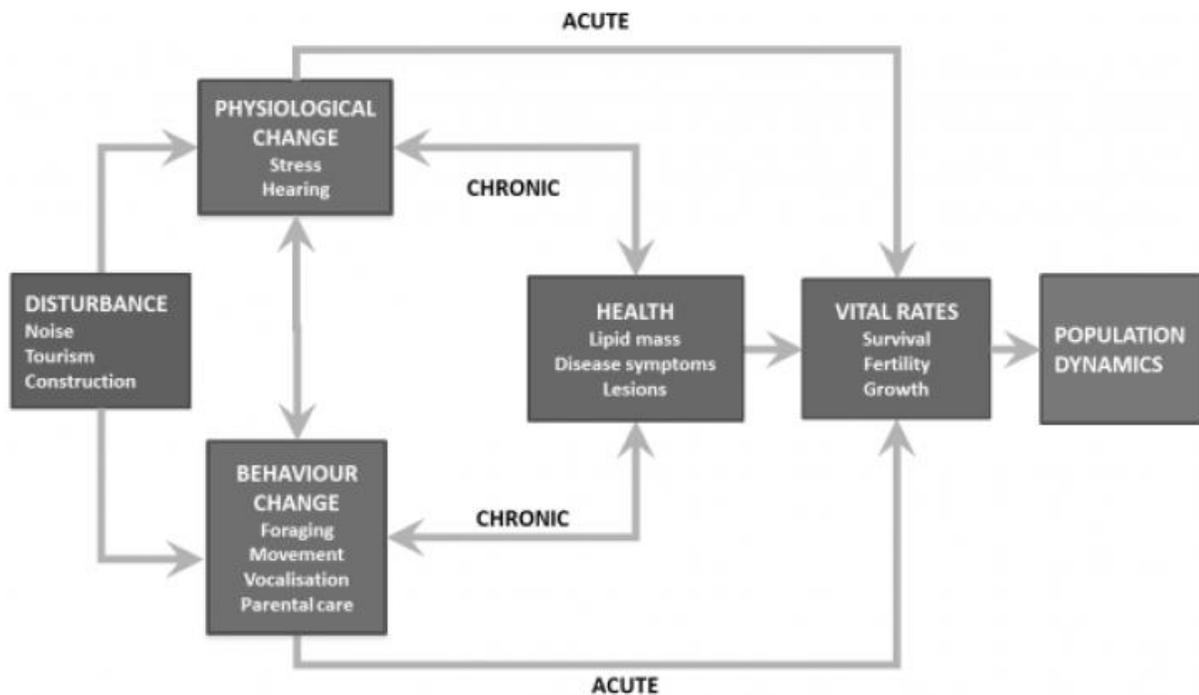


Figure 7 - Population Consequence of Disturbance (PCoD) model and the pathways of impact that lead to changes in population dynamics (adapted from Tollitt et al. 2017)

2.5.6 Interactions/Thresholds/Non-linearities [9,10,11]

The impacts of vessel noise disturbance could be exacerbated when prey abundance is low, and conversely, vessel noise disturbance may have little or no effect on overall feeding efficiency when prey abundance is high [10,11]. In killer whales, glucocorticoid hormones (which increase in response to nutritional or psychological stress) in scat samples have been found to be low in summer, when vessel and salmon (Fraser River) abundance is high, and high when

vessel and salmon abundance are low, indicating that short-term physiological effects of prey abundance appear to overshadow impacts from vessels (Ayres et al. 2012) [10,11]. However, the study did not collect samples from periods of time when vessels were high but salmon levels were low. The temporal relationship between vessel disturbance and stress levels has not been quantified and there may be an unobserved delay between exposure and stress hormone production that makes these results difficult to interpret.

Alternately, when prey abundance is extremely low, predators may ignore disturbance because the fitness consequences of abandoning a predation event would be higher and this may increase other mortality risks (e.g., vessel strike) if adaptive disturbance responses (e.g. startle) are overridden or ignored during pursuit of limited prey [10,11]. This is supported by basic ecological principles relating to a predator-prey relationship in the context of resource availability: prey species are only forced to forage in areas of higher risk of predation when food is scarce. The basic energy requirements of an individual mean that they may have to accept predation risk to meet their energy budget in instances where conditions make the cost of predator avoidance high (Sansom 2009; Stevens 2010). Human disturbance can be analogous to risk from predators, and similar fitness impacts can result from predation and non-lethal disturbances such as noise (Frid and Dill 2002). In the case of killer whales, impacts from vessel avoidance could be analogous to those from predator avoidance. When Southern Resident Killer Whales are food limited, mechanisms of energetic impacts such as this are of concern (Lusseau et al. 2009). The effects of masking are another component that killer whales would have to overcome in the presence of vessels if they chose to take the risk to forage. An additional possible interaction is that whales could be more likely to abandon foraging activities in a low abundance/poor quality prey patch if disturbed, because the energetic returns under those circumstances are so poor (Kuningas et al. 2013; Pirodda et al 2015).

Noise may have the potential to cause accidental beaching or entrapment, and loud noises have been demonstrated to cause strandings (DFO 2018a) [9]. It has been suggested that the reason a group of SRKWs spent 30 days in an inlet in Puget Sound in 1997 was due to an aversion to passing under a noisy bridge (Shore 1998). The evidence for interaction between acoustic disturbance and other threats is sparse and not thoroughly understood for killer whales and other cetacean species.

2.6 DISTURBANCE (PHYSICAL) [3,4]

The second component of vessel disturbance is physical disturbance, identified as a principal threat to the Resident Killer Whale populations in the DFO SARA Recovery Strategy (DFO 2011, 2018a) [4]. For this study, this threat is captured by vessel strikes only.

2.6.1 Background

Killer whales can be injured or killed as a result of a vessel collision, the outcomes of the collision can be particularly damaging if moving propeller blades are encountered (Ford et al. 2000; Baird 2001) [3,4]. The spatial overlap between Resident Killer Whale populations and maritime traffic suggests that there is a risk of injury and mortality from ship strikes (Williams and O'Hara 2010). Observations have found that recreational vessels in proximity to or engaging in SRKW-watching are increasingly being driven in a manner that heightens the risk of a collision, such as stopping in the path of the whales, chasing whales, approaching whales head on or crossing their paths (Ferrera et al. 2017). The number of vessels in proximity to the whales and vessel speed are likely to have a strong influence on the rate of collisions and the severity of injury (Conn and Silber 2013; Ferrara et al. 2017).

2.6.2 Reports of Injuries and Mortalities

Injuries [3]

The frequency and severity of injuries from ship strikes on Resident Killer Whales is uncertain, but there are some anecdotal accounts available as well as observed physical evidence of injuries from regular population surveys. These sources provide some insight into the proportion of the population that may have experienced injuries from vessel strikes. Table 4 summarises information on known vessel strike incidents that resulted in injuries (that the authors were able to locate from the literature and from experts) for NRKW and SRKW. In many cases the injured killer whales recovered, even in severe cases (Ford et al. 2000; Baird 2002).

Table 4 – Timeline of known incidents of vessel strikes causing injury in NRKW and SRKW

Year	Population	Individuals (#)	Description
1995	NRKW	1	Struck by a speed boat, dorsal fin was wounded, recovered (Baird 2002; Williams and O'Hara 2010)
1998	SRKW	1	Non-fatal strike in Haro Strait (Baird 2002; Williams and O'Hara 2010)
2003	NRKW	1	Injured by a high-speed boat but recovered (Federal Register 2007; Baird 2002)
2005	SRKW	1	K25 injured by collision with the skag of a whale watch vessel (drifting power off) (K. Balcomb, Center for Whale Research, WA, pers. comm.) and resulted in a minor injury to the whale, which recovered (Williams and O'Hara 2010)
2006	NRKW	2	One calf A59 injured near Campbell River; One serious injury to G39 (Williams and O'Hara 2010)
2014/5	NRKW	1	Superficial wounds to NRKW A61 from a propeller strike, since recovered (B. Wright, DFO, pers. comm.)
2015	NRKW	1	Photo evidence of relatively severe prop strike wounds across the back of NRKW A60, since recovered (B. Wright, DFO, pers. comm.)
2016	NRKW	1	Observed by aerial photogrammetry to have a superficial propeller wound (Ferrara et al 2017)
2018	NRKW	1	A109, a juvenile (born 2014) NRKW observed with a severe propeller strike injury across its back behind the dorsal fin, appeared vigorous despite injury and was re-sighted by others some days later and again in late August. Injuries appeared to be healing well (B. Wright, DFO, pers. comm.)

If the frequency of injuries is estimated based on the time period 1995-2016, NRKW had eight reported injuries in 21 years, and SRKW experienced two in 21 years. It is likely that these reports underestimate the true frequency of propeller wounds (Williams and O'Hara 2010). If not killed by a strike, injured killer whales may suffer fitness consequences as a result of the injury (e.g. reduced ability to hunt), potentially impacting fecundity [3].

Mortalities [4]

Attributing cause of death in Resident Killer Whales is difficult in many cases as carcasses often sink and are lost, meaning only a small proportion are recovered for necropsy examination (Ford et al. 1998; National Marine Fisheries Service 2008; DFO 2018a). This is a reason why the cause of many calf deaths in particular is unknown (Baird 2000). Most mortality events are not directly observed and are only recorded when the animal is not seen with its matriline in

subsequent encounters. Based on annual census data, it is estimated that between 1974 and 2008, 96 SRKW and 176 NRKW died, but of these only 19 SRKW and 5 NRKW carcasses were recovered, a recovery rate of 20% for SRKW and 3% for NRKW (Barbieri et al. 2013).

Data that are collected on cetaceans that are found dead, or alive on the beach and unable to return to sea are included in stranding databases. In Canada, DFO maintains a marine mammal incident database that captures stranding data (CRP unpublished data) and in the USA, NOAA maintains a database for both a national marine mammal stranding network and a west coast marine mammal stranding network which began in the 1980's. We have only presented information from stranding data that is linked to a specific killer whale population (i.e., after the population censuses began in the 1970's). Table 5 summarises the incidences of mortality events presumed to be due to vessel strike (vessels of any type were included). Using the time range in Table 5 (1974-2016), both the NRKW and SRKW populations had three mortalities [4]. Again, these are likely to be underestimated values due to the unknown causes of death in missing and presumed dead animals.

Table 5 – Timeline of reported mortalities resulting from ship strikes in NRKW and SRKW. Data used is limited to post-1970s after population censuses had begun so that mortalities of individuals can be linked to a specific population.

Year	Population	Individuals (#)	Description
1974	NRKW	1	Fatal ferry strike, possibly NRKW (Baird 2002; Ford et al. 1994)
2006	NRKW	2	One injured near Campbell River and died following year (A82) One fatal strike near Prince Rupert (C21) (Gaydos and Raverty 2010; Williams and O'Hara 2010)
2006	SRKW	2	One male (L98) killed by a tugboat propeller in Nootka Sound; One stranded female (L112) was determined to have died from blunt trauma presumed from a ship strike (Gaydos and Raverty 2010; Williams and O'Hara 2010)
2016	SRKW	1	J34, an 18 year-old male found dead in Sechart died from blunt force trauma presumed to be from a vessel strike (Ferrara et al. 2017; DFO 2017a)

2.6.3 Interactions/Thresholds/Non-linearities [9,10]

Interactions of vessel strike with other threats have not been clearly demonstrated, though it has been hypothesised that killer whales might have a higher risk of vessel strike when exposed to loud sounds, which could impair the whales' ability to detect vessels (Erbe et al. 2018) [9].

In addition, as outlined in the previous acoustic disturbance section (2.5.6), when prey abundance is low, killer whales may ignore disturbances and put themselves at greater risk of ship strikes during the pursuit of prey, as has been documented in other mammal species (Frid and Dill 2002; Sansom 2009; Stevens 2010) [11].

2.7 CONTAMINANTS [7,8,12,13]

2.7.1 Background

Resident Killer Whale populations are exposed to a variety of contaminants released into the marine environment historically and currently, via sources such as rivers, wastewater, storm

water and atmospheric deposition (Cullon et al. 2009). Exposure to contaminants is of particular concern for SRKW because they spend much of the spring, summer and autumn in the waters of Puget Sound and Georgia Basin (Krahn et al. 2007), areas which are influenced by the major urban centres of Vancouver and Seattle. There are a range of contaminants with the potential to be of concern to Resident Killer Whale populations, as outlined in a recent review by Environment Canada (Van Zandvoort, 2019 unpubl.²) and in a prioritised list of contaminants for BC killer whales (Morra and Gobas, 2017 unpubl.³). While any of the contaminants to which Resident Killer Whales are exposed have the potential to cause negative impacts, for most, the nature of their impacts on population parameters is uncertain and so out of scope for this assessment.

Currently, contaminants of particular concern to killer whales are those that biomagnify, reaching highest concentrations in animals at the top of the food chain, and are generally ones classified as being Persistent, Bioaccumulative and Toxic chemicals (PBTs) (Ross et al. 2000; Ross 2006). Although the manufacturing of many of these contaminants has been banned since the 1970's, they still persist in the environment. When these chemicals are consumed, they bioaccumulate in the tissues of organisms and can elicit chronic forms of immunotoxicity and neurotoxicity and can result in reproductive impairment (Ross et al. 2000; Ross and Desforges 2014, unpubl.⁴; Morra and Gobas 2017, unpubl.³) [7,8]. Within the PBT group, two major contaminant classes, polybrominated diphenyl ethers (PBDEs) and polychlorinated biphenyls (PCBs) have been identified as being of ecotoxicological concern for killer whales.

2.7.2 Polybrominated Diphenyl Ethers (PBDEs)

PBDEs are a group of synthetic contaminants that are used as flame retardants in a range of products including many electronic and household items. PBDEs are contaminants of concern due to their effects on the immune system, reproduction and development in mammals (Ross et al. 2006). PBDEs are fat-soluble and so can be measured by sampling killer whale blubber. The levels of PBDEs in killer whale blubber is of growing concern, as concentrations have been noted to be increasing in killer whales over time (Ross 2006; Guy 2018 unpubl.⁵).

PBDEs consist of a basic diphenyl ether structure with one to 10 bromine atoms, leading to 209 different combinations (congeners). The number of bromine atoms attached to the molecule and the degree of bromination are linked to different properties and toxicity. For example, congeners with 4-8 bromine atoms typically bioaccumulate more as they bind to sediment better than those with 9-10 bromine atoms. PBDEs have been noted to have impacts on thyroid hormones and neuro-development and may be carcinogenic (US EPA 2017).

² Van Zandvoort, A. 2019. Review of contaminants of concern to Southern Resident Killer Whales and/or Chinook salmon. An evergreen report prepared for Environment and Climate Change Canada. Last modified April 4, 2019.

³ Morra, J., and Gobas, F. 2017. Recovering British Columbia's resident killer whales by tackling pollution. Report prepared for the Department of Fisheries and Oceans Canada. Unpublished

⁴ Ross, P.S. and Desforges, J.P. 2014. Towards a framework for organizing a forward-looking pollution workshop for SARA-listed marine mammals in February 2015. Report prepared for Fisheries and Ocean Canada, Fisheries Management–Species at Risk, March 18, 2014. Ocean Pollution Research Program, Vancouver Aquarium Marine Science Center. 31pp. Unpublished.

⁵ Guy, J. 2018. A risk analysis of legacy pollutants, PCBs, PBDEs and new emerging pollutants in the Salish Sea Killer Whales. Master's thesis, Simon Fraser University. Unpublished

2.7.3 Polychlorinated Biphenyls (PCBs)

Legacy PBTs, such as Polychlorinated biphenyls (PCBs), which are structurally similar to PBDEs, were assessed as being in the threat category of greatest concern to Resident Killer Whales in BC (Morra and Gobas 2017, unpubl.³). Further, a recent health risk-based evaluation of 25 different contaminants in RKW indicated that in terms of overall chemical exposure, PCBs were the pollutant of greatest concern to RKW (Gobas and Ross 2017 unpubl.⁶).

PCBs were historically used in products such as lubricants, paints, adhesives, flame-retardants, and particularly in heat resistant oils in electrical equipment (such as transformers and capacitors) (Clark 1999; Ross et al. 2006) and were released in significant amounts into the environment from industrial practices beginning in the 1920's. It is estimated that from 1930-1993, 1.3 million tonnes of PCBs were produced worldwide (Breivik et al. 2002a) and around 1.4% of this entered the environment (Breivik et al. 2002b).

Evidence of the environmental accumulation and persistence of PCBs and their impacts lead to PCB bans in the late 1970s in the USA and Canada, and by 2001 they were banned under the Stockholm Convention (Stockholm Convention 2001⁷; Jepson et al. 2016). Although Canada banned the production, import and sale of PCBs in 1977, a ban on environmental release came later (1985) and the use of PCB-containing equipment is still allowed until the end of service life. Despite these bans, PCBs persist in the environment due to a combination of the persistent nature of these contaminants, as well as continuing PCB releases as a result of accidental spills, fires (Environment Canada 2018), river run-off, and long range atmospheric transport and deposition (Desforges et al. 2018). PCBs are also still widely present as an 'inadvertent' contaminant in the pigments and dyes of many consumer products such as newspapers, cereal packaging, plastic bags and even sidewalk chalk (Stone 2016). PCBs from these sources can enter the ocean through improper waste disposal or management.

PCBs consist of one or more combinations of man-made organic chemicals (biphenyls) similar in structure. There are 209 PCB congeners which vary based on the chlorination (number of chlorine atoms) and the position of those atoms (Heindel and Zoeller, 2006; Environment Canada 2018). The specific combination of congeners present can be an important factor in their impacts/toxicity. 136 PCB congeners have been found in killer whales (Addison and Ross 2000). The composition of PCB congeners present in SKRW and NRKW populations are similar, with congeners 153, 138, 52, 101, 118, and 180 accounting for nearly 50 percent of the total PCB load (Ross et al. 2000). PCBs are linked to cancer and nervous system problems, infant death, birth defects, and brain damage (Sullivan et al. 2007). A description of the biological impacts of PCBs is provided in section 2.7.6.

2.7.4 PCB Bioaccumulation in Resident Killer Whales

PCBs released into the marine environment end up in the sediment and water column, where they are taken up by sediment dwellers and plankton. From there, PCBs biomagnify up the food web (Pearce and Gobas 2018 unpubl.⁸). Consequently, those most affected by the toxicity and persistence of PCBs in the environment are the long-lived animals at the top of the food chain

⁶ Gobas, F., and Ross, P.S. 2017. Health risk-based evaluation of emerging pollutants in Killer whales (*Orcinus orca*): priority setting in support of recovery. Unpublished research report.

⁷ Stockholm Convention. 2001. 2256 UNTS 119; 40 ILM 532 (2001).

⁸ Pearce, R., and Gobas, F. 2018. Evaluation of trends in PCB concentration and food- web transfer of PCBs to Resident Killer Whales. Report prepared for the Department of Fisheries and Oceans Canada. Unpublished.

(such as killer whales), as PCB levels can remain elevated due to bioaccumulation. This is exacerbated by the fact that cetaceans have a limited ability to metabolise higher chlorinated PCBs (Boon et al. 1997; Ross et al. 2000). PCB levels in killer whales are influenced by age and sex (Ross et al. 2000). For example, males become increasingly contaminated as they age, while levels decrease with age in reproductively active females, as they offload a percentage of their PCB burden to their young during gestation and lactation, potentially affecting the development of young calves (Ross et al. 2000).

The consumption of Chinook salmon is a significant source of PCBs for these populations of fish-eating killer whales (Ross et al. 2000). The PCB loads of Chinook salmon stocks vary, and the majority of PCBs present in returning adult Chinook salmon are obtained while out at sea (Cullon et al. 2009; O'Neill and West 2009). The PCB concentrations in Chinook salmon sampled in BC and Washington exceed a dietary threshold (8 µg/kg) estimated as protective for 95% of killer whales (Cullon et al. 2009). Southern Chinook salmon stocks consumed by SRKW have higher PCB contamination levels than the northern stocks consumed by NRKW, in particular the PCB load of Chinook salmon sampled in Puget Sound (in the Salish Sea) was up to five times higher than other populations (O'Neill and West 2009). The higher PCB loads in southern stocks is attributed to the fact that some Chinook salmon, termed 'residents', do not migrate, but rather remain in the Salish Sea area year round (O'Neill and West 2009), which includes areas highly contaminated with PCBs (Ross et al. 2004, 2006). Approximately 29% of hatchery subyearlings and 45% of yearlings that entered Puget Sound remained as residents (O'Neill and West 2009). Further increasing the exposure of SRKW to PCBs, the lipid content of more southerly Chinook salmon stocks is also lower, and SRKW may need to increase salmon consumption to compensate (Cullon et al. 2009). A small portion of RKW diet may include local non-salmonid fish, so consumption of fish residing in the more contaminated southern habitats may also contribute to the higher PCB loads observed in the SRKW population (Ross et al. 2000; 2006). Overall, SRKW are predicted to consume 6.6 times more PCBs through their diet than NRKW (Cullon et al. 2009 Ross et al. 2006).

2.7.5 PCB Loads in Resident Killer Whales Over Time

The burden of fat-soluble PCBs can be measured from samples of killer whale blubber. DFO began collecting biopsy samples of Resident and Transient Killer Whale blubber in 1993 (Ross et al. 2000). Analysis of data from the first three years of sampling (1993-1996; SRKW n=6; NRKW n=26; Transients n=15) found \sum PCB levels in SRKW were three times higher compared to NRKW, likely due to different contaminant exposure from habitat and diet. SRKW have a higher overlap with the population centres and industrial areas of southern BC and northern Washington State, which contaminant studies on seals indicate is an area highly contaminated with PCBs (Ross et al. 2004, 2006, 2013). The sampling program by DFO continued, extending the time series from 1993-2009 for NRKW and 1993-2004 for SRKW (Ross et al. 2013 unpubl.⁹; Guy 2018 unpubl.⁵). A separate US study collected SRKW samples in 2004, 2006, and 2007 (Krahn et al. 2007; 2009), and analysis indicated that PCB levels exceeded thresholds for health effects in marine mammals, and also that juveniles had significantly higher concentrations of POPs than adults due to maternal transfer (Krahn et al. 2009). A decrease in PCB levels was reported for SRKW sampled in 2004/2006 compared to 1993-1995 (Krahn et al. 2007). However, the analysis done by Krahn looked at 45 PCB congeners, and the analysis of previous samples consisted of 205 congeners (Ross et al. 2004, 2006, 2013 unpubl.⁹). Analysis of SRKW

⁹ Ross, P.S., Desforges, J-P.W., Dangerfield, N.J., Crewe, N.F., Dubetz, C.P., Fischer, M.B., Fraser, T.L., and Ross, A. R. 2013. Blubber concentrations of PCBs, PBDEs, PCDDs and PCDFs from 1993 to 2009 in killer whales (*Orcinus orca*) from the North East Pacific Ocean. Unpublished

samples collected by NOAA (USA) was conducted in 2015. In these samples Σ PCBs from the SRKW population ranged from 10-48 (geometric mean = 24) mg/kg lipid in males and 3-44 (geometric mean = 10) mg/kg lipid in females (Guy 2018 unpubl.⁵; Appendix II).

The compilation of all Σ PCB data collected to date on the killer whale populations in BC, allowed an analysis of trends over the complete time range (1993-2015 for SKRW, 1993-2009 for NRKW) (Source of PCB data: Krahn et al. 2007;2009; Ross et al. 2013 unpubl; Guy 2018 unpubl.⁵). PCB concentrations accumulated in killer whales varied by dietary preference, calving order, reproductive history, birth year and matriline membership (Pearce and Gobas 2018, unpubl.⁸). Male and female SRKW Σ PCB levels did not significantly change from 1993-2015, indicating that PCBs continue to persist in SRKW (Table 6, Table 7; Gobas and Ross 2017 unpubl.⁶). NRKW females' Σ PCB levels also showed no significant change 1993-2009). However, male NRKW samples did exhibit a significant decline over the same time period (Table 6; Gobas and Ross 2017, unpubl.⁶). The lack of an observed decrease in females was attributed to a trend being masked by the higher individual variability in values in females as a result of differences in reproductive success.

The observed differences in PCB load between males and females may indicate that reproductive offloading can introduce significant variability in these observations, particularly in populations where reproduction has been affected and pregnancy failures may be occurring. Male PCB levels may be a more reliable indicator of temporal PCB changes in KW exposure to PCBs, as they are less likely to have as much variability as females with different reproductive histories. This is supported by evidence from a Σ PCB trend analysis from the larger and healthier Resident Killer Whale population in Alaska, where declines in PCB levels have been observed in both sexes over a similar time period (Gobas and Ross 2017, unpubl.⁶).

A recent food web analysis (Pearce and Gobas 2018, unpubl.⁸) incorporated sediment Σ PCB values (Guy 2018 unpubl.⁵) into an existing bioaccumulation model (Lachmuth et al. 2010; Alava et al. 2012, 2016) to examine PCB transfer from sediments to killer whales in different areas. The sediment measurements that inform the model indicate significant declines in total PCBs in samples from the North coast (2011-2015) and BC Strait of Georgia (2010-2017), whereas samples in US SKRW habitat showed an increase (2010-2016), although the data did not have a good linear fit. The model estimates of Σ PCB concentrations in adult male and female RKW based on these sediment values were generally close to the observed, except for when using sediment from the US SRKW habitat. In these cases, the model greatly over predicted Σ PCB levels in killer whales, as Σ PCB levels in the sediments of Puget Sound are not representative of the rest of the SRKW habitat. While Puget Sound sediments are highly PCB contaminated, it is estimated that SRKW spend only 6% of their time in that area (Pearce and Gobas 2018 unpubl.⁸). In the BC areas of SRKW habitat, the model better predicted PCBs in killer whales, and SKRW are thought to spend 18% of their time in BC SRKW critical habitat and 3% in the BC Strait of Georgia (Lachmuth 2010; Pearce and Gobas 2018 unpubl.⁸).

Table 6 - Summarised Σ PCB data for male Northern and Southern Resident Killer Whales. Age categories are based on Vélez-Espino et al. 2014a; 2014b). The category for calves (<1) was excluded due to lack of data. Values under each mean represent the range of mean values within that category. Sample data where sex was unknown were excluded.

Eco-type	Date range	Source	Congeners analysed	Σ PCB geometric mean, and range mg·kg ⁻¹ lw							
				Juveniles (1-9)	n	Young males (10-21)	n	Older males (22+)	n	All adults males (10+)	n
NRKW	1993-1996	Ross et al. 2000	205	17.27 10.80-27.90	3	20.6 20.00-21.20	2	25.2 2.42-49.60	8	24.28 2.42-49.60	10
	1993-2009	Ross et al. 2013 unpubl. ⁹ (collated by Guy 2018 unpubl. ⁵)	205	10.62 3.27-27.90	10	10.54 2.60 - 23.70	21	19.32 0.72 - 49.60	13	13.79 0.72-49.60	34
SRKW	1993-1996	Ross et al.	205	-	-	63.20 sole value	1	119.80 5.93-192.0	3	105.78 5.93-192.00	4
	1993-2009	Ross et al. 2013 unpubl. ⁹ (collated by Guy 2018 unpubl. ⁵)	205	-	-	27.93 8.53 - 63.20	4	151.98 5.93-248.00	4	89.96 5.93-248.00	8
	2004, 2006 & 2007	Krahn et al. 2007; 2009	45	38.00 34.00-41.00	3	40.43 22.00-74.00	7	91.33 38.00-180.00	3	55.70 22.00-180.00	10
	2015	Guy 2018 unpubl. ⁵	209	37.60 27.70-47.50	2	-	-	10.30 sole value	1	10.30 sole value	1
	1996-2015	All combined (Ross et al. 2000; 2013; Krahn, 2007; 2009; Guy 2018 unpubl. ⁵)	45, 205 and 209	37.84 27.70-47.50	5	35.88 8.53-74.00	11	111.53 5.93-248.00	8	67.73 5.93-248.00	19
	1996-2015	All combined excluding Krahn data	205 and 209	37.60 37.60-47.50	2	27.93 8.53-63.20	4	123.65 5.93-248.00	5	81.11 5.93-248.00	9

Table 7 - Summarised Σ PCB data for female Northern and Southern Resident Killer Whales. Age categories are based on Vélez-Espino et al. 2014a; 2014b). The category for calves (<1) was excluded due to lack of data. Values under each mean represent the range of mean values within that category. Sample data with unknown sex were excluded.

Eco-type	Date range	Source paper	Congeners analysed	Σ PCB geometric mean, and range mg·kg ⁻¹ lw									
				Juveniles (1-9)	n	Young females (10-30)	n	Older reproductive (31-50)	n	Post reproductive (51+)	n	All adult females (10+)	n
NRKW	1993-1996	Ross et al. 2000	205	40.67 9.80-109.00	4	7.09 0.48-15.40	6	1.81 1.04-2.58	2	19.65 9.45-25.50	3	9.56 0.48-25.50	11
	1993-2009	Ross et al. 2013 unpubl. ⁹ (collated by Guy 2018 unpubl. ⁵)	205	25.76 4.12-109.00	12	7.79 1.37-25.00	21	2.38 1.04-6.37	5	17.1 9.45-25.5	3	7.87 0.48-25.50	30
SRKW	1993-1996	Ross et al., 2000	205	-	-	74.70 sole value	1	34.70 sole value	1	-	-	54.70 34.70-74.70	2
	1993-2009	Ross et al. 2013 unpubl. ⁹ (collated by Guy 2018 unpubl. ⁵)	205	-	-	74.70 sole value	1	34.70 sole value	1	-	-	54.70 34.70-74.70	2
	2004, 2006 & 2007	Krahn et al. 2007; 2009	45	62 sole value	1	17.97 4.30-45.00	3	8.90 sole value	1	67.33 27.00-120.00	3	37.83 4.30-120.00	7
	2015	Guy 2018 unpubl. ⁵	209	-	-	16.85 3.01-44.10	5	4.83 sole value	1	-	-	14.85 3.01-44.10	6
	1996-2015	Guy 2018 unpubl. ⁵ (collated Ross et al. 2000; 2013; Krahn 2007; 2009; Guy 2018 unpubl. ⁵)	45, 205 and 209	62 sole value	1	23.65 3.01-74.70	9	16.14 4.83-34.70	3	67.33 27.00-120.00	3	30.89 3.01-120.00	15
	1996-2015	As above, but excluding Krahn samples	205 and 209	-	-	26.50 3.01-74.70	6	19.77 4.83-34.70	2	-	-	24.81 3.01-74.70	8

2.7.6 Biological Impacts of PCBs [7,8]

PCBs can have a variety of dose-dependent toxic effects on mammals including immunosuppression, reproductive impairment, and impacts to the endocrine system (Buckman et al. 2011; Jepson et al. 2016; Lundin et al. 2016; Mongilo et al. 2016). Reproductive impacts are of particular note as PCBs have been implicated in decreases in cetacean reproductive success [6] and increases in calf mortality [8], suggesting they may have an important role in population declines and suppression of population recovery in some killer whale populations (Jepson et al. 2016; Hall et al. 2018; Desforges et al. 2018). In grey seal pups, maternally-transferred PCBs disrupt glucose uptake and lactate production in fat (adipose) tissues, and POP levels in 3 week old seal pups are high enough to impact adipose function (the ability to regulate and generate blubber), potentially impacting survival (Hall et al. 2001; Robinson et al. 2018). The effects that POPs have on adipose function is considered to be highest in young marine mammals, and are modified by nutritional state and the depth of blubber (Robinson et al. 2018). For example, post-weaning, POP levels in fasting young seal pups can increase further as lipids mobilise, with lipophilic POPs concentrating in remaining blubber and less lipophilic POPs being released into the blood (Debieer et al. 2003a; 2003b; 2006; Louis et al. 2016). This may be relevant to other marine mammals such as killer whales as elevated POP levels have been associated with altered adipose gene expression (Buckman et al. 2011).

However, direct health effects of PCB contamination on RKW have not been clearly demonstrated or quantified to date due to the challenge of determining causal relationships amidst other factors influencing health (Hickie et al. 2007; Pearce and Gobas 2018, unpubl.⁸). One way to address this is to infer toxicological effects from other mammals and use these relationships in bioaccumulation models. Combining findings from related studies can be used in a 'weight of evidence' approach to assess population-level toxicological risk, as done by Ross (2000). In the absence of killer whale-specific information, a PCB bioaccumulation/depuration model developed by Hall et al. (2006, 2018), based on experiments on a land mammal, has been used to model the impact of PCBs on population growth as a reduction in survival of killer whale calves based on the PCB levels of their mothers at the time of their birth [8]. How PCBs impact male fecundity and the way this contributes to the overall impacts of PCBs on RKW population fecundity is uncertain, though there is some evidence that high PCB levels may impact testes development. An 18 year old male SRKW (J18) that died in 2000 had high PCB levels and undeveloped testes, unusual at that age, possibly indicating maturity was affected by PCB contamination (K. Balcomb, Center for Whale Research, WA, pers. comm.). Though this aspect has not been considered here, it is an area that may be explored further using necropsy data, and potentially incorporated into future iterations.

A recent study compared PCB concentrations in worldwide killer whale populations to concentration-response relationships for reproductive impairment and immunotoxicity-related disease mortality (Desforges et al. 2018; based on Hall et al. 2018) [7,8]. The Desforges et al. (2018) study concluded that more than half of the world's killer whale populations are at risk of long-term population level effects as a result of the impacts of PCBs on reproduction and immune function. The model linked PCB impacts on immunity to the probability of survival using relationships between immune suppression and disease mortality (Luster et al. 1993).

With normal reproductive activity, PCB levels in females should decrease over time due to offloading from mother to calf during gestation and lactation. Levels should then increase again after reproductive senescence (a U shaped pattern, as observed in NRKW by Ross et al. 2000). In cases where levels in females are not showing this decrease, but follow similar trends to males, it may indicate reproductive failure, such as is the case in some highly contaminated European populations (Jepson et al. 2016). In industrialised areas of Europe, only small killer

whale populations remain, with very low reproductive rates. The one remaining resident fish eating population in southern Europe has only 36 members, with six reproducing females, who have produced only five calves (surviving past 1 year) over 13 years of monitoring (1999-2011). Σ PCB levels in this resident tuna-eating KW population in the Strait of Gibraltar are very high, ranging from 172-315 mg/kg lipid for males and 43-858 mg/kg lipid for females (Jepson et al. 2016). The mean PCB levels for Strait of Gibraltar adult females (215 mg/kg lipid) were almost four times higher than those reported for SKRW females (55.4 mg/kg lipid - Ross et al. 2000; 2013; Krahn 2007, 2009; Desforges et al. 2018; Guy 2018, unpubl.⁵) and twice as high as in Bigg's killer whales (109 mg/kg lipid - Ross et al. 2013 unpubl.⁹).

2.7.7 Interactions, Threshold and Nonlinearities [12,13]

The impacts from PCBs are unlikely to be characterised by a simple linear impact-effect relationship, as there are other threats and factors present for both populations (Buckman et al. 2011) that may interact with or modulate the impact of PCBs, as well as other contaminants present that may also have impacts. As noted above, the effects of PCBs on killer whales may be mediated by nutritional stress and the amount of blubber stores, as observed in seals (Robinson et al. 2018) [12,13]. The PCB contamination and prey availability threats may interact, because killer whales suffering nutritional stress (from lack of prey) will metabolise adipose (fat) tissue in blubber, resulting in the mobilisation of lipophilic toxins such as PCBs into the bloodstream and causing a toxic response (Krahn et al. 2002, Mongillo et al. 2016). It has been proposed that at these times, systematic POP concentrations and associated bioavailability to organs increases (Aguilar et al. 1999; Lundin et al. 2016). Nutritional stress resulting from a lack of Chinook salmon may act synergistically with high contaminant burden resulting in higher calf mortality and reduced fecundity (Mongillo et al. 2016) [12,13]. Though the Biggs population have higher contaminant loads in the blubber, the population is reportedly healthy and increasing (Ford et al. 2007). Blubber-bound toxin levels may be higher in Bigg's whales but they may not have the same toxic effects as in prey-limited populations that are mobilising the toxins as a result of nutritional stress. Some support for this was provided by Lundin et al. (2016b) who examined the relationship between PCB levels measured from SRKW scat and Chinook salmon abundance and found that concentrations of contaminants were highest and had the highest toxic potential when prey abundance was the lowest, and that the contaminants likely originated from endogenous lipid stores (Lundin et al. 2016).

2.8 PATHWAYS OF EFFECTS DISCUSSION

The development of the Pathways of Effects conceptual model provides an illustration and summary of the evidence for the structure of the system under investigation. This structure forms the basis for the population viability analysis modelling in the subsequent section of the paper. In the proposed PoE model for Resident Killer Whales, prey availability appears to be a central node, with six linkage pathways to fecundity and mortality, including two interactions with other threats. The interactions make the assessment of impacts more difficult, as they imply that impacts are not additive and may have non-linear or threshold effects.

Additional threats to Resident Killer Whale populations were identified in the SARA recovery strategy (DFO 2018a) that were not included in the current model, and include contaminants other than PCBs and PBDEs, incidental mortality in fisheries, oil spills, disease, harmful algal blooms, as well as seismic exploration and other high-intensity sounds.

Incidental injury and mortality in fisheries appears to be a rare occurrence in Resident Killer Whale populations. Stranded killer whales have been found with fishing gear in their stomachs (Ford et al. 1998) and a small number of entanglements have been reported, but most were not fatal (DFO 2018a).

There is a low probability of killer whales being exposed to a major oil spill, but if they are exposed there can be serious consequences. The 1989 Exxon Valdez oil spill in Alaska was strongly linked to an unprecedented mortality event, after killer whales were seen surfacing in the oil slick (Matkin et al. 1999). It was hypothesised that mortality was caused by the inhalation of petroleum vapors (Matkin et al. 1999).

Three pathogens have been found in wild killer whales: *Brucella*, *Edwardsiella tarda*, and cetacean poxvirus. *Brucella* can impact killer whale reproduction, potentially affecting fecundity by causing abortions. An *E. tarda* infection was the cause of death in a SRKW male in 2000, and cetacean poxvirus has been implicated in calf mortality (Gaydos et al. 2004; Ford et al. 2000; Van Bressemer et al. 1999; DFO 2018a).

Harmful algal blooms have been identified as a possible threat to Resident Killer Whales (DFO 2018a). There may be a risk to killer whales if they are exposed to the toxins released from a harmful algal bloom, given that mortalities of other marine mammals have been linked to exposure to biotoxins (Krahn 2002; DFO 2018a). Mass mortality events such as that associated with a toxic bloom of the harmful alga *Alexandrium tamarense* in the St. Lawrence Estuary in August 2008 provided strong evidence for trophic transfer of algal toxins through the food web (Starr et al. 2017). Mortalities of marine fishes, birds and marine mammals including beluga whales (*Delphinapterus leucas*), harbour porpoises (*Phocoena phocoena*), a fin whale (*Balaenoptera physalus*), and several species of seals (mainly grey seals, *Halichoerus grypus*) were associated with this bloom. In BC, harmful algae have been linked to survival of wild salmonids; marine survival of sockeye salmon *Oncorhynchus nerka* (Fraser River Chilko stock) was more than four times higher in years when there were minor or no blooms of the alga *Heterosigma akashiwo* in the Strait of Georgia during the juvenile sockeye outmigration period versus years with major blooms (Rensel et al. 2010).

Seismic surveys employ airguns, which generate high intensity sounds. Cetacean species have mixed reactions to seismic surveys; some have been shown to avoid areas where seismic surveys are occurring (Stone 2003; Calambodkis et al. 1998). The current moratorium on oil and gas exploration in BC ensures that permits for seismic surveys are rarely issued. If the moratorium were lifted, this threat could become significant, especially to NRKW.

The pathways of effects conceptual model is iterative in nature, and should be reassessed at intervals to utilise new data being collected and to capture current research and understanding of the components making up this system.

2.9 PATHWAYS OF EFFECTS USED IN THE POPULATION VIABILITY MODEL

Based on the review of the available literature and data, only portions of the PoE conceptual model could be parameterised with any confidence in the Population Viability Analysis. The linkage pathways (both direct and interaction) that were not able to be parameterised and were removed from this assessment are identified by grey lines in the overall pathways of effects model in Figure 8. The structure of the final modified PoE diagram used in the PVA model in this assessment, (with the grey lines removed), is outlined in Figure 9. The threat names differ on the PoE diagram used to inform the PVA, as they represent the specific aspects used to parameterise threats in the assessment. The Disturbance (acoustic) threat is represented by the combined effects of vessel noise and vessel presence, as at present there is no way to tease apart impacts from vessel presence from vessel noise. Although there is some evidence in other cetaceans that there may be an interaction between acute acoustic disturbance and strandings or vessel strikes, this was not included in the PVA model. The Disturbance (physical) threat is represented by the effects from vessel strikes. Prey availability is represented by Chinook salmon abundance in the PVA model, even though it is acknowledged that other types of

salmon are also consumed. For the contaminants threat, despite the evidence that other contaminants are present in Resident Killer Whales, only PCBs were included. The details of how threats were parameterised is described in the following PVA section.

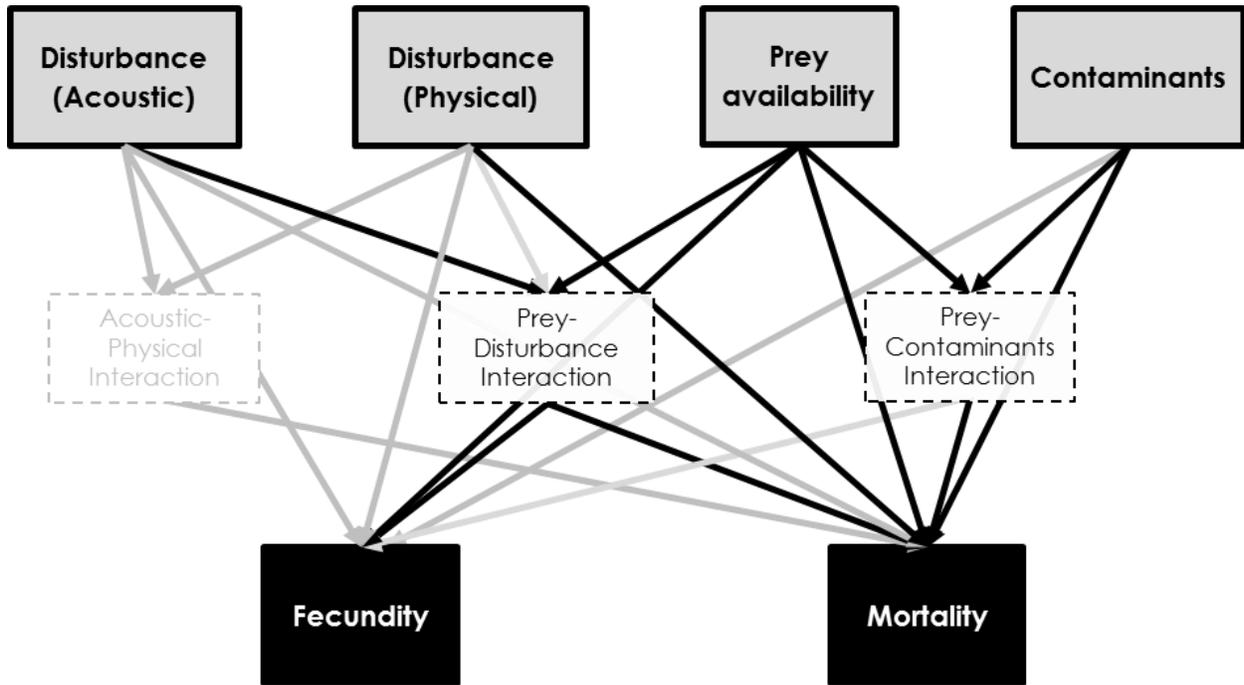


Figure 8 – Overall pathways of effects model for Resident Killer Whale populations highlighting the direct linkage pathways and interaction linkage pathways that were able to be quantified (black lines) and those which could not be quantified in this assessment (grey lines).

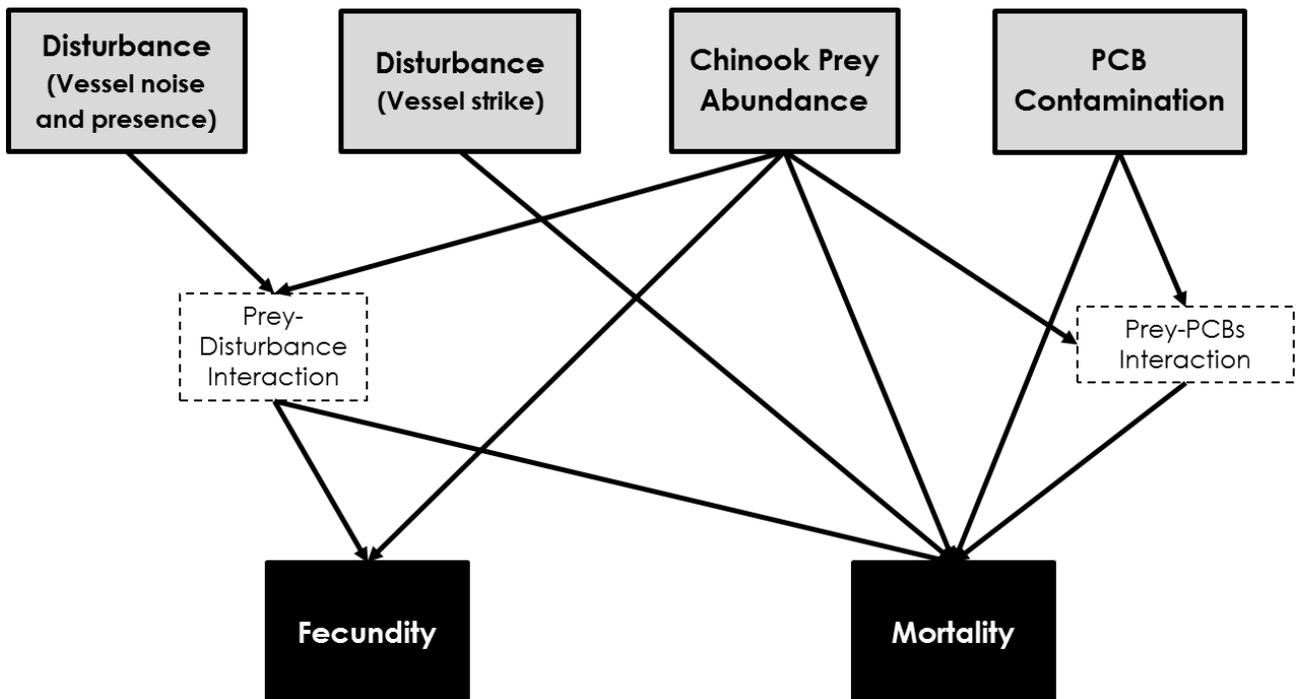


Figure 9 - Modified PoE conceptual model for Resident Killer Whale populations used for population viability analysis (PVA).

3 POPULATION VIABILITY MODEL

This cumulative effects assessment for NRKW and SRKW builds upon the methods and results of previous work (Taylor and Plater, 2001; Ward et al. 2009; Vélez-Espino et al. 2014a, 2014b; Williams et al. 2017; Lacy et al. 2017; DFO 2018a) and includes recent research advances and the compilation and review of unpublished data and results.

3.1 KILLER WHALE POPULATION MODEL METHODS

3.1.1 Killer Whale Census

Population models were constructed for each of the SRKW and NRKW populations separately using census data from DFO's Cetacean Research Program encompassing 1979-2017 (DFO CRP, unpublished data). Annual population surveys have occurred without interruption since 1973 for NRKW and 1976 for SRKW (DFO Cetacean Research Program; Center for Whale Research, CWR). For the purposes of this assessment, census data from 1979 onwards was used. By using demographic rates starting in 1979, the time series is composed mostly of data from direct observations rather than reconstructed data (Olesiuk et al. 1990).

Each annual census consists of photo-identification surveys in which individuals are identified using their unique fin shapes and saddle patch colouration (Bigg et al. 1987; Ford et al. 2000; Baird 2000, 2002). Census data were used to determine genealogical relationships and estimate life history parameters (Bigg et al. 1990; Olesiuk et al. 2005; Vélez-Espino et al. 2014a; 2014b), capturing birth and death information. A death is recorded when an individual is not observed with its matriline on several subsequent encounters where all other members of the matriline are present (this may be anywhere from one to several survey seasons, depending on the frequency that the given matriline is documented). Cause of death is difficult to assign, as killer whale carcasses are not often recovered (Olesiuk et al. 2005). When a birth is recorded, the assignment of mother to each calf is based on the observation of close associations following birth, during the same photo-identification surveys. The NRKW and SRKW populations are fully surveyed, with every individual in the population identified and tracked. Therefore there is no uncertainty around the population size numbers and any increases in population are not the result of increased sampling effort. The comprehensive census data allows for a detailed understanding of life history parameters for these Resident Killer Whale populations that can be used in population modelling.

3.1.2 Life History Parameters

The killer whale reproductive system was defined as polygynous and sexually dimorphic with observed population parameters (Table 8; Olesiuk et al. 2005; Vélez-Espino et al. 2014a; Ward et al. 2010).

Table 8 - Life history parameters for Northern and Southern Resident Killer Whales used in the population models.

Life history parameter	Males	Females	Reference
Age of sexual maturity (y)	12-18	12-18	Olesiuk et al. 2005
Maximum age of reproduction (y)	70	50	Vélez-Espino et al. 2014a
Maximum lifespan (y)	70	90	Vélez-Espino et al. 2014a; Olesiuk et al. 2005
Maximum number of calves/brood	-	1	Olesiuk et al. 2005
Sex ratio at birth	0.5	0.5	Olesiuk et al. 2005
Gestation time (months)	-	17	Olesiuk et al. 2005

Sexual maturity can begin as young as 10 y in males (Olesiuk et al. 2005); and is evident when the dorsal fin begins growing in height (Olesiuk et al. 2005). Males are presumed to be reproductively active throughout their lives and breeding success increases with age (Barrett-Lennard 2000; Olesiuk et al. 2005; Ford et al. 2011). There is some evidence that older, larger males are preferred as mates; for SRKW, only two males have sired half of the calves born since 1990 (Ford et al. 2018). It is unknown whether male breeding success is similar limited in other populations or if this is a consequence of the small population size in SRKW. In females, calving begins around age 10 years (mean age of first conception), a female's first surviving calf is most often born when she is 12-17 years old (Olesiuk et al. 2005; National Marine Fisheries Service 2008). One calf is carried per gestation, with an average gestation time of 17 months (range 15-18 months) (Olesiuk et al. 2005). Fecundity declines with age in females, with reproductive senescence occurring after age 50 y (Vélez-Espino et al. 2014a). In the model, individuals were randomly assigned an age of maturity between 12-18 y (Olesiuk et al. 2005), and the maximum age assigned for reproduction was 50 y for females (age of senescence) and 70 y for males (maximum lifespan) (Olesiuk et al. 2005; Vélez-Espino et al. 2014a).

Resident Killer Whale offspring remain with their natal family group or matriline (Bigg et al. 1990) and mating primarily occurs between individuals from different matrilines (Barrett-Lennard 2000; Ford et al. 2011). In the case of NRKW, mating typically occurs between individuals from different acoustical clans (Barrett-Lennard 2000). There is no evidence of interbreeding or dispersal between populations, and SRKW and NRKW populations are acoustically, genetically, and culturally distinct (Barrett-Lennard 2000; Barrett-Lennard and Ellis 2001; Ford et al. 1998; 2000; Ford et al. 2011). Calves are dependent on their dams for 1 year after birth, so if a mother died in the model simulations in the first year after birth of the calf, her calf also died. Sex ratio at birth was assumed to be equal.

3.1.3 Population Parameters

Killer whale vital rates (mortality and fecundity) have been estimated for a series of age classes within each population (Table 9a,b). Individual-based population models were constructed in the population modelling software Vortex (Lacy and Pollak 2014). The life cycles of these two populations were modeled as two-sex stage-structured models.

Population sizes and vital rates before the impact of human activities, such as aquarium removals, are unknown. Carrying capacity is therefore difficult to estimate, but is likely influenced by a combination of prey abundance, habitat requirements and limits imposed by the social structure of the populations. For this modelling exercise, carrying capacity was set sufficiently high as to have no effect on the simulated populations (carrying capacity for SRKW: 300; NRKW: 500 individuals), but the relationship between vital rates and prey abundance was included (see Section 3.3.3).

Small populations can be susceptible to the effects of inbreeding depression. A recent analysis of genotypes from the SRKW population suggested that inbreeding may be occurring, however there was limited evidence that these individuals have lower survival or fecundity than normal (Ford et al 2018). We have little information on inbreeding in cetaceans. Previous PVA efforts have used various levels of lethal equivalents to estimate the effect of inbreeding on reproductive success. Taylor and Plater (2001) used 2.0 lethal equivalents from the value for humans and chimpanzees (Ralls et al. 1988), while Manlik (2016) used the mid-range value for inbreeding (3.14 lethal equivalents) from Ralls et al (1988). In the current model, the mean value for wild species (O'Grady et al. 2009) was used: 6.29 lethal equivalents (Lacy et al. 2017).

Table 9a - Age-specific mortality rate for each Resident Killer Whale population: Southern Resident Killer Whales (SRKW), Northern Resident Killer Whales (NRKW) and Southern Alaska Resident Killer Whales (SARKW).

Age class (years) (male and female combined)	SRKW ¹	NRKW ¹	SARKW ²
0-1	0.215 (SD=0.284)	0.078 (SD=0.082)	0.054 (SD=0.244)
1-2	0.019 (SD=0.047)	0.028 (SD=0.019)	0.003 (SD=0.040)
2-5	0.019 (SD=0.047)	0.028 (SD=0.019)	0.010 (SD=0.054)
6-10	0.019 (SD=0.047)	0.028 (SD=0.019)	0.012 (SD=0.064)
10-16	0.015 (SD=0.033)	0.011 (SD=0.012)	0.008 (SD=0.032)
17-51	0.033 (SD=0.054)	0.011 (SD=0.025)	0.023 (SD=0.066)
51+	0.072 (SD=0.108)	0.117 (SD=0.114)	0.217 (SD=0.292)

¹ Vélez-Espino et al. 2014a – 1987-2011 ² Matkin et al. 2014 – 1984-2010

Table 9b- Age-specific fecundity rate for females in each Resident Killer Whale population: Southern Resident Killer Whales (SRKW), Northern Resident Killer Whales (NRKW) and Southern Alaska Resident Killer Whales (SARKW).

Age class (years) Females only	SRKW ¹	NRKW ¹	SARKW ²
10-30	0.116 (SD=0.077)	0.142 (SD=0.046)	0.233 (SD=0.118)
31-50	0.069 (SD=0.074)	0.101 (SD=0.051)	0.154 (SD=0.118)

¹ Vélez-Espino et al. 2014a – 1987-2011 ² Matkin et al. 2014 – 1984-2010

3.1.4 Model Structure

The Southern Alaska Resident Killer Whale (SARKW) population has a similar life history strategy but is relatively removed from the threats to which the SRKW and NRKW are exposed. To define the reference conditions for the current work, we used the mortality and fecundity rates that have been estimated from SARKW census data (census began in 1984) (Table 9a,b), as the rates expected from a population in unrestrained growth. The SARKW population is not considered to be pristine as it is exposed to anthropogenic impacts, and was notably impacted in 1989 by a major oil spill (Exxon-Valdez), resulting in a 33% loss of the resident AB matriline (Matkin et al. 2008). However, these impacts are not incorporated into the vital rates for SARKW (Table 9a,b) as these anomalous deaths were excluded from the data analysis in Matkin et al. (2014). The rates and age/sex structure of the SARKW population were found to be similar to NRKW in their period of unrestrained growth, except that the age of maturity was one year earlier for SARKW (Olesiuk et al. 2005; Matkin et al. 2014).

The SARKW vital rates data (Table 9a,b) were used in the SRKW and NRKW population models to represent the reference vital rates that determine the growth of each population in the absence of anthropogenic threats. This is an important change from the SRKW model defined by Lacy et al. (2017), where the “baseline” was defined using the mean demographic rates that were observed from recent decades and would therefore include current threat levels. Model scenarios were developed based on individual and cumulative threats. The threats (described in

further detail in section 3.3) were then included in the model as modifiers of the SARKW reference vital rates.

The population genealogical and demographic data were partitioned to allow model validation and verification; the complete set of living animals in the year 2000, with their known dams, sires, calving histories, and genealogies, were used as the starting population for each of the population models (SRKW and NRKW). This allows a comparison of the modeled populations to the observed populations as an evaluation of the ability of the model scenario output data to represent observed data.

Model simulations were run on each scenario 10,000 times and summary statistics were recorded for population growth rate (r), population size at each time step (N_t), and probability of extinction. The population size at each year (mean and standard deviation) was compared to the observed (realised) population size for each population from the census survey data. Population growth rate (r) was quantified as the exponential rate of increase, according to the following equation:

$$r = \ln\left[\frac{N_{t+1}}{N_t}\right]$$

The model results (the predicted population size resulting from threat-modified reference vital rates) were then compared to the observed (realised) population dynamics from the census data over the same time period (2000-2017). The assumption of this approach is that if we can define a model that replicates the realised dynamics for both the SRKW and NRKW populations, we have constructed an appropriate model for the system.

The observed population growth rate (r) for the two populations from 2000-2017 was -0.002 for SRKW and 0.024 for NRKW. In 2017, the population size was 77 and 308 individuals for SRKW and NRKW, respectively.

3.2 MODEL VERIFICATION AND VALIDATION

3.2.1 Inspection Approach

In order to determine how representative the simulation output data were, we examined the simulated population size, population growth rate, age structure, and sex ratio for each scenario. We used the inspection approach method to validate the models (Law et al. 1991). For the model scenarios that most closely approached the observed population parameters, we ran the simulation scenario with historical input data in place of the parameter chosen from a distribution, in this case the yearly Chinook salmon index data. A valid model should closely resemble the observed killer whale survey data when the historical salmon index data are used (details of the index data in Section 4.3.2).

3.2.2 Sensitivity Analysis

Sensitivity analysis was conducted on key parameters in the model to test the impact of uncertainty in these parameters on the results of the study. Sensitivity analysis was conducted in Vortex using the Sensitivity Testing operations. Parameters that required sensitivity analysis included: noise impact value, vessel-KW overlap, PCB impact on calf mortality, and vessel strike.

3.3 THREAT SCENARIOS

3.3.1 Aquarium Removals / Live Capture Fishery

The removal of killer whales for display in aquaria is a historic threat that complicates population modelling, in particular for the SRKW population. Comprehensive population surveys on the SRKW and NRKW populations only began soon after the end of the live capture fishery. This means that the population composition at the start of the data set is skewed and very different to the composition at present, an issue highlighted in the predictions made by Vélez-Espino et al. 2014b. Though this is not a current threat, it is important to consider in population predictions due to its long-term influence on the population.

Because of their location closer to human population centres, most removals (48 individuals) were from the SRKW population - an estimated 36% of the total population. To illustrate the residual impact of this historic threat, the 48 animals removed from the SRKW population were added into the population model at year 1980, comprised of 30 juveniles (19 males and 11 females) and 18 mature adults (10 males and 8 females). The modified demographic structure and observed vital rates over the time period (1979-2017) from Vélez-Espino (2014a) were used to project the population growth rate from the year 1980 until the present (2017).

3.3.2 Chinook Salmon Abundance

The Pacific Salmon Commission's (PSC) Chinook model estimates the number of "model fish" available from each of the 30 model stocks to 25 fisheries. Ford et al. (2010) reported that RKW survival rates were related to the modelled abundance of these Chinook stocks available to six fisheries (Alaska Troll, BC North Troll, BC Central Troll, West Coast Vancouver Island Troll, Georgia Strait Sport, and Washington/Oregon Troll). Three different sources of data have been used to represent Chinook salmon abundance in RKW survival investigations (Ford et al. 2009, 2010; Vélez-Espino et al. 2014b; Stredulinsky 2016): PSC Chinook model terminal run, Coded-Wire Tag (CWT)-based terminal run reconstruction estimates, and ocean abundance estimates from the PSC Chinook model. The PSC Chinook Model terminal run estimates include both hatchery and natural production plus terminal catch estimates and is available for eight Canadian stocks. The CWT-based terminal run reconstruction estimates stock abundance for runs from northern BC through California. It uses escapement information and CWT data from fisheries across the coast to reconstruct abundance from spawning areas as well as ocean fisheries to estimate fish abundance in terminal run areas. Ocean abundance includes reconstructed estimates of numbers of fish removed in fisheries as well as those escaping to spawning grounds to estimate the number in ocean. The ocean abundance is a measure of fish sufficient in size to be vulnerable to fishing gear (larger than the minimum size limit) and is therefore not an estimate of the total number of fish. The ocean abundance, estimated by the PSC Chinook model, does not represent absolute abundance, rather it represents the vulnerable fish from specific modelled stocks that are available to certain fisheries. Therefore, relative rather than absolute changes in abundance should be used.

Updated Chinook salmon ocean abundance data were obtained from DFO Salmon Program (A. Vélez-Espino, DFO, Pacific Biological Station) (1979-2017) from the 2018 PSC Chinook model calibration (Appendix I). Ocean abundance is an adequate representation of fish available for consumption by killer whales, given that the full time series of terminal run reconstruction data was not available, and ocean abundance has statistical support in previous analyses (Vélez-Espino et al. 2014b; Stredulinsky 2016).

Yearly model ocean abundance was converted to an index of abundance by standardising the value by the mean for the full time series. The Chinook index value was assigned in each model

year using either a uniform or normal distribution. The uniform distribution was defined by the full range of Chinook index values recorded during the time series (1979-2017). The normal distribution was defined by the median value (for a skewed distribution) and standard deviation from the entire time series (1979-2017). Selecting a value from a distribution in each year allows the model to represent the fine temporal structure and variation in Chinook salmon abundance, and its impacts on killer whale vital rates. However, this yearly random selection does not include the autocorrelation inherent in Chinook populations where fish from a cohort will contribute to the index value over a 4-year period.

Mortality

The relationship between prey availability and Resident Killer Whale mortality was first described by Ford et al. (2009) and a number of significant interactions between Chinook salmon abundance and RKW vital rates have been identified (Vélez-Espino et al. 2014b). However, in the more recent analysis, the effects of these interactions on RKW population growth and viability were relatively small and uncertain (Vélez-Espino et al. 2014b), bringing into question whether Chinook salmon resource availability alone can explain the recent observed population trends.

The effects of prey availability on mortality can depend on age class. Vélez-Espino et al. (2014b) found differences between the dependence of Female 1 (young reproductive females), Female 2 (old reproductive females), and Juvenile stage survival on various Chinook salmon stocks.

The relationship between vital rates and Chinook salmon ocean abundance index values was updated using the entire time series of RKW and Chinook salmon data (1979-2017). The Coastwide Index (excluding southeast Alaskan [SEAK] stock) as well as Chinook salmon runs deemed relevant to each population, with a one-year time lag, were investigated using linear regression. For SRKW, the WCVI + FL + OC runs were used (West Coast Vancouver Island, Fraser Late, and Oregon Coastal) and for NRKW the FE + PS + URB were used (Fraser, Puget Sound, and Upper Columbia River Bights) (Table 10). The model abundance was converted to an index by standardising with the mean over the entire time series (1979-2017). The Chinook salmon stock index that best explains the mortality patterns seen in both populations was tested using model selection (Akaike Information Criterion, AIC).

Table 10 - Chinook salmon ocean abundance indices used for mortality and fecundity analyses in each of the salmon threat models. The mean ocean abundance model estimates (OA), and the minimum and maximum values for the index are shown.

Killer Whale Population	Salmon Index	Mean OA (# fish)	Index min	Index max
SRKW	Coastwide (excl. SEAK)	1,104,884	0.608	1.445
	WCVI + FL + OC	373,151	0.467	1.611
	WCVI	113,778	0.224	2.371
NRKW	Coastwide (excl. SEAK)	1,104,884	0.608	1.445
	FE + PS + URB	261,052	0.584	1.818
	WCVI	113,778	0.224	2.371

The model selection Akaike Information Criterion (AIC) values for the WCVI and NRKW-SRKW stocks are close in relative value and cannot be excluded from further investigation (Table 11; Burnham and Anderson 2004). There is little support for the use of the Coastwide index to explain mortality in the updated analysis. The relationship between killer whale mortality and Chinook salmon abundance used all of the relevant stocks for each killer whale population. As done in previous analyses, the data for both killer whale populations were combined and the linear relationship between Resident Killer Whale mortality and the Chinook salmon index is represented by $(y = 1.6773 - 0.673x, r^2 = 0.0889, p = 0.012)$ (Figure 10). The previous Ford et al (2009) analysis used data up to 2003 and the additional fourteen years of data has reduced the explanatory power of the prey-mortality relationship, suggesting that additional threats or impacts may be necessary to explain the population fluctuations. To compare to previous analyses, varying stocks were used in the scenario modelling; the relevant stocks used for each model scenario are shown in Table 10.

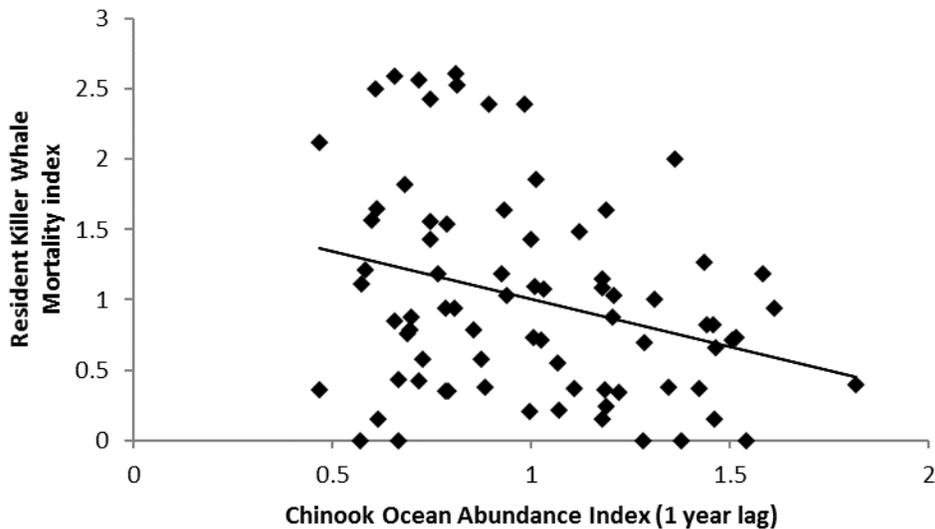


Figure 10 - Relationship between Chinook salmon index (1 year lag) for the stocks relevant to each killer whale population (WCVI + FL + OC for SRKW and FE + PS + URB for NRKW) and mortality index (difference between mortality and the long-term mean for each population). Data from 1979-2017.

Table 11 - Results of linear regression model fit analysis for the three Chinook salmon ocean abundance indices and Resident Killer Whale mortality, and their Akaike Information Criterion (AIC) values. The AIC value in bold is the lowest value, and AIC_{min}. The change in AIC (ΔAIC) gives the level of support for alternate models. Models with relative AIC values greater than 10 have essentially no support, a value less than two suggests there is substantial support for the *i*-th model.

RKW Mortality model	AIC	$\Delta AIC = AIC_i - AIC_{min}$
Coastwide Chinook Index	-15,461.8	105.8
NRKW-SRKW Chinook Index	-15,362.5	6.5
WCVI Chinook Index	-15,356.0	-

Fecundity

The availability of prey can also have significant effects on reproductive success and the probability of calving. Ward et al. (2009) assessed calving probability (fecundity) of combined NRKW and SRKW females using a logistic regression model and found that RKW fecundity was highly correlated with the PSC index of Chinook salmon abundance for the WCVI troll and recreational fishery in the prior year (one year lag). The model that best supported the data included age-structured effects on reproduction and a region effect. Though the populations were combined in this study, the inclusion of a regional effect represented the lower calving rates in SRKW than NRKW.

The logistic regression analysis was repeated with the additional 10 years of data for calving probabilities and PSC Chinook model ocean abundance salmon indices, following the statistical methods of Ward et al. (2009). The WCVI stock ocean abundance was used in the update, which differs than the data series used in Ward et al. (2009). The best model to explain calving probability (lowest relative AIC value) included the relevant Chinook salmon stocks – the NRKW Chinook index for NRKW and the SRKW Chinook index for SRKW (Table 12) and included an age structure (Figure 11).

Table 12 - Results of logistic regression model fit analysis for fecundity and the four Chinook salmon ocean abundance indices, and their AIC values. The AIC value in bold is the lowest value, and AIC_{min} . The change in AIC (ΔAIC) gives the level of support for alternate models. Models with relative AIC values greater than 10 have essentially no support, a value less than two suggests there is substantial support for the i -th model.

Population	Fecundity model	AIC	$\Delta AIC = AIC_i - AIC_{min}$
NRKW	NRKW Chinook Index	1910.6	--
	SRKW Chinook Index	1912.8	2.2
	Coastwide Chinook Index	1911.3	0.7
	WCVI Chinook Index	1912.3	1.7
SRKW	NRKW Chinook Index	696.8	1.6
	SRKW Chinook Index	695.2	--
	Coastwide Chinook Index	696.6	1.4
	WCVI Chinook Index	696.71	1.5

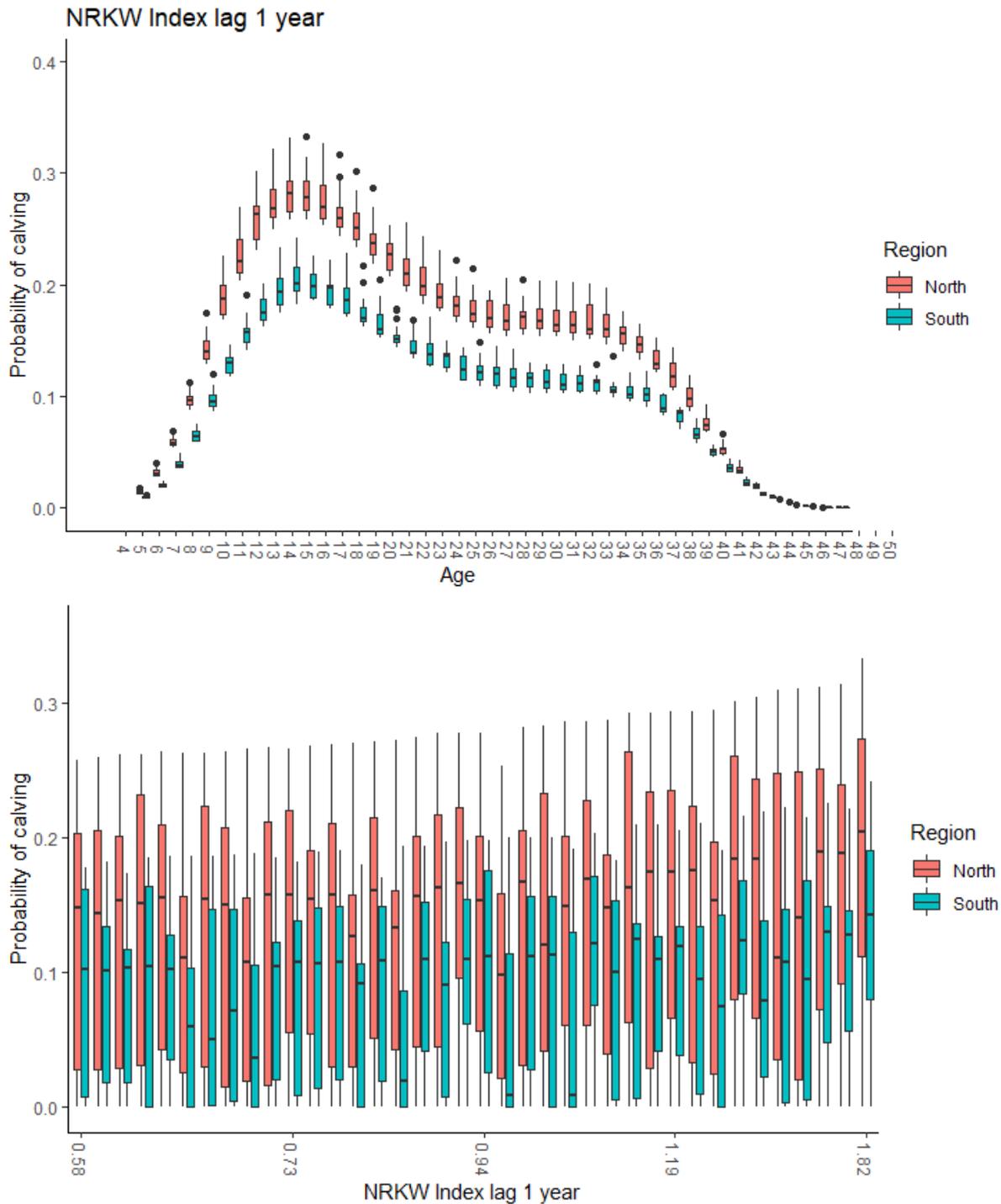


Figure 11 - Calving probability by age (upper panel) and calving probability related to the NRKW Chinook index (lower panel)

The PVA model used the updated binomial logistic regression coefficients, with the three Chinook index sources (Table 10). The stock used for fecundity effects were either the same population-relevant salmon stocks used for mortality (Coastwide, NRKW or SRKW), the same index as that for mortality, or the WCVI index specifically for fecundity effects. The NRKW-

relevant stock (FE+PS+URB) has a lower mean abundance but a higher minimum and maximum value relative to that for the SRKW-relevant stock (WCVI+FL+OC) over the time series (Table 10). This suggests that NRKW could have greater access to Chinook salmon than the SRKW. The fecundity-relevant stock (WCVI) has a low relative mean abundance but a higher range (0.224 – 2.371), suggesting that there is higher variance in this stock relative to the others.

The percentage of adult females breeding (Br) was defined as a logistic function with age structure, using separate parameters for young females (< 31 years old; Br_1) and older females (> 30; Br_2). These coefficients were re-scaled for use in the model scenarios as a reduction to the reference fecundity rate.

$$Br_1 = \frac{100 * EXP(-1.88 + 0.5395 * CHINOOK)}{1 + EXP(-1.88 + 0.5395 * CHINOOK)}$$

$$Br_2 = \frac{100 * EXP(-2.96 + 0.3 * CHINOOK)}{1 + EXP(-2.96 + 0.3 * CHINOOK)}$$

3.3.3 Disturbance - Vessel Noise/Presence

Acoustic disturbance may come from a range of anthropogenic activities but here we focus on the impacts of vessel-associated disturbance on killer whales. In Lacy et al. (2017), it was assumed that the effect on demographic rates of reduced feeding activity is the same as a comparable reduction in prey (i.e., no behavioural compensation by killer whales). Lusseau et al. (2009) observed a 25% reduction in feeding activity when boats were present. Lacy et al. (2017) estimated that vessels are present 85% of the daytime and killer whales are foraging in the presence of vessels 78% of the time. This represents a 16.6% reduction in Chinook salmon availability in the model (25% x 85% x 78%).

A Population Consequences of Disturbance (PCoD) model (National Research Council 2005; Tollit et al. 2017) has been used to quantify the chronic and acute impacts of noise disturbance on killer whales. A noise exposure model combined with the PCoD model has been used to estimate the lost foraging time for SRKW as a combination of behavioural response and masking (Tollit et al. 2017). For the Salish Sea's busy traffic areas, where SRKW spend 23-33% of their time, individual noise disturbance events can combine to have potentially substantial impacts. The 2017 model predicted that in the Salish Sea, SRKW foraging time was decreased 20-23% of each whale day. Reductions in foraging time are based on the assumption that the impact of acoustic disturbance is the same in NRKW and SRKW. There is emerging evidence that the acoustic environment is naturally noisier in the NRKW range than in the SRKW, due to storm action, waves and other factors. This may differently affect the response of the two populations to anthropogenic acoustic disturbance (S. Vagle, DFO, Institute of Ocean Sciences, pers. comm.).

A time series of vessel activity for the study region that is comparable to the data available for killer whale population dynamics and Chinook salmon was not available. In order to estimate the relative presence of vessels for each population, we gathered data on the magnitude of vessel presence (commercial, recreational and whale watching vessels) in the range of SRKW and NRKW. This rough comparison will be used to set the vessel presence parameter for noise disturbance, which was estimated to be 85% for SRKW by Lacy et al (2017).

Vessel Transits

Marine Communications and Traffic Services (MCTS)-Western provided summaries of commercial vessel movements in British Columbia, within the ranges of the NRKW and SRKW populations. Vessel movements are recorded for specific vessel types by MCTS at each call-in station and include all vessels using Automatic Identification System (AIS-A) that pass a call-in station (a single pass is one vessel movement). Vessel types included in the dataset include barges, tugs, tankers, bulk carriers, container ships, cruise ships, fishing vessels, military vessels, scientific research vessels, and large yachts. For the model comparison, all vessel types were treated equally. The data collated were for Jan 1, 2017 to Dec 31, 2017 but the vessel traffic is fairly consistent across recent years with low inter-annual variability (B. Crooks, MCTS Western Region, pers. comm.). There are seasonal differences in the vessel traffic: there was 35% more vessel traffic in summer (April – September) than winter (October – March).

Vessel movements were summed for the MCTS call in points that fall within the range of each population (Figure 12). The boundary between the NRKW and SRKW was delineated by call in point seven on the west coast of Vancouver Island, and call in point 25 on the east. There are many more vessel movements (155,556) in the SRKW range compared to the NRKW range (31,065) (see map Figure 12). Based on these data, we estimate that SRKW are exposed to five times as many commercial vessel transits than NRKW.

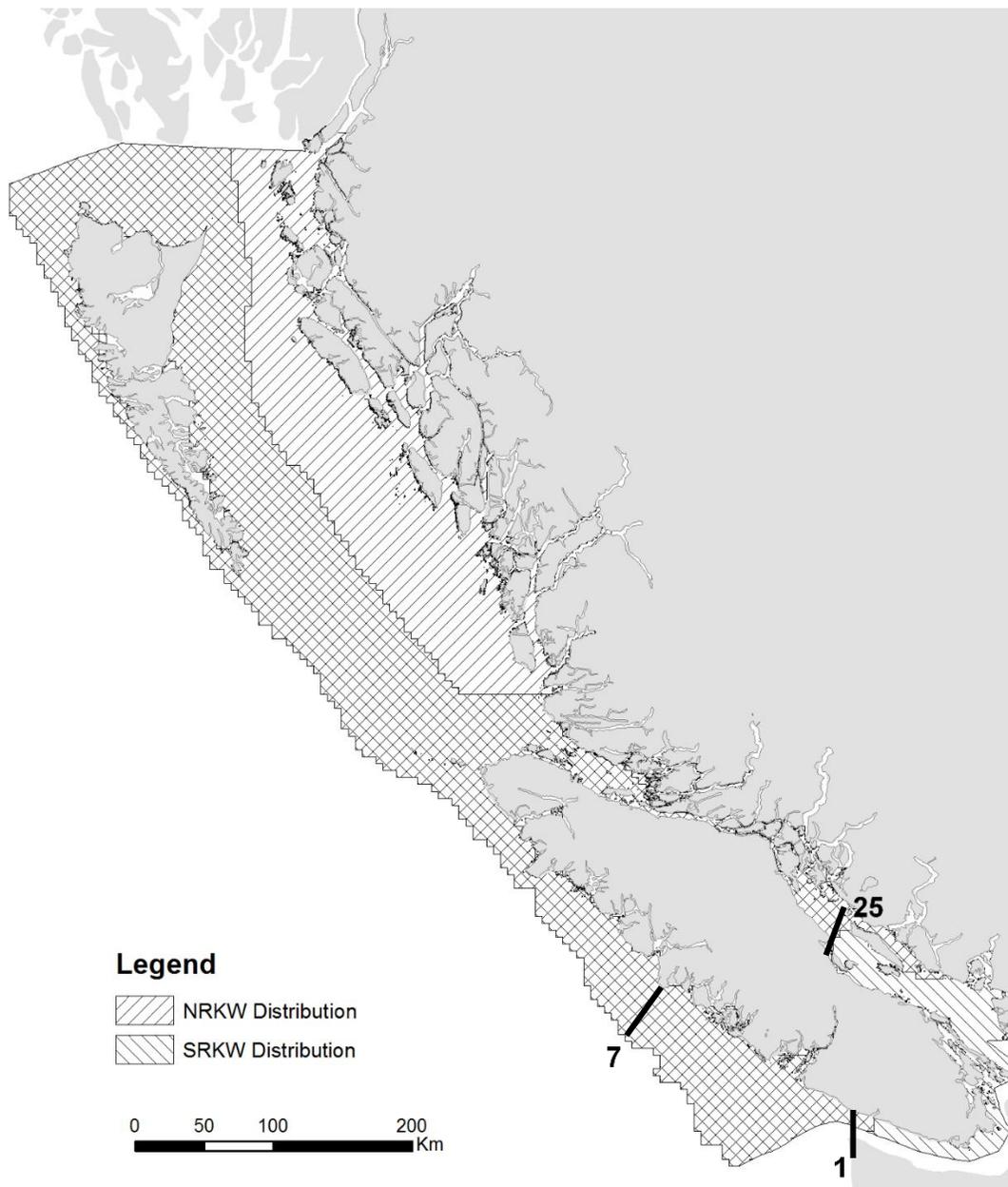


Figure 12 - Canadian range distribution of NRKW and SRKW populations (SARA, 2015) and the three relevant numbered MCTS calling-in-points used to summarise commercial vessel movements.

Whale Watching Vessels

Whale watching has increased significantly, from a few boats in the 1970s to an estimated 96 active commercial whale watching vessels operating in 2015 (Osborne 1991; Seely et al. 2017). There are far more whale watching vessels in the SRKW range than the NRKW (Seely et al. 2017).

Details on the location and operations of commercial whale watch operators in BC and Washington State were catalogued in January 2019 (Serra-Sogas 2019, unpublished report¹⁰). The Salish Sea had the largest whale watch fleet (Canadian: 30 operators; US: 26 operators) and the highest number of vessels (Canadian: 77; US: 34). The rest of BC (Johnstone Strait, Queen Charlotte Strait, Central and North Coast and West Coast Vancouver Island) had smaller commercial fleets (31 operators, 61 vessels), suggesting that SRKW are in the presence of whale watching vessels more often than NRKW. Moreover, the SRKW population is more exposed to vessels than the NRKW population, as whale watching boats are present with the SRKW population (or part of the population) from about 09:00 to sunset, as late as 21:00 h in summer (Lusseau et al. 2009), in addition to many other types of boats that are also present engaging in opportunistic whale watching.

Recreational Vessels

There are 55% more recreational marinas operating in the SRKW range (193 marinas) than in the NRKW range (87 marinas) (Figure 13; Clarke Murray et al. 2014).

¹⁰ Serra-Sogas, N. 2019. Quantifying whale watch vessel traffic in British Columbia and Washington State. Unpublished report to DFO. 23pp.

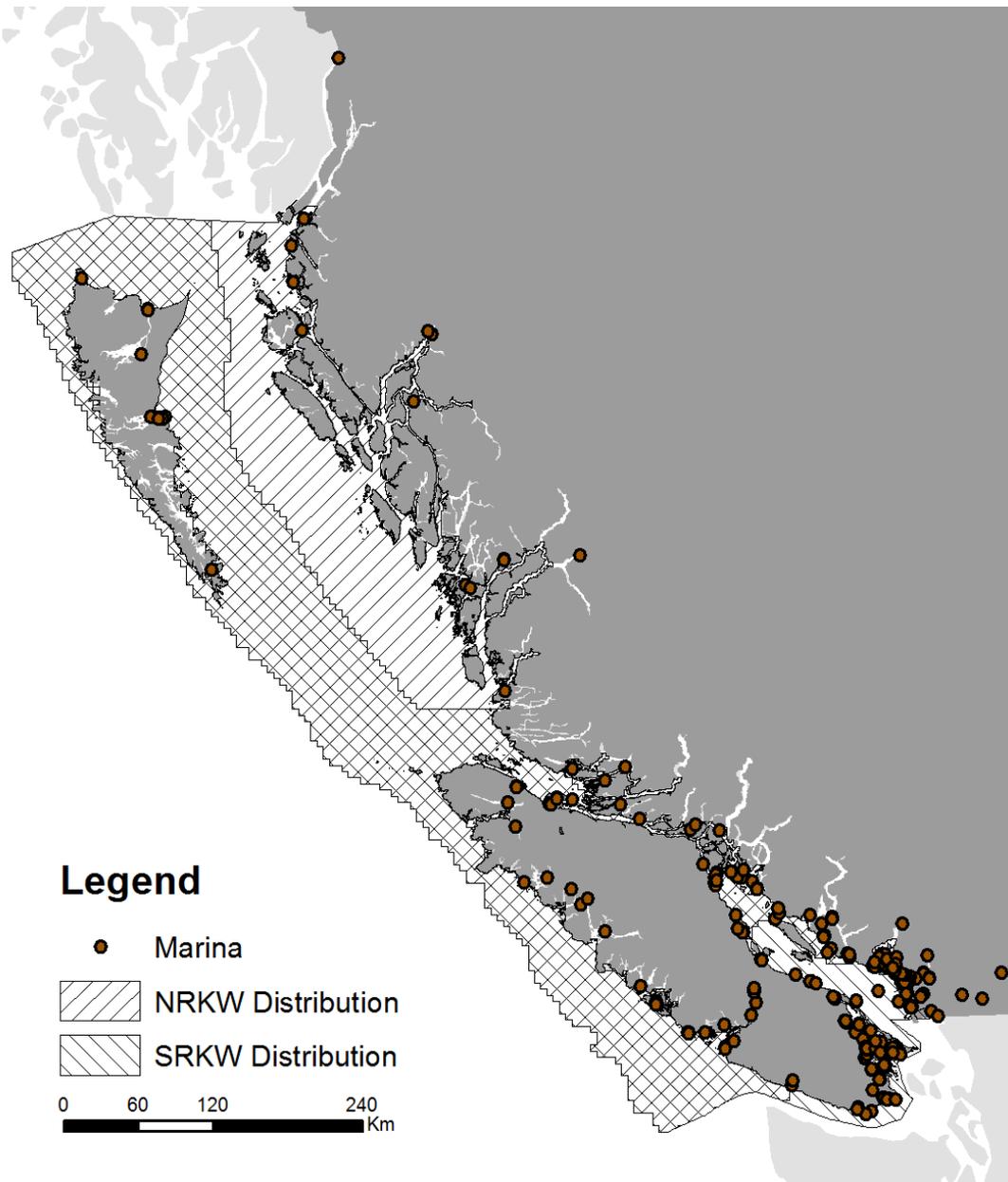


Figure 13 - Distribution of recreational marinas in British Columbia overlaid on the Canadian range distribution of NRKW and SRKW populations

Vessel Noise/Presence Model Parameters

For SRKW, the noise impact was modeled as it was in Lacy et al (2017). Noise was modeled as a reduction in feeding efficiency, and was linked with the variation in Chinook salmon abundance (see Prey availability section). The noise parameter varied between 0.85 (no effect), 1.0 (current estimate of vessel presence), and 1.25 (increased noise impact) under three scenarios (Table 13). In the model scenarios, the feeding rate was used to reduce the Chinook availability and act as a modifier for the mortality rates across all age classes. In scenario one (“Noise”, Table 13), noise was modelled to reduce feeding rate by 3.5%. In scenario two, (“Noise threshold”, Table 13), the effect of noise was modeled as a threshold effect, where vessel disturbance only affects mortality when Chinook stocks are low. When the Chinook stock

index was greater than the mean (1), there was no effect of noise disturbance but when the index was lower than 1, the noise disturbance effect was applied (3.5% reduction in feeding rate). In scenario three (“Noise threshold-high” Table 13), the effect of noise was also modeled as a threshold effect where the effect of noise increased when Chinook stock index was low. When the Chinook stock index was greater than the mean (1), the baseline effect of noise was applied (25% less feeding activity than without the presence of vessels). When the Chinook stock index was less than or equal to the mean (1), the effect of noise was increased by 25% (Table 13), which at the lowest stock index levels leads to roughly twice the mortality rate.

Based on the vessel data described above, the commercial vessel traffic in the NRKW range is five times less than that for SRKW and distributed over a much larger area, suggesting that the NRKW population spends comparably little time in the presence of vessels. Within critical habitat or key foraging areas, vessel presence may have a higher impact on foraging success of NRKW. For NRKW, noise threat parameters were modeled using the same three noise scenarios as SRKW (noise, noise threshold, and noise threshold-high) with a five-fold reduction in vessel exposure.

Table 13 - Noise parameters and feeding rate used in the noise threat scenarios for SRKW and NRKW, and a scenario specific to NRKW (NRKW-low vessels).

Scenario	Noise parameter		Feeding rate	
	High Chinook (>1)	Low Chinook (<1)	High Chinook (>1)	Low Chinook (<1)
Noise	1.0	1.0	0.965	0.965
Noise threshold	0.85	1.0	1.0	0.965
Noise threshold - high	1.0	1.25	0.965	0.907
NRKW – low vessels	0.85	0.85	1.0	1.0

3.3.4 Disturbance - Vessel Strike

Current knowledge of Resident Killer Whale mortalities from vessel strikes does not indicate that there is a difference in the risk of a vessel strike threat between the two KW populations, even with different vessel densities. Limited data on cause of mortality suggests that SRKW have a slightly higher risk of strike than NRKW, 9.5% and 7.1% of cases respectively (Ford et al. 2000; Baird 2002; Williams and O’Hara 2010). Changes in the frequency of vessel transits and the characteristics of ships (quieter ships may increase strike risk) could affect this probability in the future. The vessel strike threat was modeled as a 10% probability of a fatal vessel strike each year across the entire population resulting in an animal being removed randomly from the model adult population once every ten years. The probability was shared equally between males and females of the population.

3.3.5 PCB Contamination

The impact of contaminants on killer whale vital rates could only include PCBs. A PCB accumulation/deposition model has been developed to link PCB levels to calf mortality in cetaceans (Hall et al. 2006; 2018). This PCB model has been used in cumulative effects

assessment for SRKW (Lacy et al. 2017) and in estimating risk to global killer whale populations (Desforges et al 2018). The logistic regression model used in these studies (Figure 14; Hall et al. 2018) predicts survival based on maternal PCB level. The levels of PCBs in killer whales have been recorded from blubber samples in both populations (Ross et al. 2000), and vary greatly between sexes and through time. There were relatively few samples prior to 2000 and the high variability makes it difficult to parameterise the scenarios. The results from blubber samples obtained since 2000 have been compiled recently and analysed to update the estimated mean PCB loads, including samples from both NRKW and SRKW obtained in Canada and the US (Table 13; Pearce and Gobas 2018 unpubl.⁸; Guy 2018 unpubl.⁵; Appendix II). The PCB model simulated the accumulation of PCBs in individuals over time, based on a set accumulation rate. Females depurate (offload) an estimated 77% of PCB load to each calf during calving and nursing (Hall et al. 2006; 2018). The estimated accumulation rate can vary based on sources of PCBs in the environment and prey items (Pearce and Gobas 2018 unpubl.⁸; Desforges et al. 2018). Based on a global review and modelling of killer whale populations, Desforges et al. (2018) suggest a higher accumulation rate (6 mg/kg) for SRKW than for NRKW (1 mg/kg). Model scenarios were run using both the estimated PCB levels from Ross et al 2000 (1993-1996) and the grand mean for the entire time series (Guy 2018, unpubl.⁵; Pearce and Gobas 2018 unpubl.⁸) (Table 14), with varying accumulation rates (1, 2, and 6 mg per year). The modeled PCB concentrations were then compared to the sampled PCB levels.

The effect of PCBs on calf mortality could be in addition to the effect of prey availability. We tested an additive model which applies both mortality and fecundity impacts from prey availability and an additional impact on calf mortality from PCB accumulation. It has been hypothesised that PCBs may only be metabolised from the blubber and exhibit an impact on calf mortality when prey levels are low (Robinson et al. 2018). We next applied a threshold interaction impact so that calf mortality would only be applied when the Chinook index was less than 1 (long term mean). The modeled PCB concentrations in scenarios with various starting concentrations and accumulation rates were compared to measured killer whale PCB concentrations.

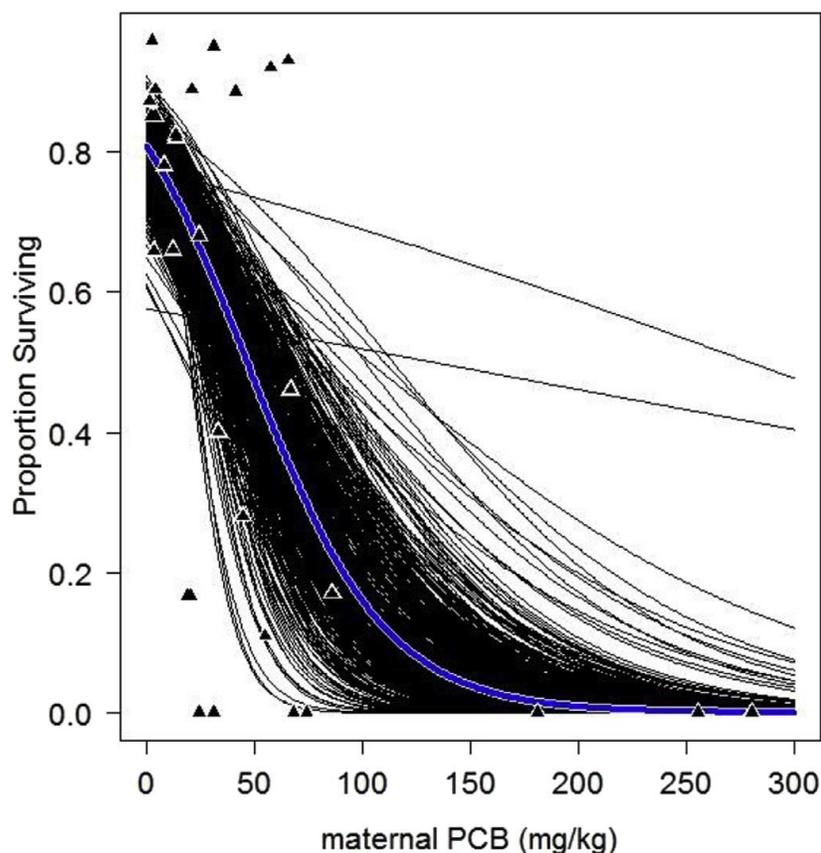


Figure 14 - (after Fig.2 in Hall 2018) Logistic regression model predicting probability of calf survival in relation to maternal blubber PCB concentration using a subset of studies. The triangles represent the data points from the six published studies and black lines show 500 resampled regression models and the blue line shows the best fit.

Table 14 – Mean PCB concentration values for each Resident population by sex and the source of the values. Full dataset used to calculate means in Appendix II.

Population	Female PCBs (mg kg ⁻¹ lw)	Male PCBs (mg kg ⁻¹ lw)	Source
NRKW	9.3 ± 2.8 (n=15)	37.4 ± 6.1 (n=13)	Ross et al. 2000
NRKW	4.9 ± 2.9 (n=42)	10.1 ± 2.4 (n=44)	Ross et al. 2000; 2013 unpubl. ⁹
SRKW	55.4 ± 19.3 (n=2)	146.3 ± 32.7 (n =4)	Ross et al. 2000
SRKW	17.5 ± 3.2 (n=16)	40.7 ± 2.9 (n=24)	Ross et al. 2000, 2013 unpubl. ⁹ ; Krahn et al. 2007, 2009; Guy 2018 unpubl. ⁵

3.3.6 Cumulative Effects

A model including all threats (cumulative effects) was constructed using the individual threat models and the interactions between them. A representative model including all threats allows the modelling and simulation of management and mitigation actions and the effects on future

population growth. Table 15 lists the details of the parameterisation of the individual threats and their interactions in the cumulative effects model.

The observed population growth is the result of the actual conditions in the environment. To test the predictive power of the cumulative model, the Chinook values for 2000-2017 were input into the model. The cumulative effects model was then projected 100 years into the future to examine the long-term forecast for the two populations under the current levels of cumulative effects.

Table 15 - Summary of the threat parameters used in the cumulative effects model for each Resident population.

Threat model	SRKW	NRKW
Prey Availability	Mortality related to Chinook index relevant to SRKW: Chinook WCVI + FL + OC (mean OA: 373,151, index min 0.467, max 1.611); fecundity impact related to SRKW index	Mortality related to Chinook index relevant to NRKW: Chinook FE + PS + URB (mean OA: 261,052, index min 0.584, max 1.818); fecundity impact related to NRKW index
Vessel Strikes	One per ten years	One per ten years
Prey-Noise High Threshold	Feeding is expected to be reduced by 16.6% (85% × 78% × 25%) due to disturbance by boats. When the Chinook stock index was less than or equal to the mean (1), the effect of noise was increased, which at the lowest stock index levels leads to roughly twice the mortality rate.	Feeding reduced by 3% (17% × 78% × 25%) because the number of vessels was five times less in NRKW range
Contaminants (PCB)	Calf survival based on maternal PCB concentration (Hall et al 2018) Females 17.46 mg/kg; Males 40.74 mg/kg; Accumulation rate 2 mg/kg/y; Depuration rate 0.77; When prey availability was low (less than mean index), the PCB impact was applied	Calf survival based on maternal PCB concentration (Hall et al 2018) Females 4.97 mg/kg; Males 10.09 mg/kg; Accumulation rate 1 mg/kg/y; Depuration rate 0.77; When prey availability was low (less than mean index), the PCB impact was applied

4 PVA RESULTS

4.1 REFERENCE MODEL

The reference scenario using the vital rates from the SARKW population demonstrates dramatic growth over time (Figure 15), reaching up to 178 (SD = 58.5) and 400 (SD = 88.5) individuals by 2017 for SRKW and NRKW, respectively (Figure 15).

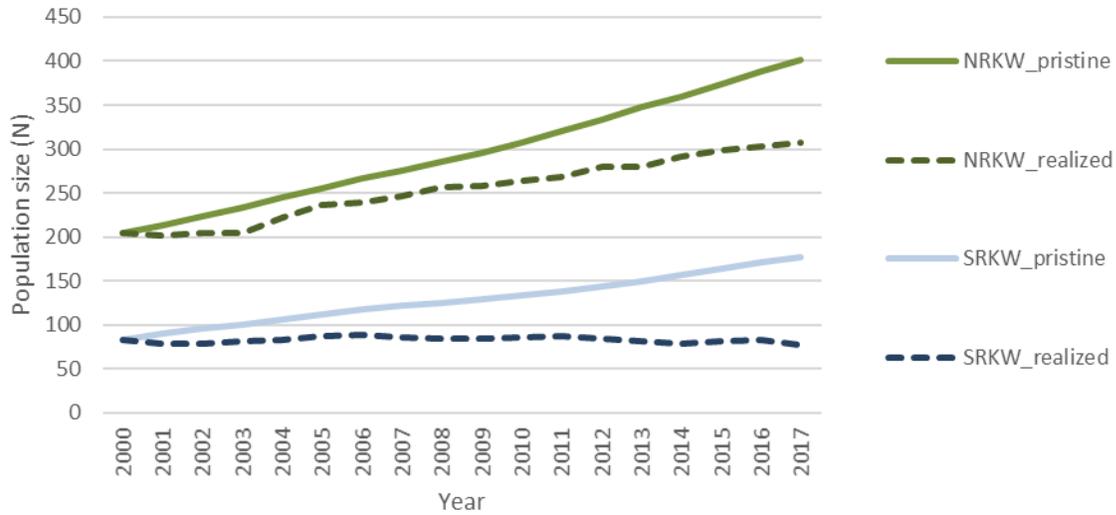


Figure 15 - Mean reference model (“pristine”) simulations for SRKW (solid blue lines) and NRKW (solid green lines), with the observed (realised) population growth (dashed lines).

4.2 AQUARIUM REMOVALS / LIVE CAPTURE FISHERY

The modeled population with removed animals (“removals” scenario) used the observed SRKW mortality and fecundity rates (not the reference rates). The removals scenario shows a similar population growth trajectory ($r = 0.004 \pm 0.039$) to the observed SRKW population ($r = -0.002$, 1979-2017), but remains at a higher population size (~ 140 animals ± 26 SD) (Figure 16).

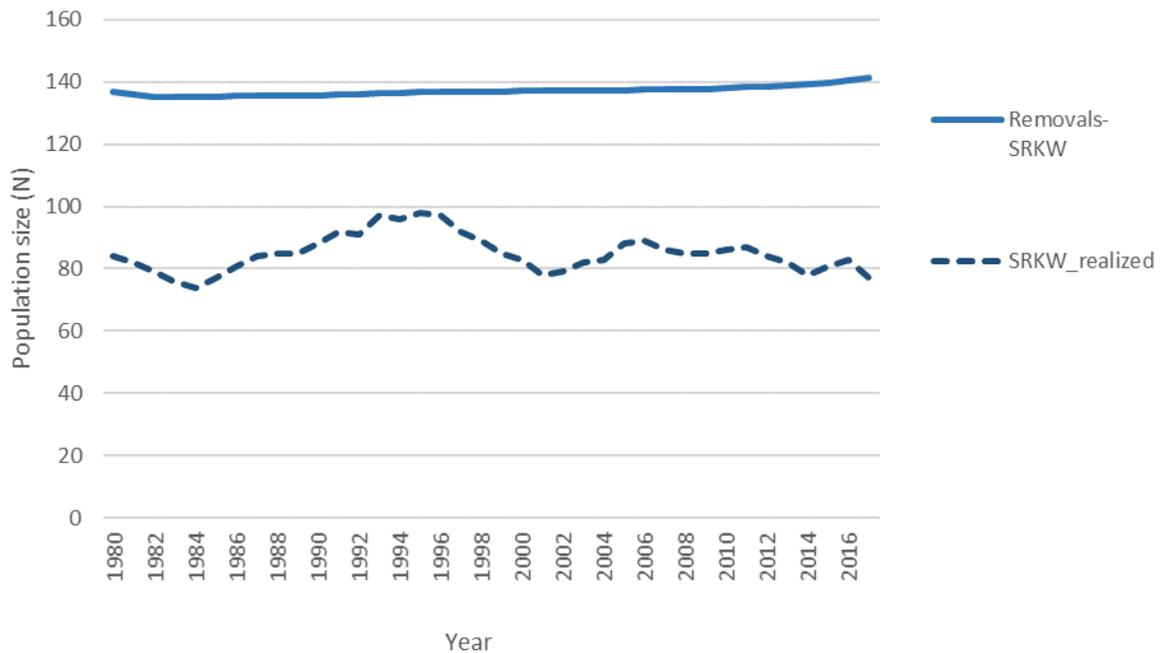


Figure 16 – Mean modeled simulation (solid blue line) and realised (dashed line) population size for SRKW, with removed animals added back into the population in 1980.

4.3 PREY ABUNDANCE

The effect of prey abundance on population size was tested with a number of scenarios to examine the effect of the Chinook index chosen, the way the index value is assigned each year, and the application of mortality and fecundity impacts (Table 16). Prey abundance effects on mortality alone does not explain the realised population growth (Figure 17). The closest model for NRKW includes the effects of prey availability on both mortality and fecundity and approaches the observed population trend, especially in the first 12 years of the simulation. For SRKW, the model scenario that incorporates impacts on both mortality and fecundity does not match the observed population growth, as it predicts slow population growth, rather than stability or decline. Therefore, salmon availability alone does not explain the population dynamics in SRKW and indicates that other threats may be affecting this population.

The distribution of Chinook stock size is not likely to be random and also not likely to be uniform across all possible values. We therefore also explored scenarios with the Chinook index being chosen from a normal distribution, with the median and standard deviation defined from the Chinook ocean abundance time series (Figure 18). Table 16 shows the results of these scenario simulations.

Table 16 - PVA scenarios for the impact of prey abundance on both SRKW and NRKW. Scenario runs used either uniform or normal distribution to assign the Chinook abundance in each year, and impacts were on mortality, or on both mortality and fecundity. The stochastic-r, standard deviation (SD(r)), probability of extinction (PE), the number of live animals (N-extant) and the standard deviation for the number of live animals (SD(Next)) are presented for each model scenario.

Distribution	Mortality Index	Fecundity Index	stoch-r	SD(r)	PE	N-extant	SD(Next)
Uniform	Coastwide		0.0396	0.0900	0.0001	429.09	92.07
Uniform	Coastwide	Coastwide	0.0147	0.0889	0.0002	290.56	88.89
Uniform	Coastwide		0.0396	0.0830	0	427.70	88.89
Uniform	Coastwide	WCVI	0.0243	0.0876	0	347.32	101.4
Uniform	NRKW/SRKWruns		0.0414	0.0833	0	435.31	86.69
Uniform	NRKW/SRKWruns	NRKW-runs	0.0224	0.0845	0	334.82	97.13
Uniform	NRKW/SRKWruns		0.0414	0.0823	0	435.90	87.01
Uniform	NRKW/SRKWruns	WCVI	0.0264	0.0869	0	359.36	101.05
Normal	Coastwide		0.0393	0.0829	0	425.94	89.72
Normal	Coastwide	Coastwide	0.0138	0.0833	0	284.61	88.14
Normal	Coastwide		0.0394	0.0831	0	427.03	90.26
Normal	Coastwide	WCVI	0.0235	0.0854	0	341.56	97.99
Normal	NRKW/SRKWruns	NRKW-runs	0.0416	0.0824	0	436.46	86.03
Normal	NRKW/SRKWruns	NRKW-runs	0.0218	0.0845	0	332.09	97.72
Normal	NRKW/SRKWruns		0.0414	0.0825	0	436.08	85.71
Normal	NRKW/SRKWruns	WCVI	0.0253	0.0858	0	353.10	100.00

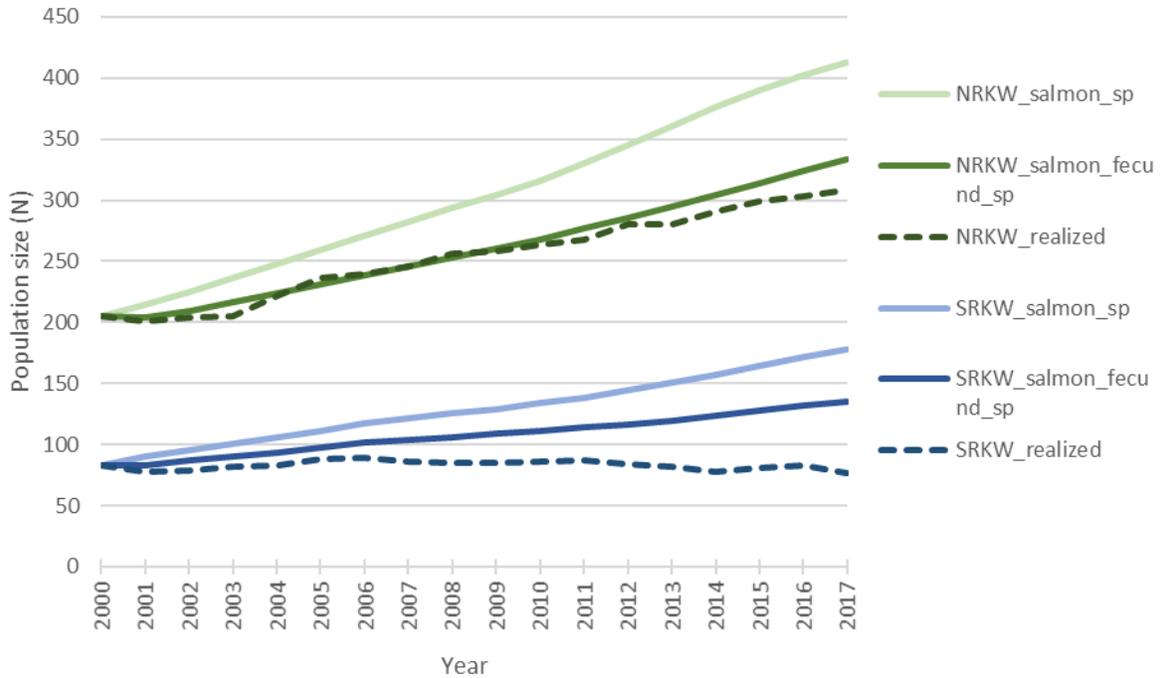


Figure 17 - Scenarios with mean modelled simulations (solid lines) and observed ("realized" - dashed black lines) population size over time for models that include the effects of prey availability on mortality and fecundity in NRKW (green) and SRKW (blue), using a uniform distribution and specific Chinook indices for each population with impacts on mortality only (salmon_sp) and with an additional fecundity impact (using the WCVI stock index, salmon_fecund_sp).

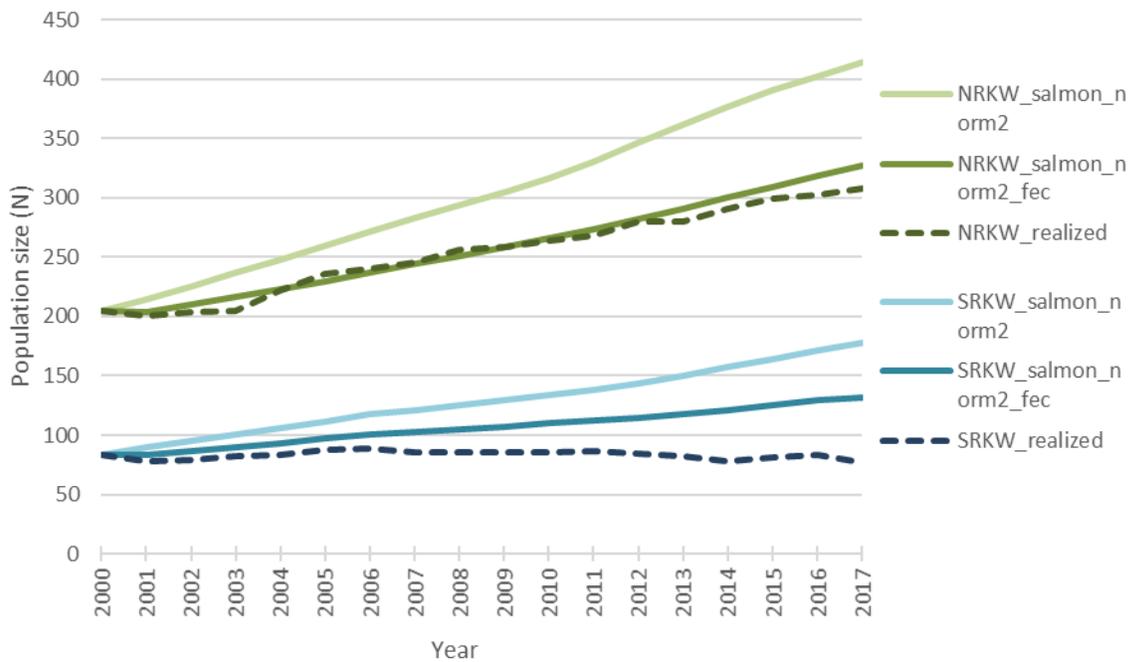


Figure 18 – Mean scenario simulations with Chinook index chosen from a normal distribution around the mean and specific Chinook indices for each population. Impacts on mortality only (salmon_norm2) and additional impacts on fecundity using the WCVI index (salmon_norm2_fec).

4.4 DISTURBANCE – VESSEL NOISE/PRESENCE

The effects of vessel noise/presence (mediated through prey abundance as a reduction in abundance of prey available) did not match the observed population dynamics (Figure 19; Table 17). The threshold scenarios were similar in impact to noise alone for SRKW, predicting higher KW abundances than the direct effect of noise scenario. The high noise threshold depressed the population more strongly in NRKW than SRKW (Figure 19). Even the high noise threshold scenario did not approach the observed population dynamics. These results suggest that vessel noise/presence disturbance alone does not control the dynamics of these populations.

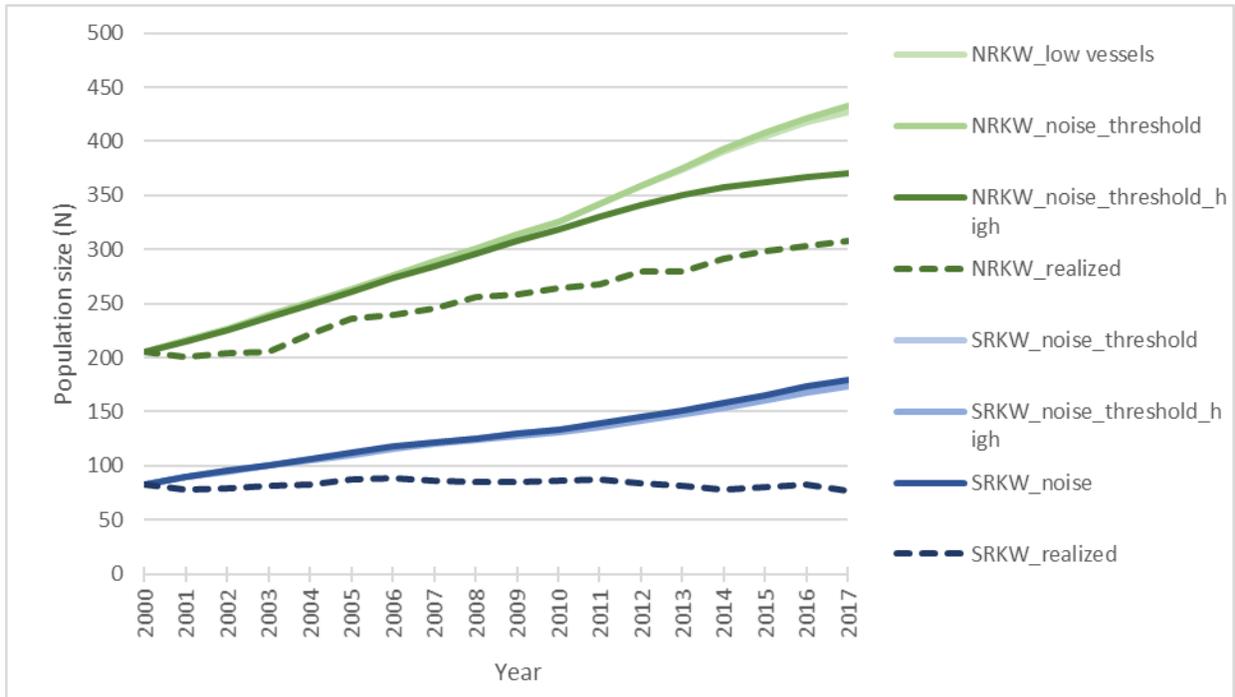


Figure 19 – Mean modeled (solid lines) and realized (dashed lines) population size for NRKW and SRKW, with scenarios of the impacts of noise set to 25% feeding rate reduction (Noise scenario), with a threshold effect (Noise threshold) and with a higher impact when prey is low (Noise threshold high), and reduced vessel presence for NRKW (low vessels).

Table 17 - PVA simulation results for each noise threat scenario, including the population growth rate (stoch-r), standard deviation of r (SDr), probability of extinction (PE), and the estimated number of living animals (N-extant) and the standard deviation (SD Next).

Model scenario	stoch-r	SD(r)	PE	N-extant	SD(Next)
SRKW_noise	0.0406	0.084	0	198.68	66.51
NRKW_low vessels	0.0444	0.075	0.0001	445.14	86.52
SRKW_noise_threshold	0.0409	0.0835	0	197.06	56.82
NRKW_noise_threshold	0.045	0.0745	0	454.83	74.95
SRKW_noise_threshold_high	0.0393	0.086	0.0001	191.52	57.05
NRKW_noise_threshold_high	0.043	0.079	0	375.73	51.99

4.5 DISTURBANCE - VESSEL STRIKE

The modeled populations affected by vessel strikes do not match the realised population growth (Figure 20; Table 18). These results suggest that physical disturbance alone does not control the dynamics of these populations.

Table 18 - PVA simulation results for each strike threat scenario, including the population growth rate (stoch-r), standard deviation of r (SDr), probability of extinction (PE), and the estimated number of living animals (N-extant) and the standard deviation (SD Next).

Model scenario	stoch-r	SD(r)	PE	N-extant	SD(Next)
SRKW_strike	0.0396	0.0849	0	192.51	58.01
NRKW_strike	0.0392	0.0799	0	367.32	54.64

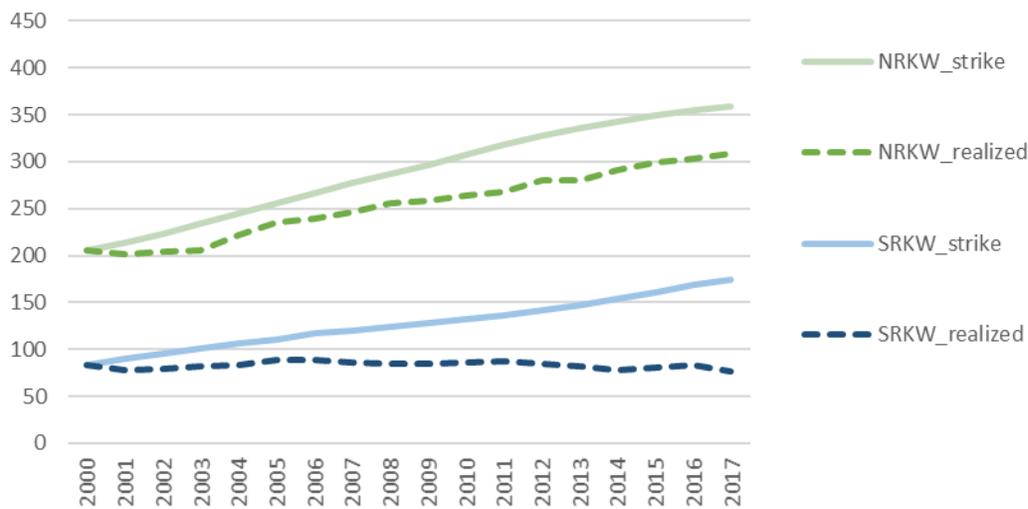


Figure 20 – Mean modeled scenarios (solid lines) and realised population growth (dashed lines) for strike threat for the SRKW (blue) and NRKW (green) populations.

4.6 PCB CONTAMINATION

The population model simulations generate a range of mean PCBs levels in adults across different initial PCB levels and accumulation rates (Table 19). The model scenarios that most closely approach the range of measured PCB levels in recent samples are those with initial PCB levels set to the grand means (Table 19), with accumulation rates slightly higher in SRKW than NRKW (2 mg yr⁻¹ and 1 mg yr⁻¹, respectively). The impact of PCBs alone does not match the realised population growth rate for either population (Figure 21; Table 19).

The results of the additive prey and PCB interaction scenario is similar to that of the prey-PCB threshold interaction impact scenario (Figure 22). The additive model applies both mortality and fecundity impacts from prey availability and an additional impact on calf mortality from PCB accumulation. In the threshold prey-pcb scenario, PCB-calf mortality was only applied when the Chinook index was less than 1 (long term mean). Both interaction scenarios resulted in

population dynamics that closely resemble the realised for NRKW and are close for SRKW (Table 20). This suggests that the effects of prey availability on mortality and fecundity swamps the effect of PCB impact alone.

Table 19 - PCB levels from blubber samples and results from model scenario simulations using varied starting PCB concentrations and accumulation rates.

Scenario	Sampled PCB Levels		Accumulation rate	Modeled PCB levels (year 2017)	
	PCB fem	PCB male		Female Mean (\pm SD)	Male Mean (\pm SD)
SRKW_pcb	55.40	146.30	2	61.00 (\pm 6.3)	112.70 (\pm 7.50)
SRKW_pcb_2mg	17.46	40.74	2	48.82 (\pm 4.5)	80.04 (\pm 3.16)
SRKW_pcb_3	17.46	40.74	6	142.77 (\pm 14.0)	239.15 (\pm 7.40)
SRKW_pcb_1mg	17.46	40.74	1	28.19 (\pm 2.8)	48.88 (\pm 2.35)
NRKW_pcb_specific	9.30	37.40	2	48.90 (\pm 4.2)	86.00 (\pm 2.60)
NRKW_pcb_2mg	4.97	10.09	2	47.75 (\pm 4.1)	79.66 (\pm 1.75)
NRKW_pcb_3	4.97	10.09	1	24.65 (\pm 2.2)	41.76 (\pm 1.00)
NRKW_pcb_specific_1mg	9.30	37.40	1	25.80 (\pm 2.2)	48.07 (\pm 2.01)

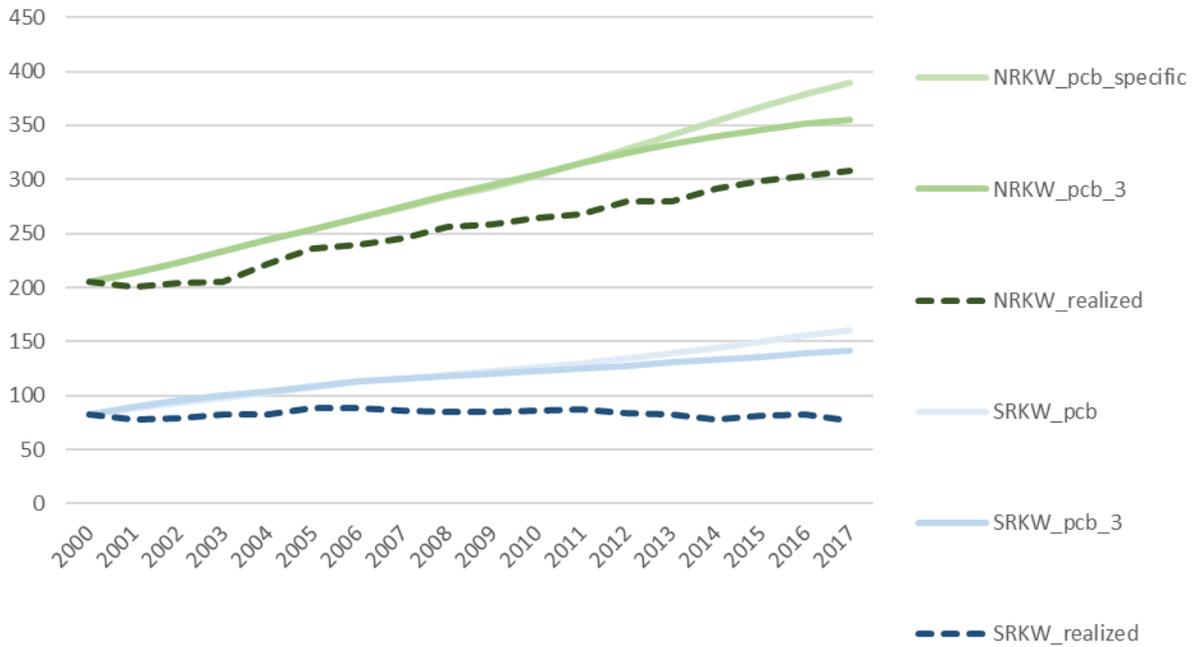


Figure 21 – Mean model simulations of PCB impacts on NRKW (green lines) and SRKW (blue lines), with realised population size (dashed lines) and PCB levels set to levels from Ross et al. 2000 (PCB specific) or with updated values collated from Ross et al. 2000; 2013 for NRKW and Ross et al. 2000; 2013; Krahn et al. 2007; 2009; Guy 2018 unpubl.⁵ for SRKW (pcb 3).

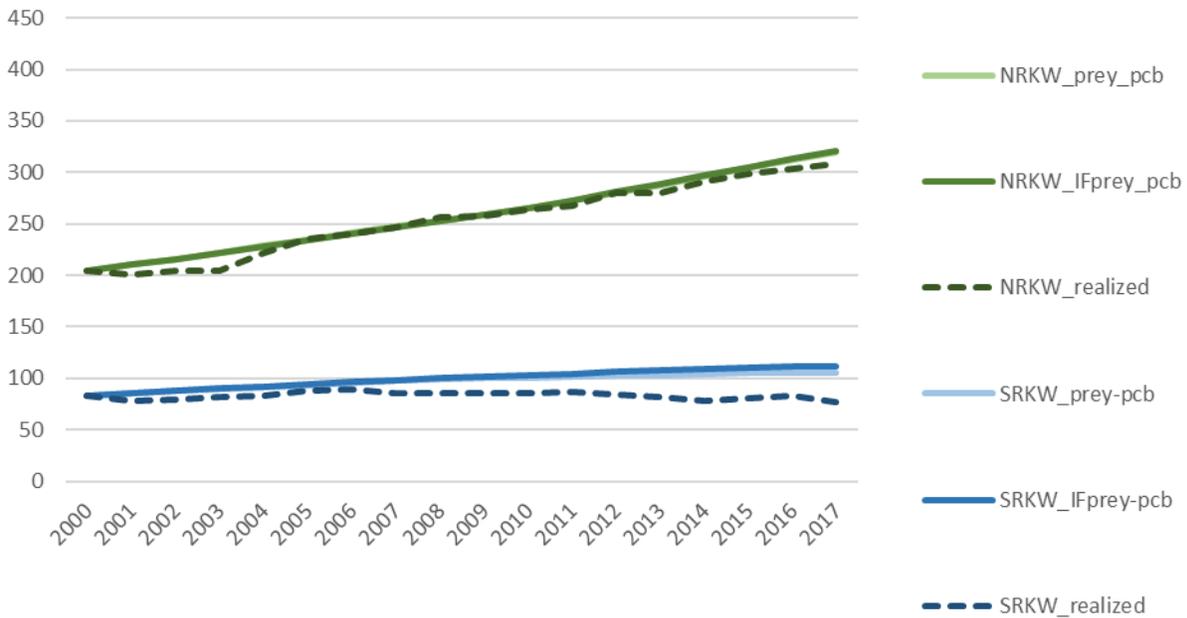


Figure 21 – Mean model simulations of the interaction between prey and PCB threats; either as an additive model (prey abundance mortality and fecundity effects) with the additional effect of PCB effects (prey-pcb), or as an interaction between prey and PCBs, where the calf mortality impact is applied only in years where salmon index is less than 1 (IFprey_pcb).

Table 20 - PVA simulation results for each PCB threat scenario, including the population growth rate (stoch-r), standard deviation of r (SDr), probability of extinction (PE), and the estimated number of living animals (N-extant) and the standard deviation (SD Next).

Model scenario	stoch-r	SD(r)	PE	N-extant	SD(Next)
SRKW_pcb	0.034	0.084	0.000	173.3	53.1
SRKW_pcb_2mg	0.038	0.087	0.000	167.4	49.2
SRKW_pcb_3	0.025	0.083	0.000	145.7	46.3
SRKW_pcb_1mg	0.038	0.086	0.000	188.3	57.7
SRKW_preypcb	0.008	0.080	0.000	103.9	34.1
SRKW_IFpreypcb	0.012	0.080	0.000	112.0	35.8
NRKW_pcb_specific	0.037	0.079	0.000	413.9	91.3
NRKW_pcb_2mg	0.037	0.080	0.000	361.6	58.4
NRKW_pcb_3	0.038	0.081	0.000	364.2	56.7
NRKW_pcb_specific_1mg	0.038	0.080	0.000	420.2	88.9
NRKW_preypcb	0.023	0.081	0.000	341.0	96.4
NRKW_IFpreypcb	0.024	0.080	0.000	342.5	96.9

4.7 CUMULATIVE EFFECTS

The cumulative effects model with all threats represented (prey abundance, PCBs, vessel noise/presence and vessel strikes) is closer to the observed population size than any of the single threat models alone (Figure 23). The cumulative model approaches the realised population growth for both populations closely, but particularly closely in the NRKW population (Figure 24; Table 21). The standard deviations for this model run encompass the observed population growth for NRKW. The mean model NRKW population size in 2017 was 309 (\pm 76 SD), the recorded NRKW population in 2017 was 308. The average model SRKW population size in 2017 was 134 (\pm 41), the recorded SRKW population in 2017 was 77.

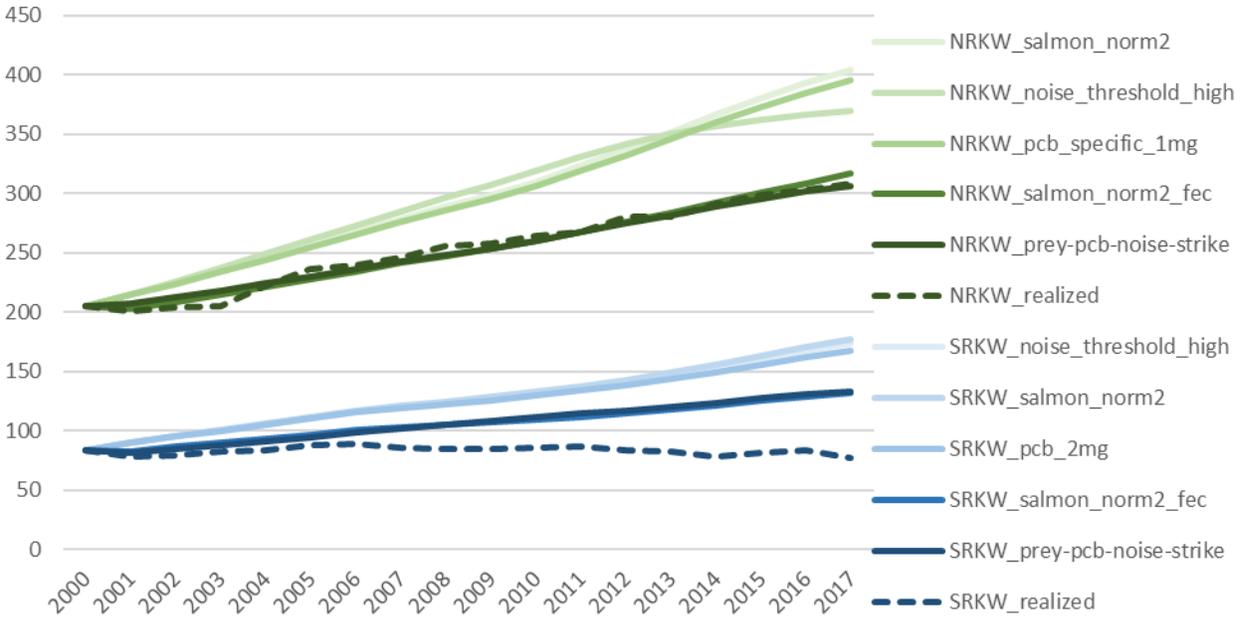


Figure 22 - Mean model simulations of single threat scenarios (prey abundance, vessel noise/presence, PCBs) and the cumulative effects model scenario (prey-pcb-noise-strike) on NRKW (green lines) and SRKW (blue lines), with realised population size (dashed lines). Scenario names norm2: normal distribution salmon index, norm2_fec: adds a fecundity impact.

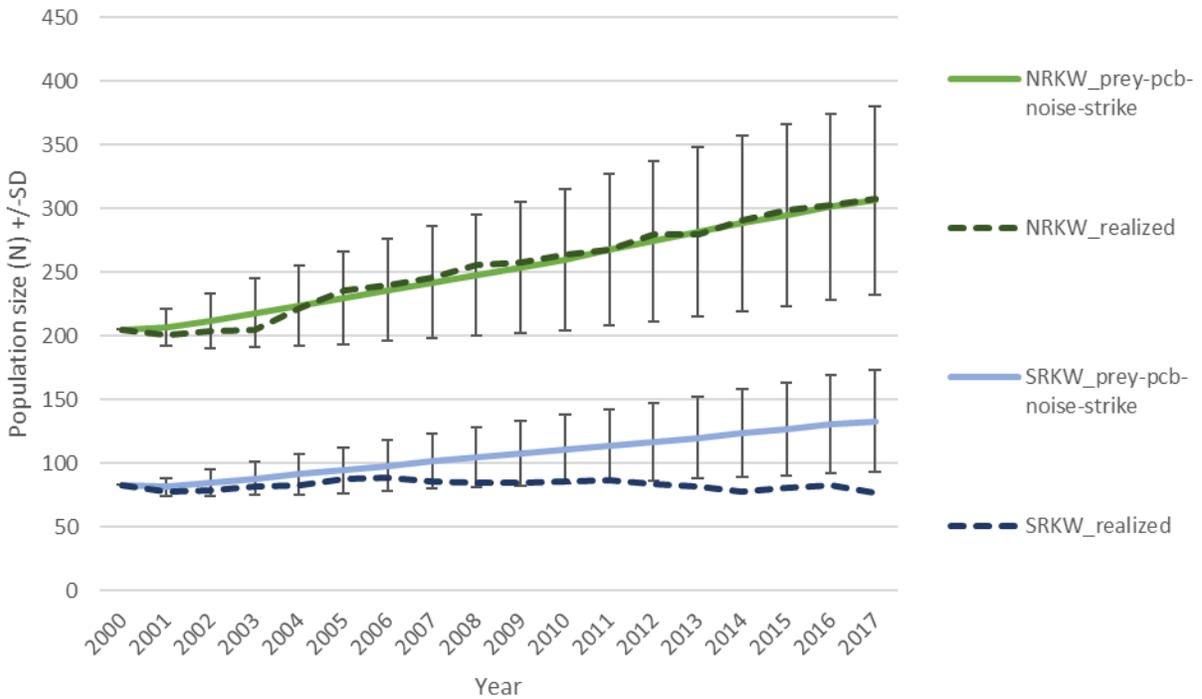


Figure 23 - Mean model simulations of the cumulative effects scenario (prey-pcb-noise-strike) on NRKW (green lines) and SRKW (blue lines), with realised population size (dashed lines). Error bars represent the ± 1 standard deviation.

Table 21 - PVA simulation results for the cumulative threat scenario, including the mean population growth rate (stoch-r), standard deviation of r (SDr), probability of extinction (PE), and the mean estimated number of living animals (N-extant) and the standard deviation (SD Next).

Model scenario	stoch-r	SD(r)	PE	N-extant	SD(N _{ext})
SRKW_pre-y-pcb-noise-strike	0.024	0.083	0.000	134.4	41.4
NRKW_pre-y-pcb-noise-strike	0.023	0.079	0.000	309.5	75.7

4.8 MODEL VERIFICATION AND VALIDATION

4.8.1 Inspection

Using the historical (rather than drawn randomly from a distribution) Chinook index values for 2000-2017, resulted in the cumulative effects model approaching the observed population growth even more closely, especially for SRKW (Figure 25; Table 22). The cumulative effects model explains the realised population growth better than any single threat model alone.

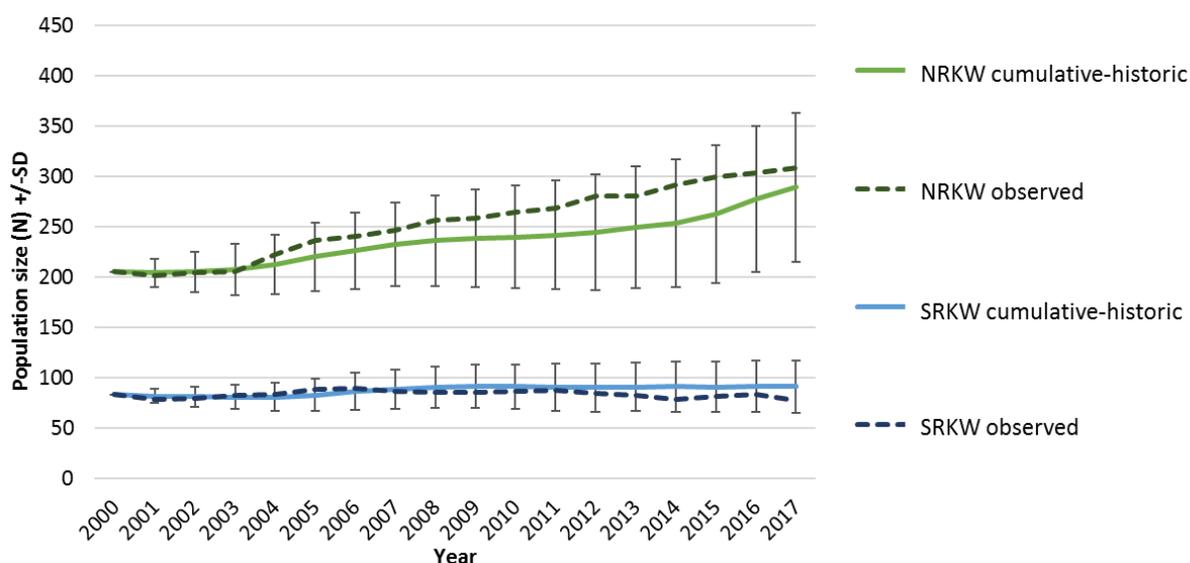


Figure 24 - Mean model simulations of the cumulative threat scenario with the historical Chinook index values (cumulative-historic), and realised population size (dashed line) for NRKW (green) and SRKW (blue). Error bars represent ± 1 standard deviation.

Table 22 - PVA simulation results for the cumulative threat scenario, using historical Chinook salmon index values, including the mean population growth rate (stoch-r), standard deviation of r (SDr), probability of extinction (PE), and the mean estimated number of living animals (N-extant) and the standard deviation (SD N_{ext}).

Model scenario	stoch-r	SD(r)	PE	N-extant	SD(N _{ext})
SRKW cumulative-historic	0.002	0.083	0.000	91.1	29.2
NRKW cumulative-historic	0.019	0.079	0.000	296.6	75.3

4.8.2 Population Structure

To further validate the model, we compared the observed and simulated population structure for the cumulative effects model (prey-pcb-noise-strike). The relative proportions of juveniles and adults were similar, as were the sex ratios for both SRKW and NRKW (Table 23). The NRKW model was extremely close to the observed values in its outputs (Modeled: 102 juveniles and 207 adults; Observed: 104 juveniles and 204 adults). The sex ratios were also similar between the cumulative effects model outputs and the observed, both predicting more females than males. For SRKW the male to female ratio was modeled to be 0.813 and the observed was 0.949, more females than males. For NRKW, the modeled sex ratio was 0.866 and the observed was 0.610. A large proportion of the NRKW population has not been identified to sex, likely because of the number of juveniles and because its members are not surveyed as often as the SRKW.

Table 23 - Observed and modeled demographic structure for the two populations (SRKW and NRKW).

SRKW	Age class	Observed				Modeled		
		Female	Male	Unknown	Total	Female	Male	Total
	0-9	7	13	0	20	22	22	44
	10+	32	24	0	56	52	39	91
	Total	39	37	0	77	74	61	135
NRKW	0-9	4	3	97	104	51	51	102
	10+	102	62	40	204	114	93	207
	Total	105	64	136	308	165	144	309

4.8.3 Projection

Model scenarios can be projected into the future to examine the long-term population growth rate and future of the populations. The projection of the cumulative effects model assumes that the current levels of threats continue into the future, with no changes in threats and no mitigation actions. When Chinook salmon abundance is taken from the long-term mean abundances (1979-2017), the cumulative model (prey-pcb-noise-strike) projects mean positive population growth for both populations, but with uncertainty that includes negative population growth: 1.6% (± 7.9 SD) for NRKW and 1.5% (± 8.1 SD) for SRKW (Figure 26; Table 24). NRKW reaches the arbitrarily-set carrying capacity early in the projections and this affects the projected population growth rate. The probability of extinction (defined in the model as a single sex remaining) for both populations is zero over 100 years. Note that under SARA, extinction is defined as no individuals remaining. In contrast, when the cumulative effects model uses the recent (2008-2017) distribution of Chinook salmon abundance indices the projection is negative population growth for SRKW (-2.5%), and a slightly lowered, but still positive, growth rate for NRKW (Figure 26, Table 24). Under the recent prey scenario, SRKW have a 26.1% probability of extinction and in those simulations where extinction occurred, the mean time to extinction was 86 years (± 11.3 years).

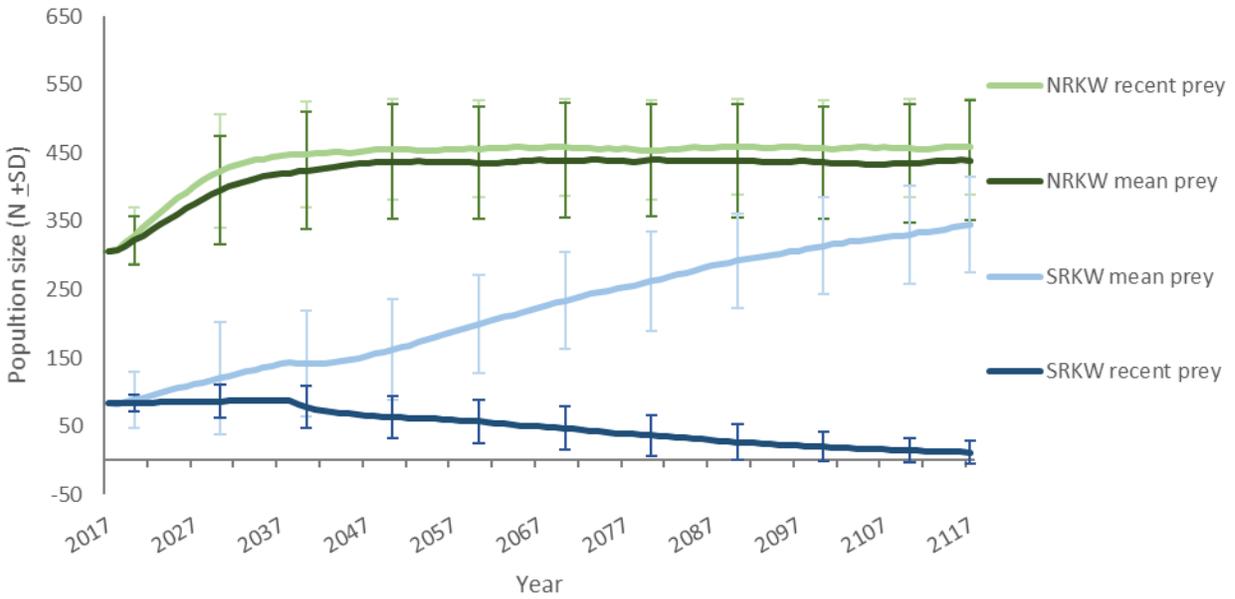


Figure 25 - Mean projection of cumulative effects model 100 years into the future (starting in 2017) for NRKW (green) and SRKW (blue), under mean Chinook index (“mean prey”: 1979-2017) or recent Chinook index (“recent prey”: 2008-2017). Error bars represent +1 standard deviation.

Table 24 - Population growth rate, population size at year 100 (N_{100}) and probability of extinction of cumulative effects model projection 100 years into the future, under two scenarios of Chinook abundance: mean prey (1979-2017) or recent prey (2008-2017).

Summary Statistics	Projection Scenario	
	Mean prey (Chinook index distribution: 1979-2017)	Recent prey (Chinook index distribution: 2008-2017)
Population growth rate	+ 1.5%	- 2.5%
N_{100}	344.7 (\pm 140) years	15.1 (\pm 17) years
Probability of extinction	0	26.1%

4.8.4 Sensitivity

The sensitivity of model parameters was tested to distinguish which parameters have the highest impact on long-term population dynamics. The cumulative effects scenario model projection for SRKW was used in sensitivity testing, which includes all four threats and their interactions as defined in Section 4.7. Single factor sensitivity testing was performed where the parameter of interest was varied across its range (minimum-maximum), with the base values used for all other parameters, using the full individual-based stochastic model. The base threat levels were the original values used in the cumulative effects model. Sensitivity testing for that parameter varied across the entire distribution (minimum-maximum) by set increments (Table 25). For the prey parameter sensitivity testing, the full range of Chinook index values for both stocks was tested (minimum = 0.4, maximum = 1.8). The vessel noise/presence parameter begins at the base level of noise – the value set to 0.85 has no reduction in feeding rate and

increases to a maximum of 1.55, to represent the possibility that the reduction in feeding time could be higher or that the presence spent near vessels was higher than estimated. The PCB value tested included the base initial PCB concentration for females, and included the full range of measured female PCB concentrations. Male PCB concentration was not used in sensitivity testing because the impact pathway occurs via maternal transfer. Strike risk was varied from 5% to as high as 50%. The most sensitive parameter for the long-term projection of the population was prey availability (the value of the Chinook index), followed by vessel noise/presence (Figure 27).

Table 25 - Parameters and values used for testing sensitivity in the SRKW cumulative effects (prey-pcb-noise-strike) scenario projection, including the base value, the range and increment of testing.

	Base	Minimum	Maximum	Increment
Prey (Chinook Index)	1.00	0.40	1.80	0.10
Noise	0.85	0.85	1.55	0.10
PCB-female	17.46	5.00	200.00	25.00
Strike risk	0.10	0.05	0.50	0.05

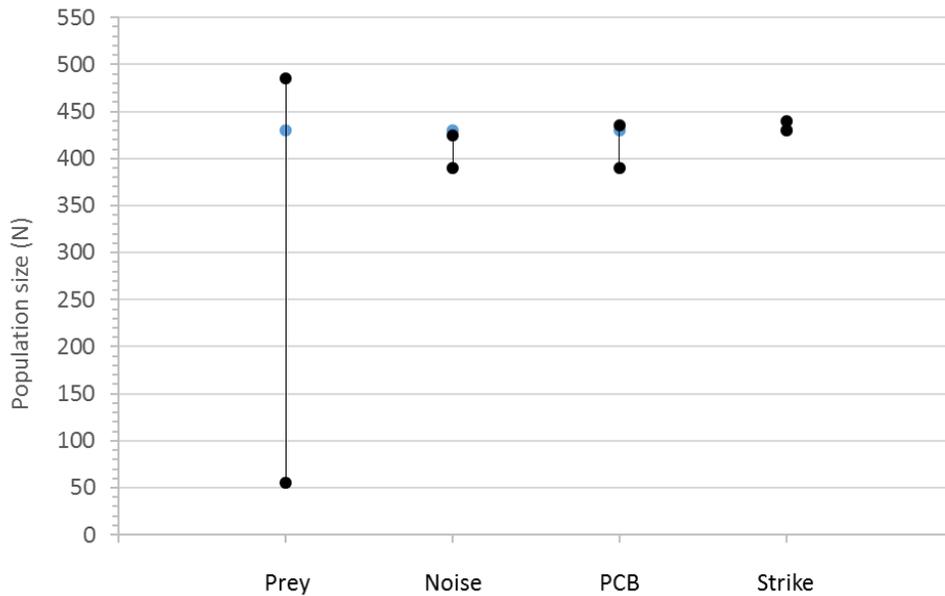


Figure 26 - Sensitivity of SRKW projected population size in the cumulative effects scenario to changes in the threat parameters: prey abundance, vessel noise/presence, PCB concentration and strike risk. Blue circles represent the base value for each threat and the vertical bars represent the range of population size (N) for each change in threat value (Table 25).

5 DISCUSSION

5.1 ASSESSING CUMULATIVE EFFECTS

This is the first cumulative effects assessment that includes a pathways of effects conceptual model and a population viability analysis incorporating the impacts of all the SARA priority threats (prey availability, disturbance and contaminants) on NRKW and SRKW. It is of considerable interest to investigate why these populations with similar life histories and diet exhibit such different population trajectories: the SRKW population is in decline while the NRKW population is increasing (Figure 2). The inclusion of both populations in the same assessment allows for comparisons to be made within the same set of models and parameters (assuming that the mechanisms of impact are similar, while threat exposure levels differ). The systematic assessment of both individual and combined threats in the model scenarios allows examination of which threats (or combination of threats) best explain the observed population growth and in turn may have a greater influence on the population trajectories and demographics of these killer whale populations.

This work relied heavily on the existing body of literature and data and built upon previous work and threat characterisations, including previous assessments of Resident Killer Whale populations (Lacy et al. 2017; Vélez-Espino et al. 2014b; Taylor and Plater 2001). We incorporated new and updated information and data for all threats and included potential interactions between threats. It is anticipated that this species-focused cumulative effects assessment method will be useful as a tool for managers and scientists for killer whales, as well as potentially for other species.

The first phase of the cumulative effects assessment provided an evidence-based pathways of effects conceptual model, creating the foundation for the assessment. Explicitly outlining the background and current state of knowledge of priority threats, interactions and impacts, and in particular the uncertainties and limitations, is valuable for understanding and having confidence that the model outputs represent the system of interest.

The second phase constructed a population model for each population and systematically tested the effects of individual and cumulative effects on population growth by comparing the model outputs to the observed population sizes from 2000-2017. An important change from previous PVAs is that the current model uses the observed vital rates from the relatively unimpacted Southern Alaska Resident Killer Whale population as the reference state, and adds threats to the model as modifiers of these rates. This allowed us to test the validity of the model scenarios in order to identify the model that most closely matches the observed data and assess which of the priority threats most contribute to cumulative effects. This is a different approach than that used by Lacy et al. (2017) for SRKW, where the “baseline” was defined with the mean demographic rates that were observed across recent decades (i.e., including current threats to the population), and then varied threat levels to investigate the impact on population performance. The Lacy et al (2017) baseline tests scenarios of varying threat levels, rather than explaining current trends or patterns in the historical data.

The definition of the model scenarios incorporated updated and/or new inputs for every threat in the assessment:

- Prey abundance - Updated Chinook salmon ocean abundance model data, extending the time series to 2017 from that used in previous studies (2008: Lacy et al. 2017, and 2011: Vélez-Espino et al. 2014b). These data were used in statistical and model selection analyses performed to update the relationships between Chinook salmon indices and mortality and fecundity.

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- Vessel noise/presence – The proportional of loss of foraging time from vessel noise was based on a new study (Tollit et al. 2017), and the incorporation of relative vessel density for both populations estimated by the number of ship transits (MCTS-Western data), and the presence of marinas (Clarke Murray et al. 2014).
 - Vessel strike – Strike risk was estimated based on collation of reports of presumed strikes.
 - Contaminants – Unpublished PCB data summarised from a number of new sources (Guy 2018 unpubl.⁵; Gobas and Ross 2017 unpubl.⁶), extended the number and time range of SRKW samples incorporated in this assessment to 40 samples (collected 1993-2015), compared to the 6 samples (collected 1993 and 1996) included in previous work (Lacy et al. 2017). For the NRKW population, data from 85 PCB samples were available (28 currently available in the published literature).

Despite the updated inputs, the individual threat models did not closely align with the observed population dynamics. However, the cumulative threats model, incorporating all priority threats (Chinook salmon abundance, vessel noise/presence, vessel strike, and PCB contamination, predicted population growth closest to the observed rates for both populations, out of all individual and combined threat models tested. The cumulative effects model scenario results matched the observed data more closely for NRKW than for SRKW (Figure 24).

The cumulative threats model slightly over-estimates growth for the SRKW population. There are a number of factors that could act singly or in concert to explain this. The individual-based simulation PVA includes stochasticity in the model simulations. The Chinook salmon index assigned annually was drawn randomly from a distribution and this index value affects mortality, fecundity, and interactions with other threats. When the historical Chinook index values were used instead of the random values, the SRKW model very closely aligned with the observed RKW data which fell within the model's standard deviations for both populations (Figure 25). This suggests that the cumulative model is a valid representation of the system but the temporal range in question (2000-2017) included a number of years with Chinook availability below the long-term mean (1979-2017). Essentially, the observed data were closely aligned with the subset of model simulations where the randomly drawn Chinook values were lower than average. The development and use of a Chinook salmon model that could better represent the amount of temporal autocorrelation could be used in the model to produce better projections. There is a coastwide Chinook model by the Pacific Salmon Commission that produces four year projections, but it will need modifications to project further into the future.

This assessment highlights the importance of considering threats collectively. Specifically, within the cumulative effects PVA assessment, Chinook salmon abundance and its interactions with vessel noise/presence and PCBs strongly influenced modelled killer whale population dynamics. The cumulative effects PVA model can be used to compare population trajectories into the future for NRKW and SKRW. These projections are best used in a comparative evaluation, rather than absolute predictions of abundance. The projected population growth was highly sensitive to the Chinook salmon abundance index. Under long-term mean Chinook abundances, the modeled SRKW population was projected to increase and when the recent Chinook salmon index values were used to set the model parameters, the average model SRKW population trajectory declined, with a chance of extinction. Model projections are based on an assumption that modeled threat conditions continue at the same levels. A continuation of the decline in the Chinook salmon stocks that SRKW depend upon can affect the future outlook of the population, and potentially increase the probability of extinction.

5.2 COMPARING INDIVIDUAL THREATS

Although the best fitting model was the cumulative impacts model, the single threat model for prey availability with both mortality and fecundity impacts did approach the observed data, indicating that variations in prey are an important contributor to predicting demographic rates in these populations. The sensitivity analysis demonstrated that prey availability is the most sensitive parameter for projecting cumulative effects, having a large impact on the population projections into the future. In the cumulative effects model, prey availability was the most important threat for these populations followed by vessel noise/presence.

The findings of this cumulative effects assessment strongly support the significant role of prey availability in determining the population trajectory of these populations, and are consistent with previous work (Lacy et al. 2017, Vélez-Espino et al. 2014b; Ford et al. 2009; 2010; Ward et al. 2009). The updated statistical analyses for the effect of prey availability on mortality and fecundity suggests that these impacts are still important to RKW, but the power of the single threat models have been reduced compared to previous studies. Ford et al. (2009) found that the coastwide Chinook index represented most of the variation in mortality index ($r^2 = 0.77$), while the updated analyses presented here showed that prey alone does not explain as much of the variation in mortality ($r^2 = 0.09$). This provides additional evidence that the cumulative threat model is a better representation of the current system.

The enduring effect of aquarium removals is evident in the SRKW population, as many more individuals were removed from the SRKW population than the NRKW. The SRKW population size would likely have been much higher in the present day if the large proportion of individuals had not been removed. The PVA model provided an opportunity to examine what may have happened to the SRKW population trajectory had these individuals remained in the population. The removals scenario results indicated that though the population growth rate may have been unchanged by the higher population numbers, SRKW population size likely would have stabilised at around 140 individuals (± 26 standard deviation), making for a more resilient and genetically diverse population.

5.3 ASSUMPTIONS AND UNCERTAINTIES

An important assumption made in this work is that the pathways of effects from threats to impacts are the same for both SRKW and NRKW, in other words, that the mechanisms by which threats affect individuals are the same for both populations. This assumption is the justification for utilising the same impact model structure for both populations, with differing threat levels. The consequence of exposure to threats is assumed to be the same while the levels of exposure to threats is assumed to be population-specific. Sub-population level variation (pods/clans) may affect the exposure to threats and were not captured in the current assessment. Knowledge about the relationships between threats and Resident Killer Whale mortality and fecundity were based on information mostly obtained in the Salish Sea area in the summer/fall period but was assumed to represent threat conditions throughout the range and throughout the year.

Further, the two populations may exploit different prey stocks that themselves have varying population growth and availability to killer whale predation. All Chinook salmon stocks went through a period of decline in the 1990s but since then have experienced differing temporal variation (Figure 6). The Chinook stocks linked to SRKW (WCVI + FL + OC) had a higher mean over the time series than the stocks linked to NRKW (FE + PS + URB), suggesting that NRKW would have less prey available if they are limited in prey choice. The ability and flexibility of these populations to exploit different stocks is unknown and may vary between NRKW and SRKW. Potential competition between populations and with other marine mammals for prey

may also affect prey availability and have not been included in the current models. Additionally, the proximity of SRKW to major population centres likely means greater exposure to sources of contamination and vessel traffic.

The projection of positive population growth under mean prey abundance with the cumulative effects model assumes that the current levels of threats remain the same and do not increase, which may not be the case in reality. Changing climate conditions and an increasing human population are having significant ongoing impacts on the marine ecosystem and are likely to continue to affect killer whales and their prey into the future. Reductions in threats may also affect the population trajectories, such as through mitigation and management actions. The US and Canada have taken a number of management actions in recent years to support the recovery of the SRKW population. Incorporating the effect of management actions, changing natural conditions and changes to threats into iterations of the cumulative effects assessment may provide useful insight into the potential effects on population trajectory.

It is cautioned that positive population growth is in no way assured by the results of this modelling exercise. The model scenarios produce mean population growth rate projections with uncertainty bounds around them. Uncertainty is addressed with stochasticity, including error rates explicitly in the model parameters. Consequently, the standard deviations for both populations include negative population growth rates, and was demonstrated by including the historical Chinook index values, which causes the SRKW population to exhibit negative growth. Therefore, if the Chinook index remains under the long-term mean in the future population growth would be expected to be negative.

The extinction probabilities and population projections of the model are only accurate if all relevant threats have been included and adequately parameterised. The representation and parameterisation of each threat in the model has associated uncertainties or limitations. Acoustic disturbance is the threat for which the least information was available. The data collected on the effects of vessels on killer whale behaviour are based on limited study time period and duration, typically from one season and/or one location (Tollit et al. 2017; Lusseau et al. 2009; Williams et al. 2002a,b). The presence and effects of vessels outside the summer season and in areas other than the Salish Sea are unknown. All vessels have been treated equally in the current treatment of vessel noise/presence and the size, acoustic signature, spatial and temporal distribution will affect the impacts on Resident Killer Whales. At present, there are no data to support clear distinctions in impacts from vessel presence and vessel noise. The effect of vessel presence may be similar to that of predator presence but little evidence exists for how to represent this component of the threat. There may be an interaction between vessel-related noise disturbance and increased incidences of other mortality events, from acoustic trauma from seismic testing or other more significant sounds, such as military munitions testing. This linkage pathway is difficult to quantify due to limited retrieval of dead killer whales and the secrecy of the spatial and temporal location of some of these disturbance events (DFO 2018a). The impact of echosounders on small vessels in close proximity to whales may also affect killer whales. Echosounder noise can be heard on Dtag deployments and the impacts are currently unknown (B. Wright, DFO, pers. comm.).

The representation of the contaminants threat also had limitations and uncertainties. From the suite of contaminants of concern for killer whales, only PCBs were able to be incorporated because this contaminant has been measured over time in this population and there is a model linking it to calf mortality (Hall et al. 2018). There is evidence that PCB contamination may affect reproductive development and disease susceptibility (Hall et al 2018), but these impacts were not able to be sufficiently quantified for use in the models. Killer whales also have significant concentrations of PBDEs but it was not possible to include this contaminant in the model due to a lack of evidence linking it to killer whale vital rates. The concentration of PCBs in killer whale

tissue was incorporated into the PVA threat model based on a model for PCB accumulation and depuration linking declines in calf survival with increased maternal PCB concentration (Hall et al. 2018). The original model was developed based on a dose-response curve extended from laboratory responses in mink, the primary concern is the uncertainty associated with converting a physiological response from a small terrestrial mammal to much larger wild killer whale population (Witting et al. 2018).

Low probability, high consequence threats, such as oil spills or disease outbreaks, are difficult to include in simulation modelling. These threats should not be ignored in management and mitigation because they can have catastrophic consequences if the population were to be exposed. The Exxon Valdez oil spill in Alaska was linked to significant declines in the resident pods that were exposed to oil, suffering significant losses in the year following the spill (Matkin et al. 1999; Matkin et al. 2008). These losses had pod-level impacts and the affected pods had not recovered to pre-spill levels 16 years after the event. One way to address high consequence events, such as a large oil spill or disease epidemic, in model simulations could be to dramatically reduce the population to 50-75% of the current levels and test if the model population would be resilient enough to recover from such a catastrophe.

5.4 CONCLUSIONS

This cumulative effects assessment further advances the field by combining a detailed Pathways of Effects conceptual model and a specific Population Viability Analysis simulation model (after Lacy et al. 2017) to evaluate how the current state of human activities affects the future persistence of the two populations. The cumulative effects population viability analysis model we have developed could be used in a number of ways by managers and scientists. One important way it could be used is to determine which threat has the most impact on long-term population persistence for a particular population, and could be extended to other killer whale ecotypes and populations. Another valuable use for the model is to test the impacts of different theoretical mitigation and management scenarios for individual threats on the population trajectory, for example to test whether the complete mitigation of acoustic disturbance would cause the population trajectory to increase over time and how long it may take for a change in population trajectory to be observable. The cumulative effects PVA model can be run using different input parameters (e.g., increased shipping) in order to consider the potential impacts of proposed developments and other anthropogenic changes in the Southern and Northern RKW's range. The ongoing research being conducted under initiatives such as Oceans Protection Plan and Whales Initiative, and others, can be used to refine the model and test possible mitigation and management actions and the impact on the long-term survival and recovery of the threatened and endangered Northern and Southern Resident Killer Whale populations. Information from ongoing and/or planned further research on RKW such as prey competition in key foraging areas, foraging efficiency, RKW diet composition, prey field analysis, underwater acoustic monitoring and modelling, contaminant sources and levels, will all help to inform future iterations of the model. Model use, and refined future versions, can help to adaptively inform and/or implement RKW recovery measures identified in the *Action Plan for the Northern and Southern Resident Killer Whale (Orcinus orca) in Canada*, such as investigating the benefits of management actions to protect important areas, evaluating potential impacts of disturbance and prey competition from fisheries, assessing the potential impact of salmon enhancement on RKW, and assessing project impacts on RKW and their habitat to provide advice on avoidance and mitigation measures as required.

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APPENDIX I: CHINOOK MODEL OCEAN ABUNDANCE AND INDEX VALUES

Appendix Table 1 - The model Chinook ocean abundance values for five stock indices, and their corresponding index values (1979-2017) with the summary statistics (mean, standard deviation, minimum and maximum values)

Year	Model Chinook Ocean Abundance					Chinook Index			
	Total	Total (excl. SEAK)	FE+PS+URB	WCVI+FL+OC	WCVI	Coastwide Index Total (excl. SEAK)	NRKW Index (FE+PS+URB)	SRKW Index (WCVI+FL+OC)	WCVI Index
1979	1,336,622	1,328,288	259,998	439,869	60,945	1.199	0.996	1.179	0.536
1980	1,203,711	1,194,004	223,034	418,398	101,538	1.080	0.854	1.121	0.892
1981	1,169,582	1,158,360	189,635	440,400	95,855	1.049	0.726	1.180	0.842
1982	1,232,148	1,221,029	180,362	508,465	171,707	1.106	0.691	1.363	1.509
1983	1,188,448	1,174,663	211,145	544,413	182,846	1.066	0.809	1.459	1.607
1984	1,323,238	1,307,815	278,484	590,565	151,996	1.187	1.067	1.583	1.336
1985	1,215,562	1,199,250	289,653	502,120	88,084	1.091	1.110	1.346	0.774
1986	1,278,693	1,260,447	335,530	442,340	66,658	1.147	1.285	1.185	0.586
1987	1,553,371	1,534,810	381,238	382,529	60,717	1.394	1.460	1.025	0.534
1988	1,483,370	1,468,110	376,360	385,109	109,780	1.331	1.442	1.032	0.965
1989	1,300,421	1,290,017	307,754	455,769	140,389	1.167	1.179	1.221	1.234
1990	1,249,674	1,241,650	263,908	546,608	213,518	1.121	1.011	1.465	1.877
1991	1,193,851	1,185,273	174,244	575,046	269,064	1.071	0.667	1.541	2.365

Year	Model Chinook Ocean Abundance					Chinook Index			
	Total	Total (excl. SEAK)	FE+PS+URB	WCVI+FL+OC	WCVI	Coastwide Index Total (excl. SEAK)	NRKW Index (FE+PS+URB)	SRKW Index (WCVI+FL+OC)	WCVI Index
1992	1,169,643	1,160,515	152,430	601,057	270,790	1.050	0.584	1.611	2.380
1993	1,059,739	1,049,669	182,438	535,455	245,199	0.951	0.699	1.435	2.155
1994	862,123	853,603	187,404	377,408	167,428	0.774	0.718	1.011	1.472
1995	677,078	669,434	160,574	223,197	71,120	0.608	0.615	0.598	0.625
1996	732,393	724,537	178,309	229,071	47,080	0.657	0.683	0.614	0.414
1997	870,410	862,521	195,083	367,002	105,969	0.781	0.747	0.984	0.931
1998	820,971	812,464	171,553	372,964	129,090	0.737	0.657	0.999	1.135
1999	764,740	754,682	211,522	244,700	61,718	0.686	0.810	0.656	0.542
2000	737,322	725,553	195,232	212,933	25,496	0.662	0.748	0.571	0.224
2001	1,072,818	1,060,708	242,166	294,405	46,236	0.963	0.928	0.789	0.406
2002	1,559,654	1,549,335	314,406	477,966	116,719	1.399	1.204	1.281	1.026
2003	1,610,659	1,601,251	359,540	530,834	153,526	1.445	1.377	1.423	1.349
2004	1,434,584	1,424,749	310,027	451,426	150,160	1.287	1.188	1.210	1.320
2005	1,176,463	1,165,783	310,805	333,769	111,600	1.056	1.191	0.894	0.981
2006	961,736	949,908	262,978	295,609	112,908	0.863	1.007	0.792	0.992
2007	746,441	734,905	205,288	227,601	100,940	0.670	0.786	0.610	0.887

	Model Chinook Ocean Abundance					Chinook Index			
Year	Total	Total (excl. SEAK)	FE+PS+URB	WCVI+FL+OC	WCVI	Coastwide Index Total (excl. SEAK)	NRKW Index (FE+PS+URB)	SRKW Index (WCVI+FL+OC)	WCVI Index
2008	745,175	738,184	199,959	174,200	55,353	0.669	0.766	0.467	0.487
2009	768,658	761,879	228,120	174,788	44,701	0.690	0.874	0.468	0.393
2010	1,029,980	1,022,934	245,614	293,418	65,508	0.924	0.941	0.786	0.576
2011	1,053,968	1,048,424	279,302	304,161	104,360	0.946	1.070	0.815	0.917
2012	901,835	897,959	243,656	214,059	70,964	0.809	0.933	0.574	0.624
2013	1,210,298	1,205,330	395,354	278,833	69,049	1.086	1.514	0.747	0.607
2014	1,484,397	1,479,438	474,485	330,696	90,298	1.332	1.818	0.886	0.794
2015	1,324,091	1,319,306	392,910	267,736	82,484	1.188	1.505	0.718	0.725
2016	1,033,825	1,030,649	342,221	259,093	113,298	0.928	1.311	0.694	0.996
2017	925,453	923,050	268,321	248,889	112,249	0.830	1.028	0.667	0.987
mean	1,114,440	1,104,884	261,052	373,151	113,778	1.000	1.000	1.000	1.000
SD	262,628	261,397	78,367	126,860	60,441	0.236	0.300	0.340	0.531
minimum	677,078	669,434	152,430	174,200	25,496	0.608	0.584	0.467	0.224
maximum	1,610,659	1,601,251	474,485	601,057	270,790	1.445	1.818	1.611	2.380

APPENDIX II: MEASURED PCB CONCENTRATIONS FOR NRKW AND SRKW

Appendix Table 2 - Summary of total polychlorinated biphenyls (Σ PCB) concentrations measured in Northern Resident Killer Whale blubber from 1993-2015 (adapted from Guy, 2018 unpubl.⁵ - Appendix E, Table E1). Source of PCB data Ross et al. 2013 unpubl.⁹

Source	Animal ID	Sex	Age	Year	# congeners analysed	Lipid (%)	Σ PCB (mg·kg ⁻¹ lw)	Σ PCB mg/kg
Ross et al. 2013 unpubl	A56	F	3	1993	205	64.3	9.80E+00	9.80
Ross et al. 2013 unpubl	A54	F	4	1993	205	64.3	9.96E+00	9.96
Ross et al. 2013 unpubl	A52	F	6	1993	205	64.3	3.39E+01	33.90
Ross et al. 2013 unpubl	A48	F	10	1993	205	64.3	1.07E+01	10.70
Ross et al. 2013 unpubl	A43	F	12	1993	205	64.3	7.41E+00	7.41
Ross et al. 2013 unpubl	A35	F	19	1993	205	64.3	1.68E+00	1.68
Ross et al. 2013 unpubl	A24	F	26	1993	205	64.3	4.79E-01	0.48
Ross et al. 2013 unpubl	A11	F	35	1993	205	64.3	1.04E+00	1.04
Ross et al. 2013 unpubl	A23	F	46	1993	205	64.3	2.58E+00	2.58
Ross et al. 2013 unpubl	A9	F	53	1993	205	64.3	2.40E+01	24.00
Ross et al. 2013 unpubl	A60	M	1	1993	205	64.3	1.31E+01	13.10
Ross et al. 2013 unpubl	A59	M	1	1993	205	64.3	1.08E+01	10.80
Ross et al. 2013 unpubl	A13	M	15	1993	205	64.3	2.12E+01	21.20
Ross et al. 2013 unpubl	A27	M	22	1993	205	64.3	2.42E+00	2.42
Ross et al. 2013 unpubl	A6	M	29	1993	205	64.3	1.79E+01	17.90
Ross et al. 2013 unpubl	B2	M	41	1993	205	64.3	2.69E+01	26.90
Ross et al. 2013 unpubl	B1	M	42	1993	205	64.3	6.90E+00	6.90

Source	Anima IID	Sex	Age	Year	# congener s analysed	Lipid (%)	ΣPCB (mg·kg- 1lw)	ΣPCB mg/kg
Ross et al. 2013 unpubl	A57	F	5	1996	205	64.3	1.09E+02	109.00
Ross et al. 2013 unpubl	A42	F	16	1996	205	64.3	1.54E+01	15.40
Ross et al. 2013 unpubl	C10	F	25	1996	205	64.3	6.90E+00	6.90
Ross et al. 2013 unpubl	I2	F	57	1996	205	64.3	9.45E+00	9.45
Ross et al. 2013 unpubl	C5	F	71	1996	205	64.3	2.55E+01	25.50
Ross et al. 2013 unpubl	B13	M	9	1996	205	64.3	2.79E+01	27.90
Ross et al. 2013 unpubl	B12	M	12	1996	205	64.3	2.00E+01	20.00
Ross et al. 2013 unpubl	H4	M	22	1996	205	64.3	2.20E+01	22.00
Ross et al. 2013 unpubl	A5	M	39	1996	205	64.3	3.82E+01	38.20
Ross et al. 2013 unpubl	I5	M	42	1996	205	64.3	3.77E+01	37.70
Ross et al. 2013 unpubl	R6	M	42	1996	205	64.3	4.96E+01	49.60
Ross et al. 2013 unpubl	A70	F	1	2000	205	64.3	4.12E+00	4.12
Ross et al. 2013 unpubl	I80	F	3	2000	205	64.3	1.94E+01	19.40
Ross et al. 2013 unpubl	A69	F	4	2000	205	64.3	1.06E+01	10.60
Ross et al. 2013 unpubl	G51	F	8	2000	205	64.3	1.57E+01	15.70
Ross et al. 2013 unpubl	I63	F	10	2000	205	64.3	1.79E+01	17.90
Ross et al. 2013 unpubl	I51	F	14	2000	205	64.3	7.85E+00	7.85
Ross et al. 2013 unpubl	I15	F	48	2000	205	64.3	1.86E+00	1.86
Ross et al. 2013 unpubl	I85	M	2	2000	205	64.3	5.75E+00	5.75
Ross et al. 2013 unpubl	C20	M	7	2000	205	64.3	5.46E+00	5.46

Source	Animal ID	Sex	Age	Year	# congeners analysed	Lipid (%)	ΣPCB (mg·kg ⁻¹ lw)	ΣPCB mg/kg
Ross et al. 2013 unpubl	C17	M	11	2000	205	64.3	6.89E+00	6.89
Ross et al. 2013 unpubl	D13	F	18	2002	205	64.3	3.48E+00	3.48
Ross et al. 2013 unpubl	D12	F	20	2002	205	64.3	1.51E+00	1.51
Ross et al. 2013 unpubl	I50	F	20	2002	205	64.3	3.76E+00	3.76
Ross et al. 2013 unpubl	I21	F	23	2002	205	64.3	2.45E+00	2.45
Ross et al. 2013 unpubl	C18	M	11	2002	205	64.3	6.28E+00	6.28
Ross et al. 2013 unpubl	I52	M	16	2002	205	64.3	8.24E+00	8.24
Ross et al. 2013 unpubl	A62	F	10	2003	205	64.3	3.54E+00	3.54
Ross et al. 2013 unpubl	I68	F	11	2003	205	64.3	7.57E+00	7.57
Ross et al. 2013 unpubl	D9	F	31	2003	205	64.3	6.37E+00	6.37
Ross et al. 2013 unpubl	A74	M	3	2003	205	64.3	6.83E+00	6.83
Ross et al. 2013 unpubl	A60	M	11	2003	205	64.3	1.53E+01	15.30
Ross et al. 2013 unpubl	I67	M	12	2003	205	64.3	2.37E+01	23.70
Ross et al. 2013 unpubl	A55	M	13	2003	205	64.3	2.60E+00	2.60
Ross et al. 2013 unpubl	R28	M	15	2003	205	64.3	3.48E+00	3.48
Ross et al. 2013 unpubl	I42	M	20	2003	205	64.3	7.15E+00	7.15
Ross et al. 2013 unpubl	A33	M	32	2003	205	64.3	1.12E+01	11.20
Ross et al. 2013 unpubl	C16	F	15	2004	205	64.3	6.65E+00	6.65
Ross et al. 2013 unpubl	A71	M	5	2004	205	64.3	3.27E+00	3.27
Ross et al. 2013 unpubl	I110	*	2	2007	205	64.3	5.14E+00	5.14

Source	Animal ID	Sex	Age	Year	# congeners analysed	Lipid (%)	ΣPCB (mg·kg ⁻¹ lw)	ΣPCB mg/kg
Ross et al. 2013 unpubl	I80	F	10	2007	205	64.3	2.50E+01	25.00
Ross et al. 2013 unpubl	B14	F	16	2007	205	64.3	1.37E+00	1.37
Ross et al. 2013 unpubl	I35	F	33	2007	205	64.3	1.21E+00	1.21
Ross et al. 2013 unpubl	R43	M	5	2007	205	64.3	1.16E+01	11.60
Ross et al. 2013 unpubl	A71	M	8	2007	205	64.3	8.18E+00	8.18
Ross et al. 2013 unpubl	C22	M	10	2007	205	64.3	8.52E+00	8.52
Ross et al. 2013 unpubl	R31	M	10	2007	205	64.3	2.12E+01	21.20
Ross et al. 2013 unpubl	A61	M	13	2007	205	64.3	9.78E+00	9.78
Ross et al. 2013 unpubl	R30	M	13	2007	205	64.3	6.59E+00	6.59
Ross et al. 2013 unpubl	I64	M	17	2007	205	64.3	4.47E+00	4.47
Ross et al. 2013 unpubl	I46	M	22	2007	205	64.3	7.22E-01	0.72
Ross et al. 2013 unpubl	A86	*	2	2008	205	64.3	5.97E+00	5.97
Ross et al. 2013 unpubl	A84	*	3	2008	205	64.3	9.30E+00	9.30
Ross et al. 2013 unpubl	I110	*	3	2008	205	64.3	1.07E+01	10.70
Ross et al. 2013 unpubl	A84	*	3	2008	205	64.3	9.99E+00	9.99
Ross et al. 2013 unpubl	A78	*	5	2008	205	64.3	2.78E+00	2.78
Ross et al. 2013 unpubl	A72	F	7	2008	205	64.3	1.45E+01	14.50
Ross et al. 2013 unpubl	A75	F	7	2008	205	64.3	1.29E+01	12.90
Ross et al. 2013 unpubl	C24	F	8	2008	205	64.3	6.14E+01	61.40
Ross et al. 2013 unpubl	A67	F	12	2008	205	64.3	5.33E+00	5.33

Source	Animal ID	Sex	Age	Year	# congeners analysed	Lipid (%)	ΣPCB (mg·kg ⁻¹ lw)	ΣPCB mg/kg
Ross et al. 2013 unpubl	A54	F	19	2008	205	64.3	1.77E+00	1.77
Ross et al. 2013 unpubl	A51	F	22	2008	205	64.3	1.11E+01	11.10
Ross et al. 2013 unpubl	R43	M	6	2008	205	64.3	1.33E+01	13.30
Ross et al. 2013 unpubl	I78	M	11	2008	205	64.3	8.52E+00	8.52
Ross et al. 2013 unpubl	R30	M	14	2008	205	64.3	1.02E+01	10.20
Ross et al. 2013 unpubl	R28	M	16	2008	205	64.3	9.11E+00	9.11
Ross et al. 2013 unpubl	I67	M	17	2008	205	64.3	9.38E+00	9.38
Ross et al. 2013 unpubl	I64	M	18	2008	205	64.3	6.81E+00	6.81
Ross et al. 2013 unpubl	I62	M	20	2008	205	64.3	8.36E+00	8.36
Ross et al. 2013 unpubl	I46	M	23	2008	205	64.3	1.53E+01	15.30
Ross et al. 2013 unpubl	I42	M	25	2008	205	64.3	1.46E+01	14.60
Ross et al. 2013 unpubl	A39	M	33	2008	205	64.3	7.68E+00	7.68
Ross et al. 2013 unpubl	A79	*	4	2009	205	64.3	1.21E+01	12.10
Ross et al. 2013 unpubl	R44	*	5	2009	205	64.3	1.34E+01	13.40
Ross et al. 2013 unpubl	R39	F	8	2009	205	64.3	7.88E+00	7.88
Ross et al. 2013 unpubl	R35	F	11	2009	205	64.3	1.33E+01	13.30
Ross et al. 2013 unpubl	R29	F	15	2009	205	64.3	9.00E+00	9.00

Appendix Table 3 - Summary of total polychlorinated biphenyls (Σ PCB) concentrations measured in Southern Resident Killer Whale blubber from 1993-2015 (adapted from Guy 2018 unpubl⁵ - Appendix E, Table E1). Source of PCB data: Krahn et al. 2007, 2009; Ross et al. 2013 unpubl.⁹; Guy 2018 unpubl⁵

Source	Animal ID	Sex	Age	Year	# congeners analysed	Lipid (%)	Σ PCB (mg·kg ⁻¹ lw)	Σ PCB Converted (mg/kg)
Ross et al. 2013 unpubl	J6	M	37	1993	205	64.3	5.93E+00	5.93
Ross et al. 2013 unpubl	J3	M	40	1993	205	64.3	1.62E+02	162.00
Ross et al. 2013 unpubl	J20	F	16	1996	205	64.3	7.47E+01	74.70
Ross et al. 2013 unpubl	J18	M	20	1996	205	64.3	6.32E+01	63.20
Ross et al. 2013 unpubl	J11	F	41	1996	205	64.3	3.47E+01	34.70
Ross et al. 2013 unpubl	J1	M	46	1996	205	64.3	1.92E+02	192.00
Ross et al. 2013 unpubl	J18	M	23	2000	205	64.3	2.48E+02	248.00
Ross et al. 2013 unpubl	L78	M	15	2004	205	64.3	8.53E+00	8.53
Krahn et al. 2007	L78	M	15	2004	45	15.2	2.20E+01	22.00
Ross et al. 2013 unpubl	L74	M	18	2004	205	64.3	2.22E+01	22.20
Ross et al. 2013 unpubl	L71	M	18	2004	205	64.3	1.78E+01	17.80
Krahn et al. 2007	L71	M	18	2004	45	9.6	3.60E+01	36.00
Krahn et al. 2007	L74	M	18	2004	45	18	4.50E+01	45.00
Krahn et al. 2007	J39	M	3	2006	45	40.9	3.40E+01	34.00
Krahn et al. 2007	J27	M	15	2006	45	30.4	7.40E+01	74.00
Krahn et al. 2007	L85	M	15	2006	45	24.8	5.00E+01	50.00
Krahn et al. 2007	J19	F	27	2006	45	29.4	4.50E+01	45.00
Krahn et al. 2007	L57	M	29	2006	45	19.4	5.60E+01	56.00
Krahn et al. 2007	J1	M	55	2006	45	21.9	1.80E+02	180.00
Krahn et al. 2009	J38	M	4	2007	45	20.9	4.10E+01	41.00
Krahn et al. 2009	K34	M	6	2007	45	22.3	3.90E+01	39.00
Krahn et al. 2009	K36	F	4	2007	45	18.3	6.20E+01	62.00
Krahn et al. 2009	L87	M	15	2007	45	25.6	2.40E+01	24.00
Krahn et al. 2009	J22	F	22	2007	45	28.4	4.60E+00	4.60

Source	Animal ID	Sex	Age	Year	# congeners analysed	Lipid (%)	ΣPCB (mg·kg ⁻¹ lw)	ΣPCB Converted (mg/kg)
Krahn et al. 2009	L67	F	22	2007	45	29.2	4.30E+00	4.30
Krahn et al. 2009	L73	M	21	2007	45	23.8	3.20E+01	32.00
Krahn et al. 2009	K21	M	35	2007	45	26.6	3.80E+01	38.00
Krahn et al. 2009	K13	F	35	2007	45	22	8.90E+00	8.90
Krahn et al. 2009	L26	F	51	2007	45	22.1	2.70E+01	27.00
Krahn et al. 2009	L21	F	57	2007	45	18.7	5.50E+01	55.00
Krahn et al. 2009	K7	F	97	2007	45	28.5	1.20E+02	120.00
Guy 2018 unpubl	J49	M	4	2015	209	64.3	2.77E+01	27.70
Guy 2018 unpubl	L103	F	13	2015	209	64.3	1.33E+01	13.30
Guy 2018 unpubl	L116	M	6	2015	209	64.3	4.75E+01	47.50
Guy 2018 unpubl	J37	F	15	2015	209	64.3	3.01E+00	3.01
Guy 2018 unpubl	Blubber	F	18	2015	209	64.3	4.41E+01	44.10
Guy 2018 unpubl	K22	F	29	2015	209	64.3	1.42E+01	14.20
Guy 2018 unpubl	L72	F	30	2015	209	64.3	9.66E+00	9.66
Guy 2018 unpubl	K25	M	25	2015	209	64.3	1.03E+01	10.30
Guy 2018 unpubl	K13	F	44	2015	209	64.3	4.83E+00	4.83

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Automatic Whale Detector, version 1.0

11 February 2015, by Rich Press



A gray whale Mother and Calf are shown. Credit: NOAA

Every year, gray whales migrate from their summer feeding grounds in the Arctic to their wintering grounds off Baja California in Mexico. And roughly every other year, scientists with binoculars count them as they funnel past a point on the California coast a bit south of Monterey Bay. Scientists conduct this survey to keep track of how well the population is doing, and this year they have a new set of eyes to help with the job.

Three eyes, to be exact. Each one is a [thermal imaging camera](#) that captures the blow from a whale as it surfaces to breathe.

"A whale is this great big motor that takes in a breath of air and holds it inside for a long time," said Wayne Perryman, a NOAA Fisheries scientist who helped develop the new system. "When it exhales, the air is much warmer than the background, and we can detect that difference very easily, both day and night."

New Tech on the Job

The cameras themselves are nothing new—they're similar to the infrared cameras that police use when searching for suspects from a helicopter.

What is new is software that automatically analyzes the video to detect when a whale blows. To do that, it has to distinguish the blow of a whale from other signals that might confuse it, such as a bird diving into the water or a small boat passing by.

"The biggest challenge was getting the detector to be as accurate as possible without having it get fooled by false alarms," said Dave Weller, the NOAA Fisheries scientist who leads the survey team.

In addition, every time the computer sees a blow, it predicts where and when that same whale will surface to blow again. That prediction algorithm, which is based on years of research into [gray whale](#) diving behavior, allows the computer to track individual whales. "If you don't have a way of tracking who's who, you can double-count some whales or miss them altogether," Weller said.



A set of three thermal imaging cameras are used to automatically detect migrating whales based on the difference in temperature between the whales' blow and the surrounding environment. Credit: NOAA

Previously, two scientists would conduct the survey—one a spotter with high-powered binoculars and the other a record-keeper. For now, human

observers are still working the survey to ensure that the automated system produces accurate results. However, human observers can only count whales during daylight, and limited budgets mean that they're onsite only during the peak weeks of the migration. But the [thermal imaging](#) system works 24/7 throughout the entire migration—it was already counting when the earliest migrants made their way south, and it will still be counting when the stragglers take up the rear.

"The biggest advantage of the new system is that it vastly increases our sample size," Weller said.

"That means we can more accurately estimate the size of the population."

A Conservation Success Story

Gray whales were hunted nearly to extinction during the whaling days. But this population of gray whales has been making a steady recovery since then, and they were taken off the endangered species list in 1994 (though a second population on the Russian side of the Pacific remains endangered). "They're a real success story as far as the management of large [whales](#) goes," said Perryman. "Today the population is up around 20,000, and that appears to be pretty stable,"

Scientists still need to keep track of gray whale populations, however. Despite their recovery, the animals are still at risk from ship strikes, entanglement in fishing gear, and other human impacts. Also, as the climate changes, scientists want to know if changes in the amount of sea ice are correlated with changes in [population](#) size or timing of the migration.

And so the count goes on. But as of this year it's more high tech, and the numbers will be a bit more accurate.

Provided by NOAA National Marine Fisheries Service

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BELUGA WHALE (*Delphinapterus leucas*): Cook Inlet Stock

STOCK DEFINITION AND GEOGRAPHIC RANGE

Beluga whales are distributed throughout seasonally ice-covered arctic and subarctic waters of the Northern Hemisphere (Gurevich 1980) and are closely associated with open leads and polynyas in ice-covered regions (Hazard 1988). In Alaska, depending on season and region, beluga whales may occur in both offshore and coastal waters, with genetically distinct summer concentrations in upper Cook Inlet, Bristol Bay, and the eastern Bering Sea (i.e., Yukon Delta and Norton Sound), eastern Chukchi Sea, and Beaufort Sea (Hazard 1988, O’Corry-Crowe et al. 2018) (Fig. 1). Data from satellite transmitters attached to whales from the Beaufort Sea, Eastern Chukchi Sea, and Eastern Bering Sea stocks show month to month ranges that include summering areas and autumn migratory routes that are relatively distinct for each population (e.g., Hauser et al. 2014, Citta et al. 2017). Tag data for beluga whales found in Bristol Bay (Quakenbush 2003; Citta et al. 2016, 2017) and Cook Inlet (Hobbs et al. 2005, Goetz et al. 2012a, Shelden et al. 2015a, 2018) show tagged whales remained in those areas throughout the year.

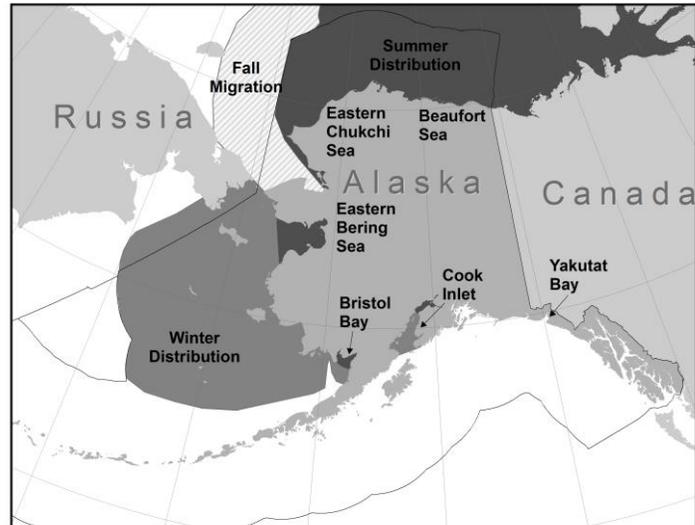


Figure 1. Approximate distribution for all five beluga whale stocks. Summering areas are dark gray, wintering areas are lighter gray, and the hashed area is a region used by the Eastern Chukchi Sea and Beaufort Sea stocks for autumn migration. The U.S. Exclusive Economic Zone is delineated by the solid black line.

Beluga whale stock structure was based on the Dizon et al. (1992) phylogeographic approach: 1) Distributional data: geographic distribution discontinuous (Frost and Lowry 1990); 2) Population response data: possible extirpation of local populations, distinct population trends among regions occupied in summer (O’Corry-Crowe et al. 2018); 3) Phenotypic data: unknown; and 4) Genotypic data: mitochondrial DNA analyses indicate distinct differences among populations in summering areas (O’Corry-Crowe et al. 2002). Based on this information, five beluga whale stocks are recognized within U.S. waters (Fig. 1): 1) Cook Inlet, 2) Bristol Bay, 3) Eastern Bering Sea, 4) Eastern Chukchi Sea, and 5) Beaufort Sea.

During ice-free months, Cook Inlet beluga whales are often concentrated near river mouths (Shelden et al. 2015a). The fall-winter-spring distribution of this stock is not fully determined; however, there is evidence that most whales in this population inhabit upper Cook Inlet year-round (Lammers et al. 2013, Castellote et al. 2015, Shelden et al. 2015a). From 1999 to 2002, satellite tags were attached to a total of 18 Cook Inlet beluga whales to determine their movement patterns (Goetz et al. 2012a; Shelden et al. 2015a, 2018). All tagged beluga whales remained in Cook Inlet, primarily in the upper inlet north of the East and West Forelands, with brief trips to the lower inlet (Shelden et al. 2015a, 2018).

A review of all marine mammal surveys and anecdotal sightings in the northern Gulf of Alaska between 1936 and 2000 found only 28 beluga whale sightings, indicating that very few beluga whales occurred in the Gulf of Alaska outside Cook Inlet (Laidre et al. 2000). Yakutat Bay is the only area in the Gulf of Alaska outside of Cook Inlet where multiple sightings have occurred (Laidre et al. 2000, Lucey et al. 2015, O’Corry-Crowe et al. 2015). Based on genetic analyses, traditional ecological knowledge (TEK), and observations by fishermen and others, the Yakutat beluga whales likely represent a small, resident group (fewer than 20 whales) that has been observed year round and is reproductively separated from Cook Inlet (Lucey et al. 2015, O’Corry-Crowe et al. 2015). Furthermore, this group in Yakutat appears to be showing signs of inbreeding and low diversity due to their isolation and small numbers (O’Corry-Crowe et al. 2015). Although the beluga whales in Yakutat Bay are not included in the Cook Inlet Distinct Population Segment (DPS) of beluga whales under the Endangered Species Act (ESA), they are

considered part of the depleted Cook Inlet stock under the Marine Mammal Protection Act (MMPA) (50 CFR 216.15; 75 FR 12498, 16 March 2010) because insufficient information was available to identify Yakutat beluga whales as a separate population when Cook Inlet beluga whales were designated as depleted under the MMPA. Thus, Yakutat Bay beluga whales remain part of the Cook Inlet stock, are designated as depleted, and are provided the same protections as the Cook Inlet stock, including limitations on hunting.

POPULATION SIZE

Aerial surveys during June documented the distribution and abundance of Cook Inlet beluga whales and were conducted by NMFS each year from 1994 to 2012 (Rugh et al. 2000, 2005; Sheldon et al. 2013), after which NMFS began biennial surveys in 2014 (Sheldon et al. 2015b) (Fig. 2). NMFS changed to a biennial survey schedule after analysis showed there would be little reduction in the ability to detect a trend given the current growth rate of the population (Hobbs 2013).

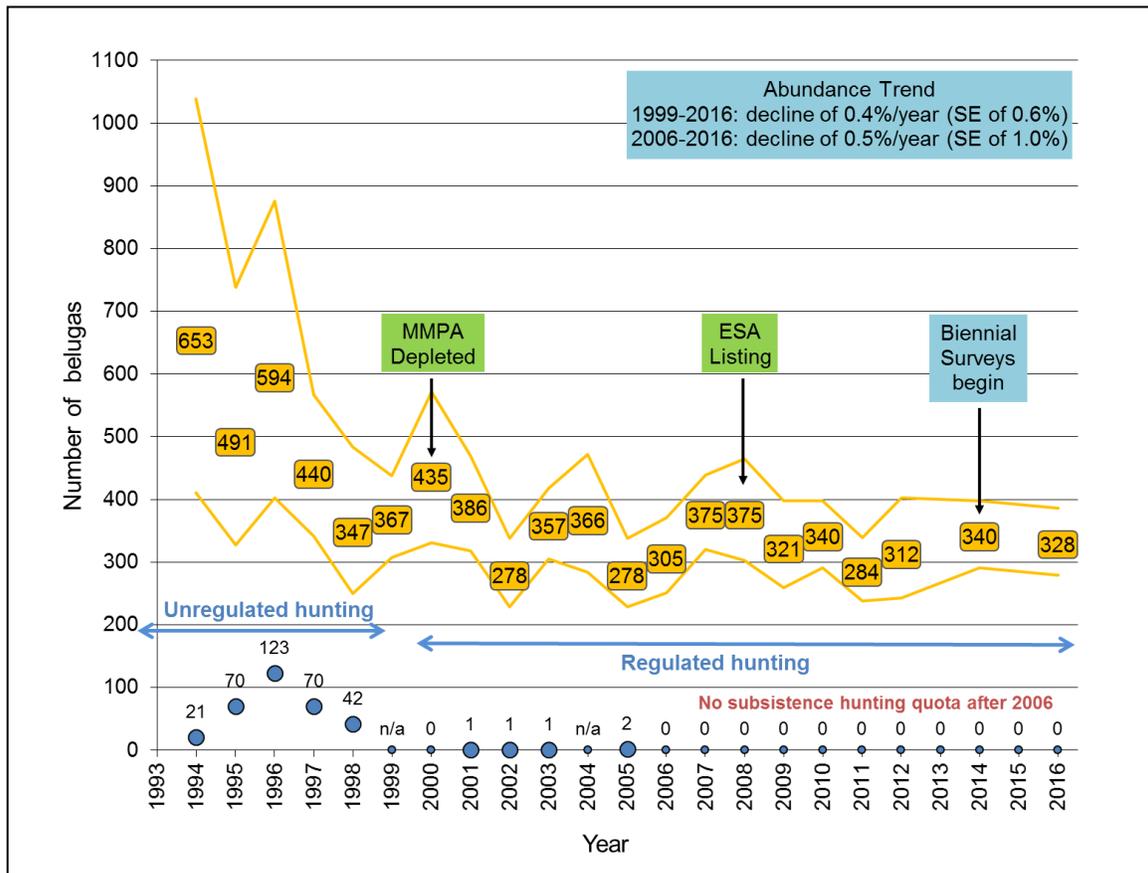


Figure 2. Annual abundance estimates of beluga whales in Cook Inlet, Alaska, 1994-2016 (Hobbs et al. 2015a, Sheldon et al. 2017). Circles show reported removals (landed plus struck and lost) during the Alaska Native subsistence hunt. A struck and lost average was calculated by the Cook Inlet Marine Mammal Council (CIMMC) and hunters for 1996, 1997, and 1998. Lines above and below each abundance estimate (number shown in box) depict the upper and lower confidence limit.

The abundance estimate for Cook Inlet beluga whales is based on counts by aerial observers and video analysis of whale groups. Paired, independent observers count each whale group while video is collected during each counting pass. Each count is corrected for subsurface animals (availability correction) and animals at the surface that were missed (sightability correction) based on an analysis of the video tapes (Hobbs et al. 2000). When video counts are not available, observers' counts are corrected for availability and sightability using a regression of counts and an interaction term with an encounter rate against the video count estimates (Hobbs et al. 2000). The estimate of the abundance equation variance was revised using the squared standard error of the average for the abundance estimates in place of the abundance estimate variance and the measurement error (Hobbs et al. 2015a). This reduced all coefficients of variation (CVs) by almost half (Hobbs et al. 2015a). The June 2016 survey resulted in an abundance estimate of 328 whales (CV = 0.08) (Shelden et al. 2017). Annual abundance estimates based on aerial surveys of Cook Inlet beluga whales during the most recent 3-survey period were 312 (2012), 340 (2014), and 328 (2016) resulting in an average abundance estimate for this stock of 327 beluga whales (CV = 0.06). An abundance survey was conducted in June 2018 and results are undergoing analysis.

Minimum Population Estimate

The minimum population estimate (N_{MIN}) is calculated according to Equation 1 from the potential biological removal (PBR) guidelines (Wade and Angliss 1997). Thus, $N_{\text{MIN}} = N/\exp(0.842 \times [\ln(1 + [\text{CV}(N)]^2)]^{1/2})$. Using the 3-survey average population estimate (N) of 327 whales and an associated $\text{CV}(N)$ of 0.06, N_{MIN} for the Cook Inlet beluga whale stock is 311 beluga whales.

Current Population Trend

The corrected annual abundance estimates for 1994-2016 are shown in Figure 2. From 1999 to 2016, the rate of decline was 0.4% (SE = 0.6%) per year, with a 73% probability that the growth rate is negative, while the 10-year trend (2006-2016) is -0.5% per year (with a 70% probability the population is declining) (Shelden et al. 2017).

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

A reliable estimate of the maximum net productivity rate (R_{MAX}) is not available for the Cook Inlet beluga whale stock. Until additional data become available, the cetacean maximum theoretical net productivity rate of 4% will be used for this stock (Wade and Angliss 1997).

POTENTIAL BIOLOGICAL REMOVAL

PBR is defined as the product of the minimum population estimate, one-half the maximum theoretical net productivity rate, and a recovery factor: $\text{PBR} = N_{\text{MIN}} \times 0.5R_{\text{MAX}} \times F_{\text{R}}$. The recovery factor (F_{R}) for this stock is 0.1, the value for cetacean stocks that are listed as endangered (Wade and Angliss 1997). Using the N_{MIN} of 311 beluga whales, the calculated PBR for this stock is 0.62 beluga whales ($311 \times 0.02 \times 0.1$).

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Information for each human-caused mortality, serious injury, and non-serious injury reported for NMFS-managed Alaska marine mammals in 2012-2016 is listed, by marine mammal stock, in Helker et al. (in press); however, only the mortality and serious injury data are included in the Stock Assessment Reports. No human-caused mortality or serious injury of Cook Inlet beluga whales was documented in 2012-2016. There are no observers in Cook Inlet fisheries, so the mean annual mortality and serious injury in commercial fisheries is unknown; although, it is likely low given that an observer program conducted in Cook Inlet in 1999-2000 did not observe mortality or serious injury of beluga whales (Manly 2006). Other potential threats most likely to result in direct human-caused mortality or serious injury of this stock include ship strikes.

Fisheries Information

Information (including observer programs, observer coverage, and observed incidental takes of marine mammals) for federally-managed and state-managed U.S. commercial fisheries in Alaska waters is presented in Appendices 3-6 of the Alaska Stock Assessment Reports.

The estimated minimum average annual mortality and serious injury rate incidental to U.S. commercial fisheries is unknown, although probably low, given that an observer program directed at the Cook Inlet commercial set and drift gillnet fisheries in 1999-2000 did not observe mortality or serious injury of beluga whales (Manly 2006).

One entanglement in a subsistence fishery was reported to the NMFS Alaska Region on 7 May 2012; a fisherman reported a juvenile beluga whale entangled and dead in a salmon set net near Kenai, Alaska. The beluga

whale was necropsied and the results indicated it was in poor health and the cause of death was drowning. However, it was not determined whether the beluga whale died before or after the net entanglement.

Alaska Native Subsistence/Harvest Information

Subsistence harvest of Cook Inlet beluga whales is important to the Native Village of Tyonek and the Alaska Native subsistence hunter community in Anchorage. Between 1993 and 1998, the annual subsistence take ranged from 17 to more than 123 beluga whales (Fig. 2), including struck and lost whales (NMFS 2016).

Following a significant decline in Cook Inlet beluga whale abundance estimates between 1994 and 1998, the Cook Inlet hunters voluntarily stood down in 1999 and the Federal government took actions to conserve, protect, and prevent further declines in the abundance of these whales. Public Laws 106-31 (1999) and 106-553 (2000) established a moratorium on Cook Inlet beluga whale harvests unless such taking occurs pursuant to a cooperative agreement between NMFS and affected Alaska Native organizations. A cooperative agreement, also referred to as a co-management agreement, was not signed in 1999 and 2004. In December 2000, an administrative hearing was held to create interim harvest regulations for 2001 through 2004 (69 FR 17973, 6 April 2004). Three Cook Inlet beluga whales were harvested under this interim harvest plan (2001-2004). In August 2004, an administrative hearing was held to create a long-term harvest plan, which allowed up to eight whales to be harvested between 2005 and 2009 (NMFS 2008). Two whales were harvested in 2005 and whales were not successfully hunted in 2006. The long-term harvest plan was signed in 2008 and established a harvest level for a 5-year period, based on the average abundance in the previous 5-year period and the growth rate during the previous 10-year period (NMFS 2008). A harvest is not allowed if the previous 5-year average abundance is less than 350 beluga whales. Under the long-term harvest plan, the 5-year average abundance during the first review period (2003-2007) was 336 whales and, therefore, a harvest was not allowed during the subsequent 5-year period (2008-2012) (73 FR 60976, 15 October 2008). The average abundance of Cook Inlet beluga whales remained below 350 whales during the second review period (2008-2012); therefore, a harvest was not allowed for the current 5-year period (2013-2017). NMFS changed to a biennial survey schedule after 2012, therefore, the 5-year average abundance is now based on either two or three surveys in a 5-year period. Hobbs (2013) showed that biennial rather than annual surveys may lead to higher variation in harvests, but it is not expected to change the probability of recovery while using the algorithm that determines the allowable harvest level.

Other Mortality

Reports from the NMFS Alaska Region stranding network are another source of information on beluga whale mortality. Beluga whale carcasses are found along the shore from lower Cook Inlet to Knik and Turnagain Arms.

Mortality related to live stranding events, where a beluga whale group strands as the tide recedes, has been regularly observed in upper Cook Inlet (Table 1). Improved reports include the number of live stranded beluga whales, as well as floating and beachcast carcasses (NMFS 2016; <https://alaskafisheries.noaa.gov/sites/default/files/16strandings.pdf>, accessed December 2018). Most whales involved in a live stranding event survive, although some associated deaths may not be observed if the whales die later from live-stranding-related injuries (Vos and Sheldon 2005, Burek-Huntington et al. 2015). Between 2012 and 2016, there were approximately 116 beluga whales involved in six known live stranding events, with two associated deaths reported (Table 1; NMFS 2016). In 2014, necropsy results from two whales found in Turnagain Arm suggested that a live stranding event contributed to their deaths as both had aspirated mud and water. No live stranding events were reported prior to the discovery of these dead whales, suggesting that not all live stranding events are observed (Table 1). Most live strandings occur in Knik Arm and Turnagain Arm, which are shallow and have extensive mudflats and strong currents. Turnagain Arm has the largest tidal range in the U.S., with a mean of 9.2 m (30 ft).

Table 1. Cook Inlet beluga whale strandings investigated by NMFS during 2012-2016 (NMFS 2016).

Year	Floating and beachcast carcasses	Number of beluga whales per live stranding event (number of associated known or suspected resulting deaths)
2012	3	12 (0), 23 (0), 3 (0)
2013	5	0
2014	10	unknown (2), 76+ (0)
2015	3	2 (0)
2016	8	0
Total	29	116+ (2)

Another source of beluga whale mortality in Cook Inlet is predation by transient-type (mammal-eating) killer whales. Killer whale sightings were not well documented and were likely rare in the upper inlet prior to the mid-1980s. From 1982 through 2016, NMFS received 31 reports of killer whale sightings in upper Cook Inlet (north of the East and West Forelands). Up to 12 beluga whale deaths, inlet-wide, were suspected to be a direct result of killer whale predation (NMFS 2016). The last confirmed killer whale predation of a Cook Inlet beluga whale occurred in 2008 in Turnagain Arm. From 2012 through 2016, NMFS received two separate killer whale sighting reports (both in 2015) in upper Cook Inlet, but there were no reports of predation attempts. Transient killer whale signals have been detected on acoustic moorings in upper Cook Inlet (Castellote et al. 2016a), but only once in a 5-year period (Castellote et al. 2016b).

Between 1998 and 2013, 38 necropsies were performed on beluga whale carcasses (23% of the 164 known stranded carcasses) (Burek-Huntington et al. 2015). The sample included adults (n = 25), juveniles (n = 6), calves (n = 3), and aborted fetuses (n = 4). When possible, a primary cause of death was noted along with contributing factors. Cause of death was unknown for 29% of the necropsied carcasses. Other causes of death were attributed to various types of trauma (18%), perinatal mortality (13%), mass stranding (13%), single stranding (11%), malnutrition (8%), or disease (8%). Several animals had mild to moderate pneumonia, kidney disease, and/or stomach ulcers that likely contributed to their deaths.

A photo-identification study (Kaplan et al. 2009) did not find any instances where Cook Inlet beluga whales appeared to have been entangled in, or to have otherwise interacted with, fishing gear. However, in 2010, a beluga whale with a rope entangled around its girth was observed and photo-documented during May through August. The same whale was photographed in July and August 2011, August 2012, and July 2013, still entangled in the rope line (McGuire et al. 2014). This whale is currently considered to have a non-serious injury (Helker et al. in press).

STATUS OF STOCK

The Cook Inlet beluga whale stock was designated as depleted under the MMPA in 2000 (65 FR 34590, 21 May 2000) and listed as endangered under the ESA in 2008 (73 FR 62919, 22 October 2008). Therefore, the Cook Inlet beluga whale stock is considered a strategic stock.

There are key uncertainties in the assessment of the Cook Inlet stock of beluga whales. The stock decline is well documented. While the early decline was likely due to unrestricted subsistence hunting, it is unknown what has prevented recovery of this stock, because subsistence harvest has not been allowed since 2006, and the mortality and serious injury in commercial fisheries is likely low. PBR is designed to allow stocks to recover to, or remain above, the maximum net productivity level (Wade 1998). An underlying assumption in the application of the PBR equation is that marine mammal stocks exhibit certain dynamics. Specifically, it is assumed that a depleted stock will naturally grow toward Optimum Sustainable Population and that some surplus growth could be removed while still allowing recovery. However, the Cook Inlet beluga whale population is far below historical levels and yet, for unknown reasons, is not increasing. If the Cook Inlet beluga whale population was increasing at an expected rate of ~2-4%, it would currently be adding, on average, about 7-13 whales per year to the population. Although there is currently no known direct human-caused mortality (e.g., from fisheries bycatch, hunting, or other sources), even if the PBR level (~1 whale every 2 years) was taken, it is clear this would have little consequence for the overall population trend given the unexplained lack of increase by 7-13 whales per year. However, given the endangered status of this population, even one take every 2 years may still impede recovery.

HABITAT CONCERNS

Beluga whale critical habitat includes two geographic areas of marine habitat in Cook Inlet that comprise 7,800 km² (3,013 mi²), excluding waters of the Port of Anchorage (76 FR 20180, 11 April 2011). Based on

available information from aerial surveys, tagged whales, and opportunistic sightings, beluga whales remain within the inlet year-round. Review of beluga whale presence data from aerial surveys, satellite tagging, and opportunistic sightings collected in Cook Inlet from the late 1970s to 2014 show their range has contracted remarkably since the 1970s (Shelden et al. 2015a). Almost the entire population is found in northern Cook Inlet from late spring through the summer and into the fall. This differs markedly from surveys in the 1970s when beluga whales were found in, or would disperse to, lower Cook Inlet by midsummer. Since 2008, on average, 83% of the total population occupied the Susitna Delta in early June during the aerial survey period, compared to roughly 50% in the past (1978-1979, 1993-1997, 1998-2008). The 2009-2014 distribution was estimated to be only 25% of the range observed in 1978-1979 (Shelden et al. 2015a). Rugh et al. (2000) first noted that whales had not dispersed to the lower inlet in July during surveys in the mid-1990s. This was also evident during aerial surveys conducted in July 2001 (Rugh et al. 2004). Whales transmitting locations from satellite tags during July in 1999 and 2002 also remained in the northern reaches of the upper inlet (Shelden et al. 2015a). During surveys in the 1970s, large numbers of whales were scattered throughout the lower inlet in August (Shelden et al. 2015a). This was not the case in 2001, when counts in the upper inlet in August were similar to those reported in June and July (Rugh et al. 2004). In August, only 2 of 10 tagged whales spent time in offshore waters and the lower inlet (Shelden et al. 2015a). The number of whales observed during the August calf index surveys, conducted from 2005 to 2012, was similar to the June surveys (Hobbs et al. 2015a, Shelden et al. 2015a), suggesting the contraction in range continued into late summer. While surveys were not conducted in September during the 1970s and 1980s, aerial surveys in 1993 showed some dispersal into lower inlet waters by late September (Shelden et al. 2015a). However, surveys in September and October of 2001 resulted in counts that were similar to June (Rugh et al. 2004). With the exception of three whales that spent brief periods of time in the lower inlet during September and/or October, most whales transmitting locations in 1999, 2000, 2001, and 2002 remained in the upper inlet north of the East and West Forelands (Shelden et al. 2015a). Counts during aerial surveys in September 2008 were also similar to June (Shelden et al. 2015a). The population appears to be consolidated into habitat in the upper-most reaches of Cook Inlet for much longer periods of time, in habitat that is most likely to be noisy (e.g., Moore et al. 2000, Lowry et al. 2006, Hobbs et al. 2015b, Kendall and Cornick 2015, Norman et al. 2015). Whether this contracted distribution is a result of changing habitat (Moore et al. 2000), prey concentration, or predator avoidance (Shelden et al. 2003) or can simply be explained as the contraction of a reduced population into small areas of preferred habitat (Goetz et al. 2007, 2012b) is unknown.

Goetz et al. (2012b) modeled habitat preferences using NMFS' 1994-2008 June abundance survey data. In large areas, such as the Susitna Delta (Beluga to Little Susitna rivers) and Knik Arm, there was a high probability that beluga whales were in larger group sizes. Beluga whale presence also increased closer to rivers with Chinook salmon (*Oncorhynchus tshawytscha*) runs, such as the Susitna River. The Susitna Delta also supports two major spawning migrations of a small, schooling eulachon (*Thaleichthys pacificus*) in May and June (Goetz et al. 2012b). Identified in the Cook Inlet Beluga Recovery Plan (NMFS 2016) are potential threats of 1) high concern: catastrophic events (e.g., natural disasters, spills, mass strandings), cumulative effects of multiple stressors, and noise; 2) medium concern: disease agents (e.g., pathogens, parasites, and harmful algal blooms), habitat loss or degradation, reduction in prey, and unauthorized take; and 3) low concern: pollution, predation, and subsistence hunting. The recovery plan did not treat climate change as a distinct threat but rather as a consideration in the threats of high and medium concern.

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Marine seismic surveys and ocean noise: time for coordinated and prudent planning

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Marine seismic surveys use intense (eg ≥ 230 decibel [dB] root mean square [RMS]) sound impulses to explore the ocean bottom for hydrocarbon deposits, conduct geophysical research, and establish resource claims under the United Nations Convention on the Law of the Sea. The expansion of seismic surveys necessitates greater regional and international dialogue, partnerships, and planning to manage potential environmental risks. Data indicate several reasons for concern about the negative impacts of anthropogenic noise on numerous marine species, including habitat displacement, disruption of biologically important behaviors, masking of communication signals, chronic stress, and potential auditory damage. The sound impulses from seismic surveys – spanning temporal and spatial scales broader than those typically considered in environmental assessments – may have acute, cumulative, and chronic effects on marine organisms. Given the international and transboundary nature of noise from marine seismic surveys, we suggest the creation of an international regulatory instrument, potentially an annex to the existing International Convention on the Prevention of Pollution from Ships, to address the issue.

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The time has come to improve international coordination among industries, governments, scientists, and environmental organizations in understanding and managing the risks that marine seismic surveys can pose to individual animals, populations, and ecosystems. Marine seismic surveys represent a major contributor to ocean noise in terms of overall energy and spatiotemporal ranges of influence; other important contributors include commercial

shipping (Hildebrand 2009). Technological improvements and economic market forces in petroleum and natural gas exploration have extended the spatial and temporal reach of seismic surveys, notably into higher latitudes and deeper waters, during most months of the year. This not only increases the potential total area for development, but in some regions (eg the Mediterranean and northeastern North Atlantic) also raises issues regarding overlapping jurisdiction and governance. Recently, enabled in part by declines in sea ice, seismic exploration has expanded rapidly into many parts of the Arctic. This has motivated countries bordering the Arctic Ocean to gain exclusive access to seabed resources by claiming sovereign rights over the extended continental shelf, under Part VI of the United Nations (UN) Convention on the Law of the Sea (UNCLOS 1982). These developments, coupled with the demand for hydrocarbon resources, are key drivers of the expansion of seismic surveys worldwide. Although the market forces governing this expansion are ephemeral, these commercial and political activities are generally occurring at ever-larger scales and extending into previously unexplored areas. An integrated program for monitoring, mitigating, and reporting would facilitate development of a knowledge-based understanding of potential risks and solutions; the establishment of such a program would necessitate coordination and prudent planning.

Efforts to monitor the undersea acoustic environment and manage the impacts of noise generated by human activities have reached a critical juncture. The European Union (EU) has recognized ocean noise as an indicator of environmental quality under its Marine Strategy Framework Directive (EU 2008) and is in the process of developing targets for achieving “good environmental

In a nutshell:

- Marine seismic surveys produce intense sound impulses to explore the ocean floor for energy sources and for research purposes
- Environmental reviews of seismic surveys are seldom undertaken at scales necessary to meaningfully assess, mitigate, and monitor their impacts; managing exposure of marine animals to these sounds requires additional attention and data
- Current exposure threshold criteria fail to account for the best available science and the cumulative effects of simultaneous seismic surveys and prolonged, repeated exposures
- Increasing marine seismic surveys, especially in ecologically sensitive areas, require multi-institutional and international collaboration to effectively manage risks
- We propose that anthropogenic ocean noise be addressed through the revision of the existing MARPOL Convention or negotiation of a new convention that more comprehensively evaluates the associated risks, benefits, and procedures

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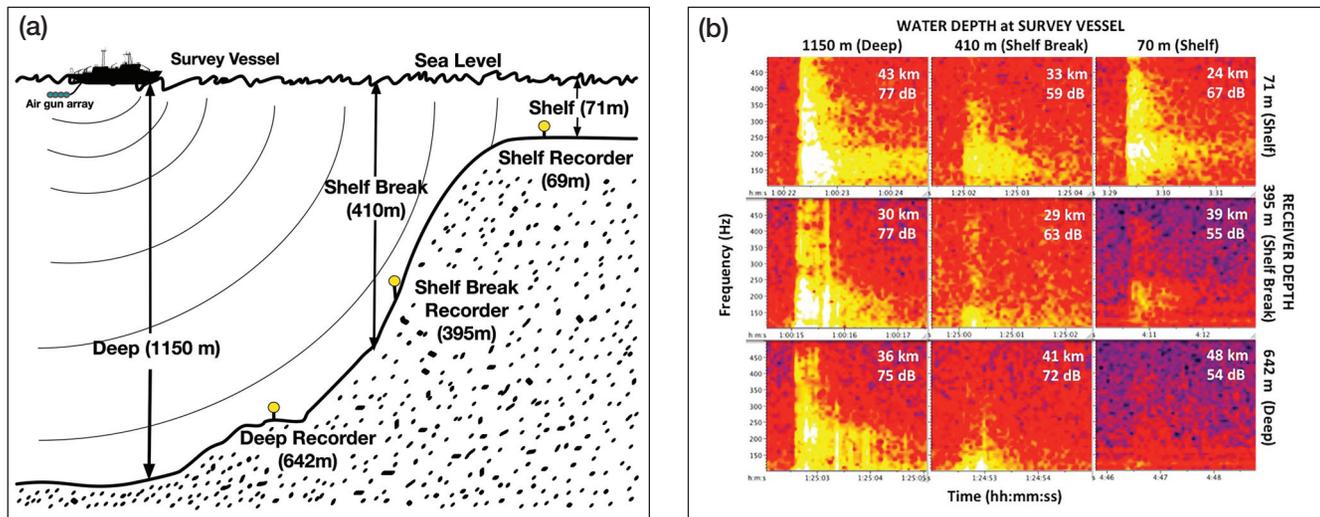


Figure 1. (a) Depiction of a marine seismic survey occurring in deep water. The diagram shows a seismic survey vessel relative to acoustic recorders (data from these recorders are shown in [b]), which are deployed 2 m above the seafloor at three water depths: deep (642 m), shelf break (395 m), and shelf (69 m). The seismic airgun array is towed at ~10-m depth off the stern. In this case, the seismic vessel is operating over deep water (1150 m), and later operating over the shelf break (410 m) and shelf (71 m) areas. Note that the data samples shown in (b) are from sounds from the vessel operating at all three water depths, but seismic signals were recorded at each recording station when the ship operated at every depth. (b) Spectrograms (each representing higher relative intensity as brighter color with time on the x axis and frequency on the y axis) illustrating seismic impulses as received by recorders at three depths (rows) when the seismic survey vessel was operating at three water depths (columns). The three water depths and the three recorder depths for each of the nine examples are given in (a). The distance (km) from the source (vessel) to the receiver (recorder) and the received levels (dB re: $1 \mu\text{Pa}^2\text{-sec}$) for each impulse are given in the upper right corner of each spectrogram. In the examples shown here, when the source is operating in deep water (1150 m), the higher received level (77 dB) occurs at the shallowest receiver (69 m; row 1, column 1) at the longest range (43 km). In contrast, when the source is operating in shallow water (71 m), the lowest received level (54 dB) occurs at the deepest receiver (642 m; row 3, column 3) at the longest range (48 km).

status” for ocean noise and acute noise-producing activities; moreover, in 2014, the EU identified seismic-survey noise as a factor in the preparation of environmental impact assessments (EIAs; EU 2014). Similarly, the US recognizes underwater noise in the preparation of EIAs for oil and gas development in regions under its jurisdiction, particularly the Gulf of Mexico, the Atlantic Ocean, and the Arctic Ocean (eg BOEM 2014, a and b). These efforts, which are still under development, are indicative of the stage and scale of actions required to address these critical issues.

Advances in integrated monitoring, assessment, and planning are essential for nations in the early phases of offshore hydrocarbon exploration, such as Greenland (administered by Denmark), which is presently assembling its initial regulatory structure with regard to seismic surveys. An integrated approach requires increasing both the breadth and depth of baseline data on the demographic trends and overall health of marine animal populations, as well as analyzing the cumulative effects of exposure to multiple noise sources and the potential interactions between those sources and other anthropogenic stressors. These analyses must be conducted on appropriate temporal and spatial scales, which may span jurisdictional boundaries or extend beyond national legislation. Under certain conditions, seismic survey signals can be detected at great distances – in one instance,

4000 km – from their respective sources (Nieukirk *et al.* 2004, 2012). While the specific effects of such signals on marine species at these ranges are not known, current monitoring, assessment, and mitigation approaches fail to consider both the spatiotemporal extent of the acoustic phenomena and the potential impacts even at moderate ranges (10–100 km), distances through which acoustic energy from the pulses can propagate efficiently (Figure 1). Notably, the survey from which the data in Figure 1 were acquired occurred in Arctic waters, where the sound velocity profile favors a surface duct [Urick 1983], thus resulting in the retention of energy near the surface and efficient propagation of energy onto the shelf when the source vessel is operating over deep water. These observations contrast with those reported by Nieukirk *et al.* [2012], in which conditions favored downward propagation such that energy from sources in shallow and shelf-break waters propagated very efficiently to deep-water recorders located thousands of kilometers away). An integrated approach to assessment must be coupled with appropriate mitigation that focuses on the acoustic ecology of marine animals and the minimization of cumulative acoustic exposures (Rio Declaration 1992).

Here, we offer perspectives on the management of seismic operations and the mitigation of the accompanying risks.

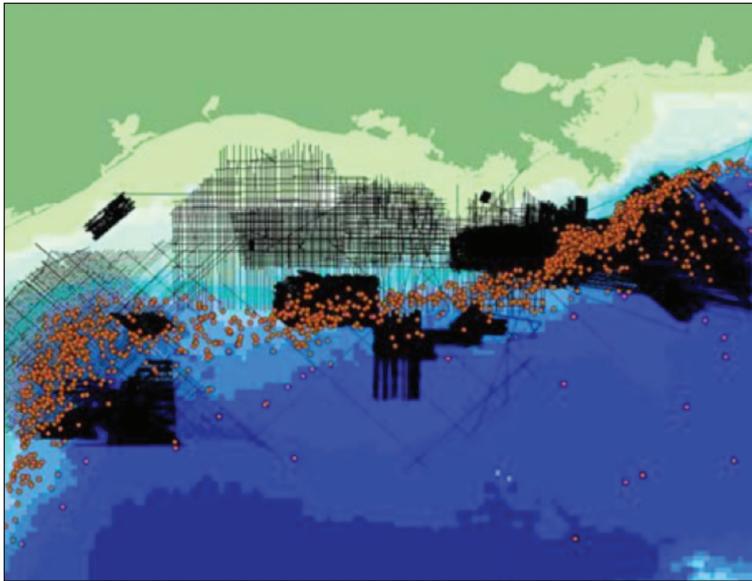


Figure 2. *Overlap of seismic surveys and sperm whale (*Physeter macrocephalus*) locations in the Gulf of Mexico. The black lines and blotches are seismic survey tracklines reported by the International Association of Geophysical Contractors (IAGC) for the period from July 2002 to October 2006. Red dots are locations of sperm whales during the same period, as determined by satellite-linked tags attached to individual whales. Blue colors in the background indicate depth contours, with the darkest blue being the 2000-m contour, the next lighter shade being the 1000-m contour, and so on. From Jochens *et al.* (2008).*

■ Appropriate impact thresholds

The potential impacts of seismic surveys, as with other anthropogenic noise sources, are typically assessed as the results of individual activities (eg a single survey) through relatively simple methods based entirely on expected sound exposure levels and decades-old guidelines (HESS Team 1999). Impact is evaluated on the estimated number of animals subjected to a sound level high enough to possibly cause harm or disturbance. While sound-exposure levels are clearly important for individual animals over the short spatial and temporal scales generally analyzed, recent documentation of the areas affected by seismic signals indicates that a broader paradigm of assessment is required (Guerra *et al.* 2011; Nieuwkerk *et al.* 2012). Given the ubiquity of seismic surveys in some areas (Figure 2) and the potential for impacts in the large areas currently being opened for resource exploration (Figure 3), we are concerned about the simplicity, artificial rigidity, and increasingly outdated nature of impact thresholds and the methods used to quantify the potential impacts of discrete activities in environmental assessments and rulemaking. To explore this subject further and to elaborate on related issues (eg masking [Clark and Ellison 2004; Clark *et al.* 2009; Hatch *et al.* 2012], stress [Warner and Heimstra 1971; Evans 2003; Otten *et al.* 2004; Wright *et al.* 2007], and behavioral responses [Castellote *et al.* 2012]), we provide additional information in WebPanel 1 and Table 1.

To move beyond the currently applied acute-impact thresholds and to capture a more realistic metric for the potential impacts of seismic surveys and other marine noise-generating activities, regulatory agencies should implement an analytical function to provide a probabilistic assessment of impact severity. Such a function should incorporate the spatial and temporal dynamics and spectral characteristics of the acoustic field generated by the specific activity (eg a seismic survey), as well as account for the aggregate sound field resulting from multiple anthropogenic activities, industrial or otherwise.

■ Increasing the breadth and depth of baseline data

The fact that insufficient data existed for many Gulf of Mexico species prior to the 2010 *Deepwater Horizon* disaster – because of inadequate sampling – indicates a broad failure on the part of federal resource-management agencies in the US. This failure limits scientists' ability not only to assess the true impacts of the Gulf disaster in retrospect but also to anticipate and plan for future prevention and remediation. Unfortunately, this lack of baseline biological data is not unique to the Gulf of Mexico.

Indeed, many places around the world where considerable seismic exploration is ongoing or is projected to occur, suffer from similar, or worse, baseline data shortfalls. For most cetacean populations, numerical abundance and trends in abundance are the standard measurements for assessing population status, but such measurements are difficult to obtain and have limitations (eg require decades of research and are complicated by resolution of temporal trends; see Taylor *et al.* 2007). It should therefore be a priority to develop additional metrics for population health (eg Harwood *et al.* 2011), such as measures of vital rates, distribution, ranging patterns, population structure, and body condition. Where possible, such information should be collected in areas where seismic activities are planned or ongoing, and existing data should be used to the maximum extent possible. Next, a reasonable understanding of the ecosystem is needed, as changes in prey availability and distribution can also drive changes in predator populations and distributions. Obtaining such data is especially critical, as climate-driven changes may be altering community structure and function (eg Grebmeier *et al.* 2006).

Without sufficient baseline data, we believe it is unrealistic for regulators to reach scientifically reliable conclusions about the risks to marine life from marine seismic surveys. The process for permitting surveys must take these data needs into account and be adjusted accordingly, and perhaps even paused while such information is gathered.

■ Cumulative effects

Assessing the impact of seismic surveys has typically been limited to individual environmental assessments that often focus only the loudest sound source (eg seismic airgun arrays) and for a single survey. This ignores sub-bottom profilers, support vessels, undersea communication systems, shipping vessels, and other major sources of noise that must be quantitatively analyzed in combination with airgun surveys to comprehensively estimate the potential impacts on marine life (eg Southall *et al.* 2013). Given our rapidly improving understanding of the spatial, temporal, and spectral scales of the acoustic footprints generated by these seismic activities, this single-source regulatory approach is no longer appropriate. In some

countries, national legislation – such as Australia’s Environment Protection and Biodiversity Conservation Act, Canada’s Species at Risk Act, and the US’s Marine Mammal Protection Act and Endangered Species Act – make regulators responsible for ensuring that activities have minimal impacts on marine populations, particularly those of threatened or endangered species, and for prescribing mitigation strategies that would reduce impacts to the lowest possible level. Regulators are failing to meet their statutory obligations if the cumulative exposure to and potentially interacting influences of the full suite of anthropogenic activities occurring in the same region are being inadequately evaluated.

We acknowledge the practical challenges involved in trying to understand and manage interacting and/or cumulative impacts (eg limitations in quantifying impacts, regulatory governance within borders that are largely artificial given the transboundary nature of sound propagation, animal movements). However, it is essential that standards be applied programmatically and internationally to include all seismic exploration and associated anthropogenic activities that could potentially affect marine species or populations thereof, and we propose a structure for such application below. For migratory and resident marine animals, particularly those with limited dispersal abilities, the potential for disturbance from cumulative impacts is high. Migratory baleen whales, for example, are likely to encounter seismic surveys in many parts of their home ranges, including feeding and breeding grounds and the migration routes between them (eg Rosenbaum *et al.* 2014). Furthermore, seismic activities need to be managed programmatically and through multi-year processes, rather than through separate harassment authorizations, as has been the standard procedure in the US. Recent steps toward that end, including the

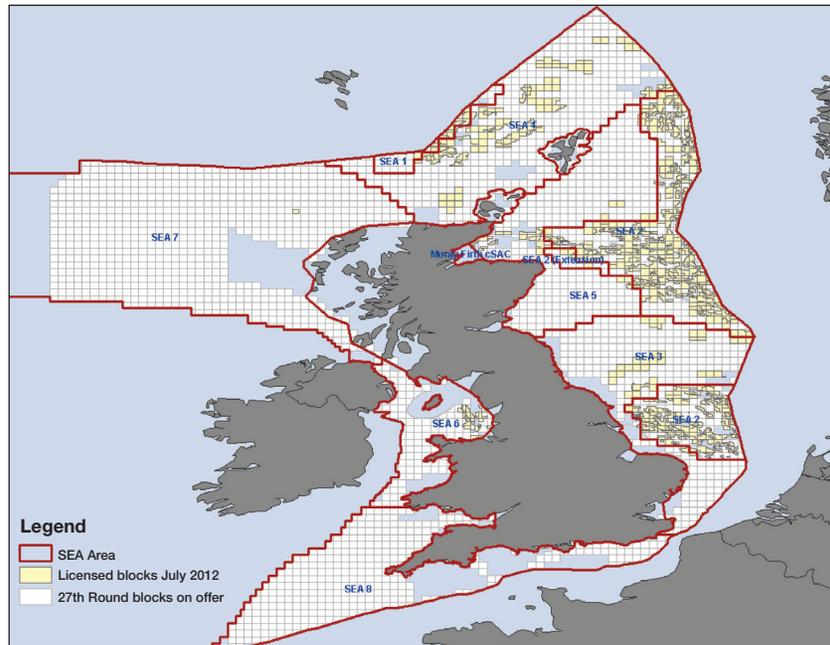


Figure 3. Undersea oil and gas lease blocks (rectangles) on offer on submerged land around the UK and those licensed in 2012 by the UK government (data obtained from the UK Department of Energy and Climate Change). The areal extent that is available for development is of concern with respect to seismic surveys for several reasons; the Joint Nature Conservation Committee, for example, receives inadequate notice of survey activity (sometimes only days), which can occur in any block (M Tasker, pers comm). “SEA” indicates a Strategic Environmental Assessment area, and the “27th Round” indicates the round of licensing of oil/gas leases from the UK government and includes seismic as well as other activities; licenses are time-limited.

recent Draft Environmental Impact Statement for industrial development in the US Arctic (NMFS 2013), are encouraging, at least in that cumulative and potentially interacting effects are considered. Specifically in the case of Arctic development, though the National Marine Fisheries Service identified and described the problem, the agency neglected to propose an explicit process for analyzing or accounting for its cumulative impacts. This is clearly a challenging management task, and some tangible efforts and measures – using a risk assessment paradigm, for instance – are sorely needed.

The relative value of individual habitats (eg feeding, breeding, migratory) should be considered, and repeat exposures of animals that display a strong fidelity to a particular habitat should be part of cumulative effects analyses. The sustained presence of animals in an area under development is an insufficient indicator of the absence of adverse impacts, particularly given the challenges of detecting population trends (Taylor *et al.* 2007). Some animals may have limited abilities to move elsewhere, and their decision to remain in an area may likely reflect tolerance (ie persisting in an important area despite the cost) rather than habituation (Bejder *et al.* 2009). Indeed, recent studies on seismic-survey impacts have documented responses such as declines in prey capture signals, a proxy for foraging success, in cetaceans that have not abandoned the affected portions of their home ranges (eg

Table 1. Summary of documented effects of seismic surveys on fish and marine mammals

Species	Location	Response/effect	Received level	Reference(s)
Bowhead whale (<i>Balaena mysticetus</i>)	Arctic	Change in surface respiration; avoidance; call cessation	120–130 dB re: 1 μ Pa RMS; 116–129 dB re: 1 μ Pa RMS	Richardson <i>et al.</i> (1999); Robertson <i>et al.</i> (2013); Blackwell <i>et al.</i> (2013)
Sperm whale (<i>Physeter macrocephalus</i>)	Gulf of Mexico	Buzz (feeding) rate decline	135–147 dB re: 1 μ Pa RMS	Miller <i>et al.</i> (2009)
Harbor porpoise (<i>Phocoena phocoena</i>)	North Sea	Temporary displacement; buzz (feeding) rate decline	145–151 dB re: 1 μ Pa ² -sec; 130–165 dB re: 1 μ Pa ² -sec	Thompson <i>et al.</i> (2013); Pirootta <i>et al.</i> (2014)
Beluga whale (<i>Delphinapterus leucas</i>)	Arctic	Temporary displacement	~130 dB re: 1 μ Pa RMS	Miller <i>et al.</i> (2005)
Humpback whale (<i>Megaptera novaeangliae</i>)	Angola	Singing and singers declined	120–150 dB re: 1 μ Pa peak	Cerchio <i>et al.</i> (2014)
Fin whale (<i>Balaenoptera physalus</i>)	Mediterranean	Altered singing and abandonment of habitat	~15 dB 1 μ Pa above background	Castellote <i>et al.</i> (2012)
Fish (herring, blue whiting [<i>Micromesistius poutassou</i>])	Norway	Displacement, horizontal and vertical	Unknown, occurred over large study area	Slotte <i>et al.</i> (2004)
Fish (cod, pollock [<i>Pollachius spp.</i>])	UK (Scotland)	Short-term startle, no long-term effects	Variable	Wardle <i>et al.</i> (2001)
Fish (pink snapper [<i>Pagrus auratus</i>])	Captive	Hearing system damage	Variable 150–180 dB re: 1 μ Pa RMS	McCauley <i>et al.</i> (2003)

Notes: This is not intended to be a complete compilation, only a sample showing representative species, geographic locations, and documented responses/effects. Ongoing studies (eg Cato *et al.* 2011) should provide additional information about effects of seismic surveys. dB = decibel; μ Pa = micropascal; RMS = root mean square.

Miller *et al.* 2009; Pirootta *et al.* 2014). The cumulative, synergistic, and chronic effects of elevated noise levels, including those from “intermittent” sounds such as seismic airguns and sounds at relatively low received levels (eg Figure 1), are detrimental in humans and other mammals, affecting hormone systems as well as behavior (eg Warner and Heimstra 1971; Evans 2003; Otten *et al.* 2004; Wright *et al.* 2007). These effects of elevated noise levels should be an explicit component of environmental impact statements and rulemakings, rather than being vaguely acknowledged but not substantively addressed.

■ A responsible way forward

Given the transboundary scale and numerous sources of anthropogenic sound in the world’s oceans – including noise from marine seismic surveys, which are ubiquitous and increasing in abundance – we believe that a responsible path forward should focus on the creation of legally binding international commitments.

Successful precedents exist for crafting such agreements, including the Cartagena Protocol on Biosafety (CPD 2000; UNSCBD 2000) organized under the Convention for Biological Diversity (CBD 1992) and the Convention on Long-range Transboundary Air Pollution (CLRTAP 1979). The CLRTAP was the first international legally binding instrument to address issues of air pollution on a broad regional basis and, notably, created an institutional framework for integrating research and policy. Annex VI,

the most recent substantial amendment to the International Convention for the Prevention of Pollution from Ships (MARPOL 1973/1978; Annex VI entered into force 19 May 2005), regulates emissions of air pollution from ships. Many parallels exist between air pollution and noise in the ocean; for instance, sound from seismic surveys, similar to atmospheric emissions from ships, may travel thousands of kilometers from its source. The precedential authority of an air pollution convention is strengthened because the EU and various international authorities, such as the Convention on Biological Diversity and Convention on Migratory Species, now classify ocean noise as a pollutant. Furthermore, the International Organization for Standardization (ISO 2012, 2014) and the International Maritime Organization (IMO; IMO 2014) have produced guidelines to, respectively, measure and reduce underwater noise from commercial vessels.

There are several options for creating new and legally binding commitments to control sources of noise in the oceans. First, member states of the IMO could pursue an annex to MARPOL 1973/1978 through the Marine Environmental Protection Committee. This approach is attractive because it leverages an existing and effective framework, and could include radiated underwater noise from vessel operation along with geophysical survey noise. But it is not without challenges. Provisions within MARPOL that cover underwater noise would have to amend the Convention’s definition of “harmful substances”, which currently does not capture energy or

sound, and perhaps revise the definition of “discharge”, which presently omits the release of harmful substances associated with offshore mineral development (although oil and gas do not have the physical properties of “minerals”, they are included as such in the legal regimes governing the seabed in Part XI and the resources of the continental shelf in Part VI of the Law of the Sea Convention, and would also have to be considered). MARPOL applies solely to “ships”, however, and some might suggest that the term’s definition, while broad, excludes towed airgun arrays. But these issues present only semantic obstacles, which can be overcome if states are committed to the regulation of noise. As an alternative, states could negotiate a brand new convention to regulate all non-military sources of underwater noise, including those emanating from industrial, geophysical, and civilian vessel sources. The process for negotiation of such an instrument is lengthy but the obstacles would not be insurmountable. Building on previous ideas for sustainable governance of ocean issues (Costanza *et al.* 1998) and integrated ocean management (Foley *et al.* 2013), we propose that a convention on ocean noise could draw upon leading analytical frameworks of decision-making behavior in the context of policy and governance processes of change (eg the Institutional Analysis and Development [IAD] Framework [McGinnis 2011] or the social-ecological system framework [Ostrom 2009; Basurto and Nenadovic 2012]). Within one of these frameworks, different scenarios for sustainable planning can be explored. Scenario planning has been used extensively in the commercial sector, is a well-developed tool in business planning (Schoemaker 1995), and is emerging as an essential element in studies of the environment (Peterson *et al.* 2003a, b) and global change (Bennett *et al.* 2003). Importantly, an accepted international convention would facilitate long-range planning in the use of acoustic sources.

We suggest the following as an initial list of measures to be included in a new convention:

(1) Empirically based restrictions on the time, duration, and/or area of activities in known biologically important habitats

Many habitats could be included in this measure, such as bowhead whale (*Balaena mysticetus*) feeding areas in the Beaufort and Chukchi Seas; feeding areas, calving areas, and migration routes for North Atlantic right whales (*Eubalaena glacialis*); areas in the North Sea inhabited by acoustically sensitive harbor porpoises (*Phocoena phocoena*) and fish; blue whale (*Balaenoptera musculus*) habitat south of Australia; and important sperm whale (*Physeter macrocephalus*) and Bryde’s whale (*Balaenoptera brydei*) habitats in the Gulf of Mexico. Understanding marine species’ habitat requirements throughout their entire annual cycle in any particular region – relative to the occurrence of seismic surveys conducted therein – is essential. Such biologically important habitats could be proposed as marine pro-

tected areas (MPAs), or, in keeping with the proposal to rely on existing international instruments used by the IMO, they could be designated as areas to be avoided (ATBAs) or as particularly sensitive sea areas (PSSAs). The effectiveness of MPAs, ATBAs, or PSSAs could be minimal, however, given the distances that ocean noise – including seismic survey signals – can travel underwater. Time and/or area restrictions are often a risk-assessment calculation and represent trade-offs. Time-sharing may be impossible when, for example, the waters are available to animals and surveys for only short windows due to the presence of ice (eg Nowacek *et al.* 2013).

(2) Requirements for sustained monitoring of acoustic habitat indicators (eg spatial, seasonal ambient noise levels across species-specific frequencies), with limitations and targets based on the cumulative noise contributions of human activities

Such an approach is consistent with the Marine Strategy Framework Directive currently being implemented in the EU (EU 2008). Monitoring should include data-gathering efforts that improve baseline knowledge about species of concern. This objective can be achieved in part by systematically integrating passive listening capabilities into ocean-observing systems. While most governments have been slow to implement such practices (Southall *et al.* 2012), some examples exist, such as the Australian Ocean Data Network Portal (<http://portal.aodn.org.au/aodn>).

(3) Preconditions to develop and implement practices that reduce the acoustic footprint of seismic surveys and other activities

Any new convention should encourage the development of less invasive exploration techniques, such as vibroseis (a vibratory source that emits more continuous energy, lower in peak energy than airguns and narrower in frequency), which has been used successfully in terrestrial applications for geophysical exploration (Echtler *et al.* 1996) and shows promise in the marine environment (Weilgart 2010). Methods to minimize exploration impacts should also include data-sharing requirements or other reasonable arrangements to help reduce or eliminate duplicative surveys.

(4) Creation of an intergovernmental science organization that can coordinate and advance efforts to improve the environmental assessment of acoustic impacts

Such efforts should include the development and application of metrics for assessing the health of potentially affected organisms and populations, such as quantitative assessments of vital rates, prey availability, ranging pat-

terns, and body condition (eg population consequences of acoustic disturbance [PCAD] or simply “of disturbance” [PCoD]; Harwood *et al.* 2011).

(5) Requirements for the preparation of EIAs and strategic or programmatic environmental assessments that analyze the potential for cumulative effects

Regulators must explicitly assess and manage the risks of additive and synergistic acoustic exposures, which have demonstrated detrimental effects on humans and other mammals (Warner and Heimstra 1971; Evans 2003; Otten *et al.* 2004; Wright *et al.* 2007). Recently developed tools that map human noise and cetacean densities can be used for assessing acoustic impact, cumulative and otherwise (NOAA 2012).

In the absence of an international convention, existing regional authorities may be amenable to incorporating regulation of underwater noise in their regimes. In Europe, ocean noise has occupied the work plans of several regional accords (eg the Agreement on the Conservation of Cetaceans in the Black Sea, Mediterranean Sea and Contiguous Atlantic Area and the Agreement on the Conservation of Small Cetaceans of the Baltic and North Seas, sub-agreements of the Convention on Migratory Species that address cetacean conservation in the Mediterranean, Black, Baltic, and North Seas) for more than a decade, resulting in the first regional noise guidelines (ACCOBAMS 2010), which include seismic exploration. Multinational instruments in other parts of the world that regulate regional seas, such as the Abidjan Convention (UNEP 1984) and the Lima Convention (UNEP 1986), include ocean noise within their “competence” (a term with precise legal implications). Actions taken by these authorities generally constitute “soft laws”, however, which are useful for facilitating regional cooperation and coordination but are not strictly binding for member states and therefore cannot substitute for an international convention. Creating a pan-Arctic plan, given the pace of development and rapid environmental change occurring in the region, is an urgent matter. The intergovernmental Arctic Council (www.arctic-council.org) would be a natural partner in this regard, and could serve as a vehicle for a binding regional treaty.

Elected officials, business leaders, and members of the public, by pressuring governments and industries, possess the influence necessary to encourage sectors engaged in ocean-based commerce to address marine noise. The actions of socially responsible investors and progressive policies put forth by the regulatory community can play a large role in this process. In the near term, indicators of stewardship – such as certification of “ocean friendly” technologies and organizational strategies to limit anthropogenic noise and rates of exposure – should be incorporated by firms engaged in marine resource explo-

ration and extraction, as well as by transboundary cargo-shipping firms. Secondly, a parallel path should be created to incorporate anthropogenic noise into life-cycle (“cradle to grave”) inventory analysis for commercial products (eg Guinee 2002), thus serving as a catalyst for industry to work with researchers in developing more robust spatial and temporal datasets to better quantify the risks and impacts of marine seismic activities and improve the effectiveness of intervention strategies.

At the national level, regulatory engagement can play a critical role in driving innovation. For example, in 2011, the German government issued an action-forcing standard for noise caused by pile-driving activity, requiring operators to reduce pressure levels below a given threshold within a defined radius of the source (Umweltbundesamt 2011). After allowing the industry time to develop technology and methods of compliance, the government now includes the standard in licenses for offshore windfarm construction. Regulators should use analogous mechanisms and other prescriptive and incentive-based tools to promote noise reduction in the seismic exploration industry, which – despite a long-recognized need – has been slow to develop noise-control technologies and alternative technologies and bring them into commercial use. Marine sustainability should be considered as a race to the top, not as a scramble to keep from being the worst.

Ultimately, if we are to understand, control, and reverse the chronic deterioration of marine acoustic environments and its expected impacts on marine ecosystems over the coming decades, regulators should begin establishing cumulative-exposure limits for anthropogenic noise. These limits should be appropriately matched to the spatiotemporal scale and exposure rate of the risks to individuals and populations. The lack of baseline data should not be construed as free rein to proceed unchecked until negative effects are demonstrated. Both public and private institutions have the opportunity to correct our current path and to ensure the resilience and health of marine ecosystems.

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SUPPLEMENT ARTICLE

Using dose–response functions to improve calculations of the impact of anthropogenic noise

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Abstract

1. Estimating the number of animals impacted by a stressor typically involves combining a dose–response function with information about the distribution of animals and of the stressor.
2. Regulators often prefer a single threshold to a full dose–response function, but much of the variability observed in the threshold at which different individuals respond to a stressor is an inherent characteristic of populations that needs to be taken into account to predict the effects of stressors. When selecting an exposure threshold, regulators need information on the proportion of the population that will be protected.
3. Regulatory processes that calculate the number of animals impacted must draw from the dose–response function, the actual distribution of the animals, and a model mapping how the stressor intensity declines with distance from the source. Ignoring any of these factors can lead to significant errors in estimates of the area and numbers of animals affected.
4. This paper focuses on behavioural responses of marine mammals to anthropogenic sound and demonstrates that a common approach of selecting the threshold at which half of the animals respond (RLp50) grossly underestimates the number of animals affected. We present an example, using a published dose–response function, where the number affected is underestimated by a factor of 280. Results would be similar for any stressor whose strength decreases following an inverse-square function as it dilutes into the environment.
5. This paper presents a method to use a dose–response function to derive a more accurate estimate of animals affected and to set a threshold (the Effective Response Level) that corrects the problem with the RLp50 estimate.
6. Estimates of effects of stressors should include estimates of uncertainty, which can be used to adapt thresholds to different policy contexts and conservation problems.

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KEYWORDS

acoustic threshold, behavioural response, behavioural take, dose:response, dose-response, stressor exposure

1 | INTRODUCTION

When regulators want to protect a population from a hazard, they often aim to find a single threshold that constrains the risk to an acceptable level. For example, motorways may have a maximum speed limit, or exposure to a chemical may be limited to a maximum safe dosage. Selection of an appropriate limit often depends on a decision about what level of risk is permissible. For example, noise in US workplaces is regulated (under 29 CFR 1910.95, n.d.) by a permissible noise exposure limit of 90 dBA (a dB scale weighted for human hearing) averaged over an 8 h workday. However, the US National Institute for Occupational Safety and Health recommends a noise exposure limit of 85 dBA because it is associated with an 8% excess risk of developing hearing loss over a 40 year working life, a risk that is preferred to the 25% excess risk expected under the 90 dBA permissible limit (NIOSH, 1998).

Acoustic thresholds are also used to estimate the potential impact of noise on wildlife. Some thresholds are applied with respect to the sound source, and others are applied to the sound level as experienced by the animal. As an example of the source-based approach, the German government aims to protect the hearing of marine mammals by limiting noise from impact pile driving to a single-strike sound exposure level $L_{E,p}$ of 160 dB re $1 \mu\text{Pa}^2\text{s}$ and a peak pressure level $L_{p,pk}$ of 184 dB re $1 \mu\text{Pa}$ measured 750 m away from the sound source (Dähne, Tougaard, Carstensen, Rose, & Nabe-Nielsen, 2017). Most approaches to estimating the impacts of noise focus on the sound as received at the animal as opposed to at the source. Faulkner, Farcas, and Merchant (2018) review the process for environmental impact assessments of noise according to European and US regulations. In most jurisdictions, the environmental impact assessment process assesses environmental risk by comparing the distribution of sensitive receivers with that of the potential hazard. In the case of noise, hearing is used to identify which receivers are sensitive to a particular noise source and the sound field around the source is estimated using propagation models. Thresholds for noise exposure are then selected depending upon the characteristics of the source and receiver, and the relevant regulatory criteria.

In the US, the Marine Mammal Protection Act (MMPA; Marine Mammal Commission, 2015), prohibits the killing, injury and harassment of marine mammals. The US National Marine Fisheries Service (NMFS) has established specific levels of underwater sound exposure that are expected to injure or harass most marine mammals. NMFS (2016) provides acoustic thresholds for effects of noise on hearing, using different sound exposure levels for different taxa and sound types. The US criteria for behavioural harassment are root-mean-square sound pressure levels ($L_{p,rms}$) of 120 dB re $1 \mu\text{Pa}$ for most continuous sounds such as vessel noise and 160 dB re $1 \mu\text{Pa}$ for impulsive sounds such as pile-driving or airguns used in seismic surveys (NOAA

Fisheries West Coast Region, 2018). More sophisticated analyses that weight exposure levels by hearing capabilities of different species and that rank severity of response have been developed in Europe (e.g. Verboom, 2002) and in the US (Southall et al., 2007). These are described in NMFS (2018) technical guidance but have not yet been incorporated into regulations.

Here emphasis is placed on the importance of quantifying variability in responsiveness to sound in order to estimate the number of animals impacted, a topic that has been overlooked in most reviews of environmental impact assessment. Evaluations of the impact of chemical pollutants often use a dose-response function to estimate impact, but evaluations of the effects of noise often use a single number to estimate impact, assuming that no animals are affected below that number and that all animals exposed above that number are affected. Recent environmental assessments of seismic surveys illustrate how regulators in the US use step function thresholds to estimate the number of animals impacted by a sound source. The Bureau of Ocean Energy Management (BOEM) leases offshore areas of the US for energy development and is responsible for assessing the environmental impact of these developments, including seismic surveys. Their environmental impact statements are required to estimate the number of animals taken by killing, by potential for injury (which is called level A harassment in the MMPA) and by disruption of behaviour (which is called level B harassment in the MMPA). BOEM (2014, 4–55) states ‘The NMFS considers behavioral response criteria as a step-function (all-or-none) threshold based solely on the rms value of received levels’ and the threshold likely to cause ‘behavioral disruption for impulsive sounds [Level B harassment] is 160 dB re $1 \mu\text{Pa}$ (rms). For non-impulsive sound sources, such as those associated with vessel traffic, aircraft, and drilling and dredging activities, the sub-injurious threshold is 120 dB re $1 \mu\text{Pa}$ (rms).’ The terminology in this quote refers to root-mean-square sound pressure levels, which we refer to in this paper as $L_{p,rms}$, following ISO (2017). Thus, US regulations require an estimate of the number of animals ‘taken’ by level B harassment, which is defined as a received sound pressure level $L_{p,rms}$ above 160 dB re $1 \mu\text{Pa}$ for impulsive sounds. Calculation of these take estimates by BOEM (2014) uses sophisticated modelling of acoustic sources, sound propagation and marine mammal distribution and abundance. However, the step function criterion assumes that no animals exposed below 160 dB re $1 \mu\text{Pa}$ are impacted and that all animals exposed above 160 dB re $1 \mu\text{Pa}$ are impacted.

The NMFS acoustic criteria for behavioural harassment were based upon studies of reactions of marine mammals to anthropogenic sounds that document a range of received levels associated with response. For example, Malme, Miles, Clark, Tyack, and Bird (1984) generated dose-response functions for avoidance responses of migrating grey whales to continuous noises and impulsive noises

associated with the offshore energy industry. For several different continuous noise sources, avoidance started at received levels $L_{p,rms}$ of 110–119 dB re 1 μ Pa, with >80% of animals responding at received levels $L_{p,rms}$ of 130 dB re 1 μ Pa or more. The received level of continuous sound avoided by 50% of migrating gray whales, a criterion called the RLP50 here, was $L_{p,rms} = 120$ dB re 1 μ Pa. US regulators use this RLP50 of 120 dB re 1 μ Pa as a threshold for level B 'takes' by disruption of behaviour (Green et al., 1994; NOAA Fisheries West Coast Region, 2018). In contrast, much higher levels were required to evoke similar avoidance responses for impulsive noises, which in the Malme et al. (1984) study were generated by air guns used for seismic surveys: 10% of whales avoided exposures of $L_{p,rms} = 164$ dB re 1 μ Pa, with 90% of animals responding at $L_{p,rms} = 180$ dB re 1 μ Pa and an RLP50 of $L_{p,rms} = 170$ dB re 1 μ Pa. Malme, Würsig, Bird, and Tyack (1987) also investigated the response of feeding gray whales to airgun impulses, and found an RLP50 of $L_{p,rms} = 173$ dB re 1 μ Pa (68% confidence limits of $L_{p,rms} = 170$ –175 dB re 1 μ Pa), slightly higher than that for migrating whales. US regulators use a behavioural disruption threshold of $L_{p,rms} = 160$ dB re 1 μ Pa for response to impulsive sounds, which is not only below the RLP50 but is even lower than the 10% probability of avoidance, perhaps because other studies have demonstrated responses of other species at lower received levels (High Energy Seismic Survey, 1999; Richardson, Greene, Malme, & Thomson, 1995).

Methods have recently been developed to estimate probabilistic functions relating acoustic exposure to behavioural responses of marine mammals (e.g. Miller et al., 2014) and to integrate data to estimate dose–response functions from different behavioural response studies (Harris et al., 2015). Here a dose–response function from Miller et al. (2014) is used to illustrate how use of an RLP50 step function, as currently employed for environmental impact assessment, leads to substantial underestimates of how many animals will be impacted. The details of how impact is calculated depends on specifics of the dose–response function and how sound attenuates as it travels through the ocean, but the general point is relevant for estimating impact of exposure to all stressors for which there is variation in sensitivity within the population.

The Miller et al. (2014) dose–response function is used as an example to show how the number of impacted animals can be estimated in a way that accounts for the spatial distribution of the hazard and the subjects, and further we show how an appropriate threshold, which we call the effective received level (ERL), can be calculated. This threshold, when used as if it was a step function, gives the same number of impacted animals for specific sound propagation conditions as would be obtained from the full dose–response function, and so is an appropriate threshold if regulators prefer a single-step function to estimate the number of animals affected by a noise source in a particular site. Sometimes, a full dose–response function is not available, but estimates exist of the proportion of animals responding at various levels of dose. This summary information can be used to give a good approximation to the correct ERL.

BOEM (2017, 26287) explains why take estimates in their environmental assessments ignore known sources of uncertainty: 'confidence

intervals were not developed for the exposure estimate results, in part because calculating confidence limits for numbers of Level B harassment takes would imply a level of quantification and statistical certainty that does not currently exist'. Many of the elements used to estimate takes, including specification of acoustic sources, sound propagation modelling, and estimates of density and abundance of marine mammals, estimate the distribution of values to be expected. New methods have been developed to quantify the uncertainty of the relationship between acoustic dosage and the probability that animals will respond, which enables quantification of uncertainty in estimates of impact. Risk assessments developed by the US Navy (2016; Moretti et al., 2014) and the methods described by Miller et al. (2014) and Harris et al. (2015) all estimate continuous functions of acoustic dosage and the probability of response. Here we discuss how simulation can estimate uncertainty in take estimates using probabilistic dose–response functions and estimates of the distribution of relevant parameters.

2 | EXAMPLE: USING A DOSE–RESPONSE FUNCTION TO ESTIMATE THE NUMBER OF ANIMALS AFFECTED

Estimating the number of animals that would be affected by transmissions of an anthropogenic sound requires combining the relationship between acoustic dosage and the probability of response with the function that predicts how received sound level decreases with range from the source and overlaying this on an estimate of the spatial distribution of animals in the region of interest. Figure 1 shows the

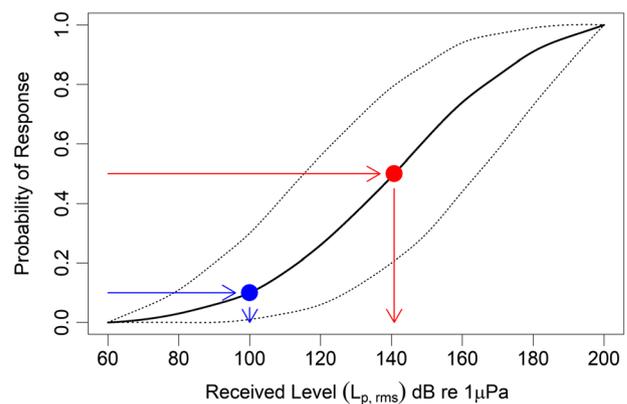


FIGURE 1 Dose–response function derived from experiments performed on free-swimming killer whales exposed to a steadily increasing level of sonar sounds (Miller et al., 2014). The x-axis shows the received level (root-mean-square sound pressure level, $L_{p,rms}$) of sonar sounds, and the y-axis shows the probability of whales responding as a function of received level. The dotted lines show the 95% posterior credible interval, illustrating important uncertainty owing to the small sample of whales in the study. The received level at which the most sensitive 50% of the population are expected to respond (RLP50) for this function is 141 dB re 1 μ Pa, illustrated by the red lines. The received level at which the most sensitive 10% of the population are expected to respond is 100 dB re 1 μ Pa, illustrated by the blue arrows

relationship between acoustic dosage and probability of response estimated by Miller et al. (2014) for avoidance responses of killer whales, *Orcinus orca*, exposed to sonar sounds. The analysis assumed that no whales would respond to sonar below a level of $L_{p,rms} = 60$ dB re 1 μ Pa, which is near the limit of hearing sensitivity of killer whales at this frequency, and that all whales would respond at a received level of $L_{p,rms} = 200$ dB re 1 μ Pa.

The dose–response function shown in Figure 1 uses data from eight controlled exposure experiments to predict the probability of a killer whale showing an avoidance response to received levels of sonar between $L_{p,rms} = 60$ and 200 dB re 1 μ Pa. The blue arrows show that the most sensitive 10% of whales are expected to respond at a received level of $L_{p,rms} = 100$ dB re 1 μ Pa and the red arrows show that half of the whales are expected to respond at a received level of $L_{p,rms} = 141$ dB re 1 μ Pa – i.e. that in this example $RLp50 = L_{p,rms} = 141$ dB re 1 μ Pa.

2.1 | Using the RLp50 threshold greatly underestimates number impacted

To estimate how many whales would be impacted by sonar transmissions, it is necessary to calculate how the intensity of the sonar sound decreases with range from the sound source. For the purposes of our example, the sonar sound is assumed to spread equally in all directions, following an inverse-square $1/r^2$ spherical spreading (where r is the distance from source to receiver). The Miller et al. (2014) dose–response function was developed for sonar signals at 1–2 and 6–7 kHz; statistical modelling provided little support for differentiating response by frequency, so here, when modelling frequency-dependent sound propagation, a nominal frequency of 3 kHz is used, splitting the difference between the two frequencies tested. For a sonar producing a sound source level of $L_S = 210$ dB re 1 μ Pa m at an assumed frequency of 3 kHz, with inverse-square spherical spreading, the received level of sound at range r can be calculated as $L_{p,rms} = 210 - 20\log_{10}(r) - 0.000185 \times r$ (this last term is a frequency-dependent absorption of sound energy; see Appendix for details); this function is plotted in Figure 2. The range at which the received level $L_{p,rms} = RLp50 = 141$ dB re 1 μ Pa is 2.7 km.

For the purposes of this example, the animals are assumed to be distributed evenly through the region of interest, with a density of 1/km² (other spatial distributions are readily incorporated). If better site-specific information is available on sound propagation or animal distributions, these should be used for specific applications. With these assumptions for the purposes of our illustration, the number of animals in an area with the RLp50 radius of 2.7 km is $\pi \times 2.7^2 = 23$. If one assumes, following many regulatory policies, that our threshold represents a step function with no animals affected at lower levels and all animals affected at higher levels, then the estimated number of animals impacted is all 23 of the animals within the 2.7 km radius.

However, this is a gross underestimate of the number of animals impacted. Figure 1 shows that the most sensitive 10% of the population responds to the sonar at a received level of 100 dB, which corresponds to a range of 71 km. At a density of 1 whale/km², 10% of the animals in an area with radius of 71 km is $0.1 \times \pi \times 71^2 = 1584$, a much

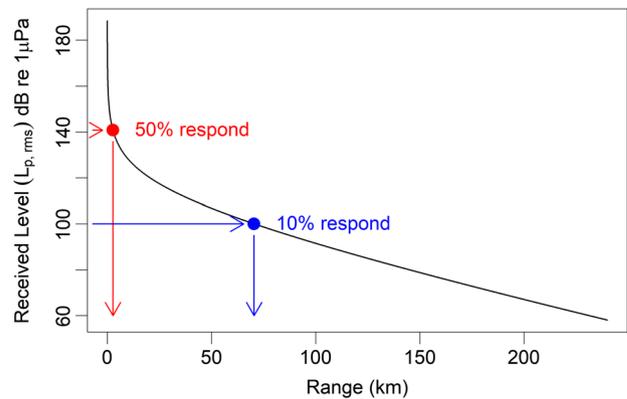


FIGURE 2 Received level as a function of range (distance from the sound source) for a sonar signal with a source level of 210 dB re 1 μ Pa m and a frequency of 3 kHz. Red arrows indicate the 2.7 km range at which 50% of whales with dose–response function shown in Figure 1 are estimated to respond, while blue arrows indicate the 71 km range at which 10% of whales are estimated to respond

higher number than all of the animals at the RLp50 range of 2.7 km. Although the estimated probability of a response at greater distances is very low, there are many more animals at these larger distances.

In fact, more than 10% of animals within radius 71 km can be expected to respond, because the probability of response is higher at closer ranges; some animals farther than 71 km will also respond. Hence even the value of 1584 is an underestimate. A core point of this paper is that sound can propagate so efficiently underwater that noise may cause impacts at greater ranges than is intuitive to humans with experience of sound in air. This can cause a mismatch between regulations and actual effects. For example, the German limitations on source levels of piling as measured at 750 m are designed to protect porpoise hearing at close ranges. However, even with mitigation measures to reduce source level, porpoises showed significant avoidance out to 12 km for up to 5 h after piling stopped (Dähne et al., 2017). Given such pronounced avoidance, habitat exclusion of many animals at large ranges is probably of greater concern than hearing damage of a few animals at close ranges. The US Navy has calculated numbers of takes using methods similar to the ones recommended here but has recently added cut-off distances beyond which they truncate the probability of responses to zero (Navy, 2017). The US Marine Mammal Commission (2017) has pointed out that ‘Including additional cut-off distances contradicts the underlying data of those functions and negates the intent of the functions themselves. The actual cut-off distances used by the Navy also appear to be unsubstantiated.’ Indeed, there are indications of distant responses in the few studies that have observed a reasonable sample size of whales far from sources. For example, Falcone et al. (2017) document statistically significant changes in dive behaviour of Cuvier’s beaked whales (*Ziphius cavirostris*) exposed to sonar as far as 100 km away, well beyond the Navy cutoff of 50 km for beaked whales (NMFS, 2018). Falcone et al. (2017) suggest that their data indicate that the probability of response may be a function of both range and received level. Once quantified,

such interactions can readily be incorporated into the approach presented here.

2.2 | Using the dose–response function to improve estimates of the number of whales impacted by a stressor

To calculate the expected number of animals responding using a dose–response function, we simply multiply the number of animals expected to be at each distance from the source by the probability that these animals will respond. The number of animals at each distance is obtained from our assumption about animal density. The probability of response at each distance is obtained from the dose–response function and sound propagation model. Mathematically, the way to do this accurately is through integration; a simple approach is to divide the area around the source into a large number of equally spaced range bins between zero and the distance at which probability of response becomes equivalent to zero for regulatory purposes, then to calculate the number of expected responses in each bin and to add them up.

Taking the Miller et al. (2014) example, in addition to showing a plot of the dose–response function, the authors provide (in their Table 4) a set of quantiles for probability of response over a range of doses. The current authors fitted a simple smooth curve to these values (a spline-based interpolation – R code given in Supporting Information) and used this to predict the probability of response at the mid-points of a set of 10,000 distance bins from 0 to 240 km (this latter distance being the range at which the received level drops below $L_{p,rms} = 60$ dB re 1 μ Pa and so the probability of response is assumed to be zero), each 24 m wide. For example, the midpoint of the first bin is at 12 m, and the predicted received level at this range is $L_{p,rms} = 210 - 21.59 = 188.41$ dB re 1 μ Pa. From the interpolated dose–response function, the probability of response at this received level is 0.95. The area of this bin is $\pi \times 0.024^2 = 0.0018$ km², hence the expected number of animals is 0.0018 (fractional animals will be the norm given such small bin widths – but rounding must not be done at this stage). Hence the expected number of animals responding in this bin is $0.0018 \times 0.95 = 0.0017$. Similarly, the midpoint of the second bin is at 36 m, the corresponding received level is $L_{p,rms} = 178.87$ dB re 1 μ Pa and the probability of response is 0.90. The area of this bin, which is a ring with inside radius 24 m and outside radius 48 m is $\pi \times 0.048^2 - \pi \times 0.024^2 = 0.0054$ km². Hence, the expected number of animals responding in this bin is $0.0054 \times 0.90 = 0.0049$. Note that this is more animals than the previous bin because, although the probability of response is lower, the area of the bin is greater. Repeating this exercise for all of the bins gives the pattern shown in Figure 3. Overall, the number of animals in each range bin increases linearly with range (shown by the dashed line in Figure 3), because the area of the rings around the source corresponding to each range bin increases linearly with range. The number of animals responding (shown by the solid black line) increases initially with range, but drops away to zero as probability of response falls to zero at just before 240 km. Summing over all range bins gives an expected

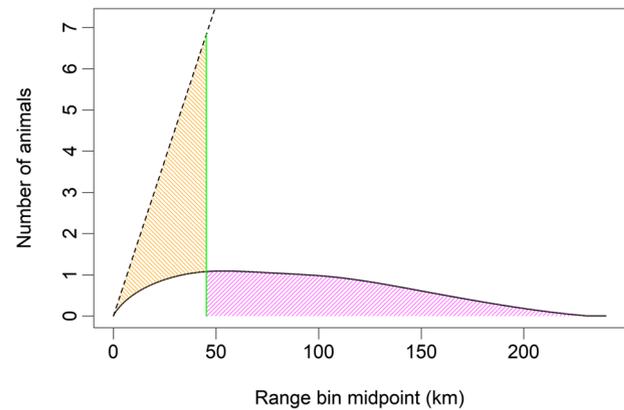


FIGURE 3 The number of animals expected to respond to sonar as a function of distance from the sound source. The solid black line shows the number of animals expected to respond to sonar in each of 10,000 equally spaced range bins from 0 to 240 km. This is calculated by multiplying the number of animals expected to be in each range bin, shown by the dashed black line, by the probability that each animal at that range will respond (derived from the dose–response function shown in Figure 1 and the received level to range conversion shown in Figure 2). Also shown, as a vertical green line, is the effective response radius (ERR), i.e. the range at which as many animals are expected to respond beyond that distance from the source (denoted by the magenta polygon) as do not respond within that distance (orange polygon, of equal area to the magenta polygon)

response of 6437 animals. This is 280 times higher than the estimate of 23 derived by assuming a threshold in response at the RLp50 range of 2.7 km.

These calculations require that researchers provide enough information to enable the probability of response to be calculated for any given acoustic dose. Miller et al. (2014) provided a table of quantiles that we used for this purpose. Malme et al. (1984) similarly tabulated the received levels and ranges at which different proportions of grey whales would be expected to avoid airguns. As a useful alternative, Moretti et al. (2014) provide a parametric equation that closely approximates the dose–response function they fitted for cessation of feeding dives in Blainville's beaked whales as a function of received sonar level. This enables probability of response to be calculated at any desired level of dose using, for example, a simple spreadsheet.

2.3 | Calculating a single threshold value that yields the same effect as the dose–response function: the effective received level

The dose–response function provides the basis for estimates of the number of animals affected by an anthropogenic sound source, but if regulators in some jurisdictions prefer an effective radius or an acoustic criterion that is just one single number, then it is possible to combine information from the dose–response function, sound source level and models of acoustic propagation and animal distribution to calculate these values for each specific case. One way to conceptualize this effective radius is to start with the estimates derived in the previous section of the number of animals expected to respond in

each distance band, and to calculate the range at which as many animals respond beyond this range as fail to respond within it (Figure 3). Then, by definition, the number of animals (responding or not) within this range is exactly equal to the total number of animals responding. We term this range the effective response range (ERR), after a similar concept used in point transect surveys of wildlife populations (Buckland et al., 2001). This is readily translated, via the propagation model, into an estimate of the corresponding received level of sound, the effective received level (ERL).

This concept is further illustrated graphically in Figure 4, using simulated animal positions. The left panel of Figure 4 shows a simulated distribution of animals, with those responding indicated in red and those not responding in black. The right panel shows the distribution if each red point outside of the effective radius is moved to replace a black point inside the radius. The ERR is the radius that encompasses an area including the total number of animals estimated to be impacted. In our case, the total number of 6437 animals corresponds to an area of 6437 km² at a density of 1 animal/km². The ERR for this area is 45.3 km, which corresponds to an ERL of $L_{p,rms} = 109$ dB re 1 μ Pa.

Note that, in general, an assumption about absolute animal density is not required to calculate the ERR or ERL. The ERR and ERL given will be identical for assumed densities of 1, 10 or 100 animals/km² (or any other value). We do require an assumption about the spatial distribution of animals around the source, and in general (in the absence of other information) the assumption is made that animals are uniformly distributed around the source. An estimate of density is, however, required to estimate the absolute number of animals impacted: this number is simply $\pi \times \text{ERR}^2 \times \text{density}$. In cases where information is available to estimate non-uniform distribution around the source, then the information about animal distribution is required to calculate ERR and ERL.

2.4 | Quantifying uncertainty

A variety of sources of uncertainty affect our ability to estimate the impact of a stressor on a population. This process starts with estimating the distributions of affected animals and of strength of the stressor, with the dose–response function linking the two to estimate the number of animals impacted. Uncertainty about the dose–response function, about the density and distribution of animals, and about the sound field in the impact zone are important for all such problems. Analysis of the distribution of underwater noise as a stressor requires knowledge about the uncertainty related to the directionality, variability and level of the sound source, and to how sound will propagate through the ocean from the source. The navies of the world have developed sophisticated models of sound propagation in the ocean because this information is critical for estimating how their sonars will perform. This means that there is usually more certainty about sound propagation than about the animal distribution and the dose–response function, which therefore become the greatest sources of uncertainty. Given the importance of estimating uncertainty to make decisions about acceptable risk, it is striking how few environmental assessments of acoustic impacts on marine mammals use this information. This section describes methods to estimate overall uncertainty about the number of animals expected to be impacted by a proposed action.

The primary obstacle to analysing uncertainty with respect to estimating the number of animals impacted by sound that is produced by a human activity has stemmed from the dose–response function. Most activities that generate sound in the ocean are able to specify variation in features of the sound that is produced. Similarly, models and measurements of sound propagation in the ocean can be used to quantify uncertainty in the level received by an animal some distance from the sound source. Biologists who estimate the sizes of wildlife populations

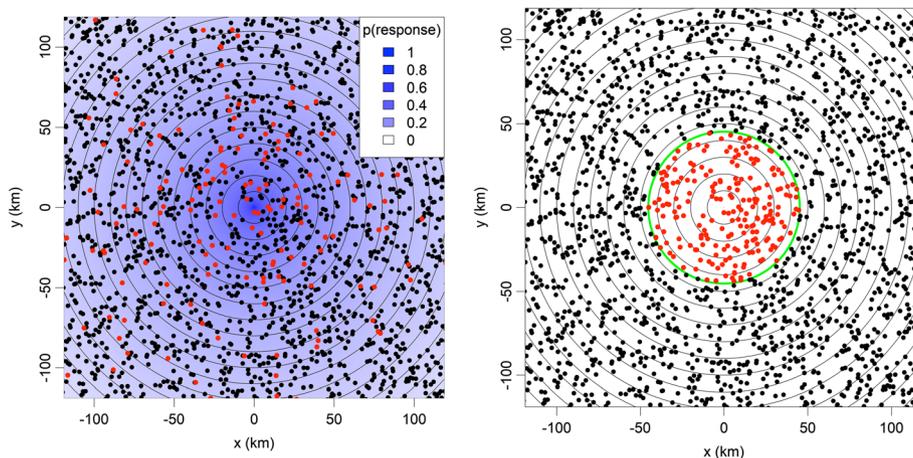


FIGURE 4 Conceptual illustration of the process for calculating an effective response range or effective received level to predict the number of animals impacted by a sound source in an environment with known properties of sound propagation. The left panel shows a simulated distribution with animals that respond indicated in red and animals that do not respond indicated in black. The intensity of the blue background scales to the probability of response at that distance from the sound source. In the right panel, each red point outside of the effective radius (the green circle) is moved to replace a black point inside the radius. The effective response range is the radius that encompasses an area including the total number of animals estimated to be impacted. In our example, the total number of 6437 animals corresponds to an area of 6437 km² at a density of 1 animal/km². The radius of a circle with this area, the effective response range (ERR) is 45.3 km, which corresponds to an effective received level (ERL) of 109 dB re 1 μ Pa

are usually very disciplined in calculating uncertainty in their estimates. The same agencies that ignore uncertainty in estimating takes recognize the critical importance of incorporating uncertainty in other management models. For example, the protocol used by NMFS to calculate an allowable mortality of marine mammals caused by humans uses a minimum population estimate defined as the lower 20th percentile of the estimated abundance distribution (Wade, 1998). Taylor, Wade, De Master, and Barlow (2000) used simulations to show that using the best estimate of population size resulted in many populations being unacceptably depleted, while use of the 20th percentile of the population estimate prevented most unacceptable outcomes.

The methods developed to derive probabilistic dose–response functions (e.g. Miller et al., 2014) make it possible to quantify uncertainty about dose–response. There are simple ways to calculate the effect of uncertainty in the dose–response function alone. For example, the dotted lines in Figure 1 indicate the 95% credible interval (the Bayesian analogue to a confidence interval) for the function relating killer whale avoidance to received levels of sonar sound. By repeating the calculation described in Section 2.2, using the 2.5 and 97.5% quantiles from Table 4 of Miller et al. (2014), rather than the mean estimate for probability of response, one can calculate a 95% interval on the expected number of animals impacted, which is 548 to 20541.

In fact, this is a slight over-estimate of the uncertainty arising from the dose–response function in this example, for a technical reason: the dotted lines in Figure 1 are *pointwise* credible intervals, i.e. they show uncertainty in probability of response for a given dose. What is actually required is a credible interval on the whole function, which will be narrower. This was not given by Miller et al. (2014), but here we calculated 1000 replicate dose–response functions sampled from the posterior distribution on their model parameters, and this was used to calculate a 95% interval on the expected number of animals impacted (code and data provided in Supporting Information) of 733–20111. In general, 95% uncertainty intervals should be provided by researchers analysing the ERL – this is readily converted into a 95% interval on numbers impacted, given a model of sound propagation and animal density (in the current example, this interval is $L_{p,rms} = 97.3\text{--}123.5$ dB re 1 μ Pa).

These intervals account only for uncertainty in the dose–response function, where in reality there are other sources of uncertainty, probably the most important of which is animal density in the impact zone. In general, where the uncertainties have been quantified, multiple sources of uncertainty can be readily combined by researchers to estimate resulting uncertainty in the numbers affected using a Monte Carlo simulation approach. A random sample is drawn from the distribution of dose–response functions, animal density, etc., and the resulting estimated number impacted is computed. This process is repeated many times, to give a distribution on the estimated number affected.

3 | DISCUSSION

This paper describes how use of a step function to define the relationship between exposure and response of wildlife to a stressor can lead

to errors in estimating the impact of the stressor if variability in responsiveness within the population is not taken into account. Newly developed methods to quantify probabilistic functions that relate acoustic dosage to behavioural response (e.g. Harris et al., 2015; Miller et al., 2014) show how prior information coupled with relatively low sample sizes of controlled experiments can be used to define probabilistic dose–response functions. These functions can be combined with site-specific information about sound propagation and animal distribution to estimate the number of animals likely to be affected by a human activity that introduces sound into the ocean.

Much of the variability observed in the threshold at which different individuals respond to a stressor is not measurement error but is an inherent characteristic of populations that needs to be taken into account to predict the effects of stressors. Every population of organisms will be expected to show variation in sensitivity to any stressor. We know that disruption of behaviour by sound depends on the characteristics of the sound and the hearing sensitivity of each animal, and the likelihood of disruption often depends upon the age/sex class of the animal, its experience with similar sounds, and the behavioural context in which it hears the sound (Ellison, Southall, Clark, & Frankel, 2012). All of these factors lead us to expect considerable variability in responsiveness across a wildlife population, which in fact has been observed by most studies on this topic.

3.1 | Dose–response

Ellison et al. (2012) review evidence that the context in which an animal is exposed to a sound can strongly affect the probability or the severity of a behavioural response. This leads them to argue that dose–response functions should only be used to predict the probability of response at high sound levels, with multivariate contextual variables being used at low sound levels either to replace acoustic exposure as a predictor for probability of response (Ellison et al., 2012, Figure 2) or in a weighted combination with acoustic exposure. It is not obvious how a management approach that ignores the dosage of sound, especially at low exposure levels, can predict the number of animals likely to be impacted. There may be some circumstances where regulators may choose to prohibit a sound source or activity within detection range of a wildlife population engaged in a specific activity (such as breeding), either because the population is particularly sensitive at that time and place or because disruption of behaviour would be likely to lead to unacceptable population impacts. This approach would be particularly difficult for intense low-frequency sound sources that can routinely be detected hundreds of kilometres away (e.g. Nieuwkirk et al., 2012). In settings where it is not possible to prevent overlap of a stressor and the affected population, it is essential to use dose–response functions coupled with estimates of intensity of exposure for individuals to estimate the number of animals impacted by the stressor. The practicality of using full dose–response functions to estimate takes is demonstrated by the long-standing use for over a decade of a sonar risk continuum function in environmental impact statements that evaluate the effect of naval sonar on marine mammals (Navy, 2002, 2016).

A significant benefit of functions that relate the probability of response to acoustic exposure is that they enable the selection of a probability of response that is appropriate for each specific policy context. In contrast, the use of a single threshold, such as the RLP50, hinders this calibration of risk in terms of the proportion of the population that is impacted. Malme et al. (1984) selected the RLP50 avoidance value 'rather than the customary 0.95 level since the 0.95 level is not adequately defined by the available data'. This may be reasonable from a scientific perspective, but limiting the focus to RLP50 to estimate the number of takes not only prevents the correct calculation of impact, but also narrows the criterion to a value that may be inappropriate for many regulatory functions. The acceptable percentage of animals impacted depends upon the policy context. For example, Norwegian support for the Miller et al. (2014) study was motivated by concerns expressed by a whale watch industry that Norwegian naval exercises caused killer whales to vacate the whale watch area, harming whale watch companies (Kuningas, Kvadsheim, Lam, & Miller, 2013). In this case, maintaining half of the whales available for whale watching might meet the needs of the industry. In contrast, the southern resident population of killer whales in Puget Sound is listed as endangered under the US Endangered Species Act, in part because of the risk of behavioural disruption by anthropogenic noise (Krahn et al., 2004; NMFS, 2011). Here it is unlikely that regulators would select an RLP50 threshold of impact that allowed half of the animals exposed above the threshold level to be adversely impacted. Similarly, acoustic criteria are used by many regulators to establish shut-down zones – an area around a sound source where the source must be shut down if animals are sighted within it to prevent them being harmed. If such a shutdown zone were established using an RLP50 based upon hearing damage, then the shutdown would only protect the least sensitive half of the population. There are few jurisdictions that would accept protective criteria that allow half of the population to be harmed even when exposure is limited to below the threshold level.

Use of the Miller et al. (2014) dose–response function to estimate how many animals are likely to be affected by sound at various distances from the source emphasizes that large numbers of animals are likely to be affected by exposure at long ranges from the source. At ranges close enough for the probability of response to be high, the area may be small enough that few animals are likely to be affected. At long ranges where the probability of response is low, if the area affected is large enough, then large numbers of animals may be affected because the small probability is multiplied by the large area. Many behavioural response studies have emphasized providing exposures with received levels high enough for high probability of response, but our analysis here emphasizes the importance of quantifying probability of response at low levels of exposure far from the source where the probability of response is relatively low. Such studies will require larger sample sizes to quantify low probabilities of response. Achieving the necessary sample sizes may be facilitated by tagging a large number of animals at varying ranges from the source and/or passive acoustic monitoring of vocal responses of many animals over large areas. The availability of tags that can measure exposure and

response over long periods of time would facilitate monitoring responses to operational use of loud sources if animals can be tagged far enough in advance of sound transmission to quantify pre-exposure behaviour, and then can log exposure and potential responses.

3.2 | Selection of appropriate exposure and response measures

An important aspect of studying dose–response functions is selecting appropriate exposure and response measures (Ellison et al., 2012; Madsen, 2005). As in toxicology, the selection of response measures depends upon a combination of science, policy and regulations. The key for estimating takes by level B harassment under the MMPA is to define responses that cross the threshold of evoking prohibited disturbance. Responses where a subject turns away to avoid exposure to a sound are often treated as a disturbance reaction (e.g. Malme et al., 1984). Avoidance responses are also relevant in other jurisdictions, especially if they involve shifts in distribution over large scales of time and space. For example, a study in Norwegian waters focused on avoidance responses after whale watching companies complained that naval sonar exercises caused a decline in killer whale sightings, harming the industry (Kuningas et al., 2013). Responses treated as disturbance also include cases where exposure to sound causes a subject to switch from one behavioural state such as foraging to another behaviour such as travelling (e.g. Goldbogen et al., 2013; Isojunno et al., 2016), and NMFS even defines specific behavioural events, such as breaching, tail lobbing, underwater exhalation or an animal leaving its group, as strong adverse reactions to human activities (NMFS, 2007). Recent efforts to estimate the population consequences of acoustic disturbance (Pirota et al., 2018) provide models to help decide which changes in behaviour may reach a threshold appropriate to trigger regulations that are driven by effects on populations. Important parameters to measure in these cases involve the energetic cost of response and the time required for a return to pre-exposure baseline conditions.

The appropriate exposure measure depends on the response being studied. For example, extensive studies on the sound exposures required to reduce hearing sensitivity (temporary threshold shift or TTS) suggest that to a first approximation best predictor is either a very high peak pressure level or the cumulative dose of sound energy (Southall et al., 2007). Most studies on what sounds might disturb a marine mammal have tended to measure the received level of individual sounds, expressed as a root-mean-square or RMS sound pressure level $L_{p,rms}$, as this can be measured directly (Southall et al., 2007; see Madsen, 2005 for issues concerning RMS measures for transient signals). However, as Ellison et al. (2012) point out, it is often useful to include additional acoustic measures for predicting probability of response. The annoyance value of a loud sound may relate to how much louder it is than a subject's hearing sensitivity. This difference, called the sensation level, is also helpful for estimating how faint a signal a subject can detect in quiet conditions. The sensation level is also used for predicting onset of a

specific response called the acoustic startle response that is shared among mammals. This aversive response is triggered in mammals by intense sounds with a sensation level >90 dB that have a rise time of 15 ms (Yeomans, Li, Scott, & Frankland, 2002; in marine mammals Götz & Janik, 2011).

In cases where the hearing sensitivity of subject species at the frequencies of an anthropogenic noise is known, audiograms can be used to calculate sensation level, which can be incorporated into dose–response studies. Ellison et al. (2012) argue that measurements of behavioural responses ‘invalidate the use of an absolute, dose–response RL approach’. However, selection of exposure measures, such as sensation level, that require audiometric measurements is problematic for species such as baleen whales with no measurements of hearing sensitivity. For species with some measurements of hearing sensitivity, the use of the sensation level will add new sources of uncertainty if information about variability in hearing sensitivity within a population is incomplete, especially if the subjects whose hearing has been measured might come from a biased sample with abnormal hearing, for example owing to injuries related to stranding. Information about hearing can be incorporated into Bayesian analyses in other ways. For example, Miller et al. (2014) assume a zero probability of response for received levels lower than the whales could hear, enabling this hearing threshold to be included in their Bayesian analyses, which used $L_{p,rms}$ as a response parameter.

For a marine mammal to detect an anthropogenic sound, the animal's hearing must be sensitive enough at the frequency of the sound and the sound must have enough energy above the ambient noise at that frequency. The hearing of marine mammals is very acute, but if a noise source lies outside the frequencies of best hearing, a marine mammal might not be able to hear it. For example, the noise generated by offshore windmills is far enough below the frequency of best hearing for bottlenose dolphins (45 kHz, Popov et al., 2007) that they would not detect windmill noise below 1 kHz recorded in a variety of shallow water habitats (Madsen, Wahlberg, Tougaard, Lucke, & Tyack, 2006). Southall et al. (2007) address these issues by pooling marine mammals into species groups defined by hearing capabilities, and they develop weighting functions to discount sound energy at frequencies the animals are estimated not to hear well. Weighting the levels of the sound stimulus by these functions makes it possible to estimate the sound energy that an animal is likely to hear, even for species without audiometric data.

For the many marine mammal species whose hearing sensitivity has not been measured, most analyses would have to assume that ambient noise limits their ability to detect acoustic signals. For analyses of noise-limited detection ranges, measurement of the noise level at the frequencies of the anthropogenic sound of concern is essential for estimating signal levels below which a subject is unlikely to respond. The signal-to-noise ratio is a critical parameter for this estimate, which requires estimates of the frequency bands over which the subjects' ears integrate acoustic energy. Most mammalian auditory systems integrate sound energy over about a third of an octave, so this is commonly assumed. It is important to note that the bandwidth

over which noise should be integrated is a critical parameter for estimating range of effect.

In addition to the frequency range of signals, their duration is also important for defining acoustic parameters of different stimulus types. The time window over which the auditory system integrates sound energy is important for estimating the perceived loudness of signals of different durations. Analysis of this integration time for marine mammal ears suggests use of a 125–200 ms window for estimating $L_{p,rms}$ values, even for signals with longer durations, along with longer time windows for cumulative sound exposure measures such as L_E (Madsen, 2005; Tougaard, Wright, & Madsen, 2015).

3.3 | Uncertainty

In addition to dealing with inherent variability within populations, there is considerable uncertainty about many of the estimates used to predict impact of human activities. Many jurisdictions adhere to precautionary regulations, which require regulators to be more conservative the less they know (Foster, Vecchia, & Repacholi, 2000). Methods to quantify uncertainty help regulators to meet the legal demands of underlying legislation that calls for such precaution. We advocate the use of Monte Carlo simulations to estimate the expected distribution of number of animals impacted based upon distributions of all of the factors that affect this.

An important source of uncertainty derives from the necessity of extrapolating dose–response functions from species that have been studied to those that have not been studied. Southall et al. (2007) solved this problem by pooling marine mammals into groups thought to have similar hearing. However, enough evidence of heightened sensitivity of beaked whales has caused NMFS to suggest a different acoustic criterion for harassment of beaked whales compared with the other members of the Southall et al. (2007) mid-frequency hearing group for cetaceans. This suggests that selecting appropriate dose–response functions for poorly studied species can be problematic (Gomez et al., 2016).

Outside of the dose–response function, a major source of uncertainty in estimated impact is often due to uncertainty in the animal density within the impact zone. One potential method to make more accurate predictions of animal density is through habitat modelling of survey data (e.g. Roberts et al., 2016) – this can be thought of as the animal density equivalent of context modelling of dose–response functions or location-specific acoustic propagation models: all of these seek to make more accurate predictions by better understanding the factors that cause variation, where these factors can be known in the time and place for which impact is to be estimated. However, as noted by Roberts et al. (2016, in their Supplementary Materials), procedures to quantify uncertainty in modelled density surfaces are currently less well developed than in acoustic propagation models.

Faced with all of this uncertainty, the reader may be tempted to go back to the use of simple thresholds, such as the RLp50 range. We argue to the contrary that coupling simple models for animal density and acoustic propagation with a dose–response function will yield a

much more realistic answer than using a single RLP50 threshold. As has been shown, the RLP50 can lead to greater than two orders of magnitude underestimation of effect, much more error than expected from simple models of animal density and acoustic propagation. The use of simplified models of uniform animal distribution and uniform sound propagation is a well-established first-order approximation that yields reasonable estimates if more precise information is not available. The RLP50 calculation on the other hand is biased and will yield incorrect estimates for the propagation model and dose–response functions selected as reasonable examples here.

4 | CONCLUSIONS

The dose–response functions discussed in this paper are more complicated to describe and to apply than the single-value-step functions that are common in today's regulations. This complexity is necessary to avoid errors in estimating the number of animals impacted, but some readers may still question whether the complexity is essential for correct implementation of policy. Once the necessary information is available, a new step function – the effective received level – is defined here to better estimate the number of animals impacted. It is important to emphasize that nearly all of the other parameters essential for evaluating impact, parameters that include (a) specifying the acoustic properties of sources, (b) how sound propagates and (c) estimating the distribution and abundance of affected animals, require quantitative analytical procedures that are at least as complex as those described here for dose–response functions. Therefore, the primary complication introduced by this approach is to force explicit quantitative judgments about risk and uncertainty about the proportion of a population that is impacted. These kinds of judgments are routine in acoustic source specifications, sound propagation modelling and population estimation. Surely the protection of species at risk deserves the same level of attention. This paper highlights the importance for conservation of not just accounting for high probabilities of impact on a few animals very near a sound source. Given the shape of the dose–response function and how efficiently sound propagates in the ocean, the number of animals whose exposure level predicts low probability of response may be the dominant impact of the sound source.

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section at the end of the article.

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APPENDIX A

PASSIVE SONAR EQUATION

Acoustic calculations are usually carried out in a logarithmic scale called the decibel, abbreviated to dB. The root-mean-square (RMS) received sound pressure level ($L_{p,rms}$) of a sound in dB is equal to $20 \log_{10} ((\bar{p}^2)^{1/2}/p_{ref})$, where \bar{p}^2 is the mean-square sound pressure and p_{ref} is the reference pressure underwater = 10^{-6} Pascal or 1 μ Pa. The intensity of a sound source is called the source level (L_S) and is evaluated with respect to a reference range of 1 m and is expressed as dB re 1 μ Pa m. When a sound spreads equally in all directions, following an inverse-square $1/r^2$ function, the spherical spreading loss in sound energy as a sound passes from 1 to r metres is equal to $20 \log_{10}(r)$. Some sound energy is also absorbed as it passes through the ocean. This absorption loss depends on the frequency of the sound. Here we will assume a sonar sound operating at a frequency of 3 kHz, which has an absorption loss of 0.000185 dB/m in normal sea conditions (Ainslie & McColm, 1998). The overall loss of sound energy as a sound passes from 1 to r metres, called the propagation loss or PL, is the sum of the spreading loss, $20 \log_{10}(r)$, and the absorption loss $r * 0.000185$. The equation is therefore $PL = 20 \log_{10}(r) + r * 0.000185$.

The passive sonar equation is used to estimate the loudness of a sound received at range r from a sound source with a source level of L_S (Urick, 2013). This equation simply states that the level received at range r equals the source level measured at 1 m minus the loss in energy as the sound travels from 1 to r metres: the received level $L_{p,rms} = L_S - PL$. So, for a sound source of $L_S = 210$ dB re 1 μ Pa m transmitting in an environment with the PL described above, the received level = $L_S - PL$ translates to $L_{p,rms} = 210 - 20 \log(r) - r * 0.000185$.

Automatic Round-the-Clock Detection of Whales for Mitigation from Underwater Noise Impacts

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Abstract

Loud hydroacoustic sources, such as naval mid-frequency sonars or airguns for marine geophysical prospecting, have been increasingly criticized for their possible negative effects on marine mammals and were implicated in several whale stranding events. Competent authorities now regularly request the implementation of mitigation measures, including the shut-down of acoustic sources when marine mammals are sighted within a predefined exclusion zone. Commonly, ship-based marine mammal observers (MMOs) are employed to visually monitor this zone. This approach is personnel-intensive and not applicable during night time, even though most hydroacoustic activities run day and night. This study describes and evaluates an automatic, ship-based, thermographic whale detection system that continuously scans the ship's environs for whale blows. Its performance is independent of daylight and exhibits an almost uniform, omnidirectional detection probability within a radius of 5 km. It outperforms alerted observers in terms of number of detected blows and ship-whale encounters. Our results demonstrate that thermal imaging can be used for reliable and continuous marine mammal protection.

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Competing Interests: The authors declare competing financial interests: 1) Pending patent application, AWI2011/09/01US, AWI2011/09/01DE: A method for automatic real-time marine mammal detection. The patent describes the ideas basic to the automatic whale detection software as used to acquire and process the data presented in this paper (<http://www.freepatentsonline.com/y2013/0070079.html>). 2) Licensing of the Tashtego automatic whale detection software to the manufacturer of IR sensor. The authors confirm that these competing financial interests do not alter their adherence to all PLOS ONE policies on sharing data and materials, as detailed online in the guide for authors.

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Introduction

Growing concerns that aquatic noise produced during naval exercises and offshore seismic surveys by the oil and gas industry may be harmful to marine mammals [1,2], have led an increasing number of regulating agencies to request mitigation measures when issuing permits for such surveys in their nations' EEZ [3]. The most common measure is to implement a "marine mammal watch", a team of observers that scans the ship's environs for signs of presence of marine mammals to trigger a shutdown of the hydroacoustic source when marine mammals are entering a predefined *exclusion zone*.

Marine mammal observers usually scan the ship's environs for whales using binoculars or the naked eye. Sightings mostly rely on spotting a whale's blow, which might rise to a height of several meters but is visible for a few seconds only. Hence, in combination with the whales' prolonged dives, sighting opportunities are rare, which, in addition to the limited field of view and finite attention span of human observers, renders this method personnel-intensive and difficult, even during fair weather and daytime. During darkness it is not feasible.

Use of infrared (IR), i.e. thermal imaging, has been suggested for night-time detection of whales [4]. In thermal imagery, a whale's blow stands out as a transient, warm feature, at least in front of cold surface waters [5]. However, up to now ship-based IR

technology has been unsuitable for detecting whales beyond distances of 150 m. Longer ranges required stable, land-based platforms [6] with tele-optics for enhanced resolution while the field of view was limited to angular segments of 45° or less. Most importantly, detections relied on (retrospective) human screening of the images, which is similarly tedious and error-prone as direct visual observation. Moreover, for mitigation purposes, observations need to cover much of the horizon and to be conducted continuously for weeks to months. Such a mode of operation requires automatic detection capabilities, which are introduced and validated in this paper. However, the system described herein is not intended to operate in an unsupervised mode, but to reliably alert a marine mammal observer about the likely occurrence of any whale blow in the ship's environs, while facilitating its immediate verification and documentation.

Materials and Methods

The infrared detection system consists of a thermal imaging device (FIRST-Navy) mounted on an actively stabilized gimbal (both by Rheinmetall Defence Electronics, Germany) in combination with a custom data acquisition and processing software (Tashtego, <http://tashtego.org>). The cryogenic sensor is cooled to 84 K using a Sterling cooler. It scans 360° horizontal × 18° vertical at 5 revolutions per second, providing a 5-Hz video stream of the

thermal field of the ship's environs at horizontal and vertical resolutions of $0.05^\circ/\text{pixel}$ and $0.03^\circ/\text{pixel}$, respectively. The sensor is installed 28.5 m above the sea surface on-board *R/V Polarstern* and was deployed for a total of 280 days during 7 expeditions to the Arctic and the Southern Ocean. All expeditions to the Southern Ocean were conducted under Permits from the German Environmental Agency with following identification codes: I 3.5 - 94003-3/218; I 3.5 - 94003-3/238; I 3.5 - 94003-3/247; I 3.5 - 94003-3/278; I 3.5 - 94003-3/273. Expeditions to the Arctic and Atlantic did not require permitting from ethical committees, since the technology used is strictly passive (i.e. observational).

Ship-whale-distances are calculated by spherical triangulation [7] using the angle below the horizon (resolved to $\pm 0.05^\circ$), providing unbiased ranges better than 12% accuracy (at 5 km) of the ship-blow distance, i.e. better than achievable by the use of handheld binoculars (Figure S1). Detailed geo-referenced maps of ship-whale encounters are derived in conjunction with bearing information (available to within 0.1°) and the ship's navigational data, allowing for inferences on the whales' behavioral response, respiration rates, and dive cycles as exemplified in Figure 1.

To develop an automatic detection system for whale blows from thermal images, we started with retrospective human screening of thermographic video recordings from multiple expeditions to extract a set of sample blows. On this basis, an automatic detection algorithm was designed to detect temporal contrast changes identified as whale blows (Figure 2) in a standard *detector/classifier* approach, using multi-scale sliding windows [8–11]. The *detector* identifies significant thermal anomalies using a modified short-term-average/long-term-average algorithm (STA/LTA) [12] (Figure S2), the *detector* identifies significant thermal anomalies which are then classified as a *blow* or a *no-blow* event. Computer classification of pertinent video snippets is performed after reduction of dimensionality through spatial and temporal centering and clipping through an Eigenimage algorithm [13,14] before applying a predetermined SVM-based (Support Vector Machine) *classification model* [15]. Training of the SVM is conducted under supervised learning from 120 manually validated *blow* and 1400 *no-blow* events selected from a period of 21 days from expedition

ANT-27.2 [16], covering different environmental conditions, distances, and whale species.

Comparisons at the encounter level were based on visual sighting data collected by MMOs conducting continuous transect counts [17] during a 31-day long expedition, ANT-28.2 [18], from Cape Town to Antarctica and back. Observers recorded sighting time and, if possible, species, but not distance. Observations were conducted from the ship's bridge for a total on-effort time of 299 hours, with 34 ship-whale encounters logged. For 3 of these encounters, the IR system was not operational, and for another 5 encounters, sighting records lacked time information of adequate precision (i.e. to the minute), resulting in a total of 26 visual sightings suitable for comparison with the automatic detection system. Concurrent (within ± 10 minutes) visual and IR encounters were considered *detected encounters*, all other *missed encounters*.

To study the impact of distance on the detection algorithm's performance, dedicated cue-based comparisons were conducted on the basis of two periods of IR recordings of 50 and 60 min duration collected on 13 and 16 January 2011 during expedition ANT-27.2. Concurrent visual observations provided 303 to-the-second recordings of whale blows, which allowed us to match blows from visual observations with blows from IR recordings. From the IR images, we also determined direction and distance of each automatically detected blow.

An automatic thermal detection was considered a *true positive* if it occurred 3 seconds prior or after a "concurrent" visual cue, or if it was unambiguously validated by retrospective human screening of the IR footage. The latter criterion is indispensable to properly classify blows that were missed by the observer. *False negatives* (events missed by the detection algorithm) were attributed to visual sighting records that lacked matching automatic detections within ± 3 s of the sighting. Blows overlooked by both human observers as well as the automatic detector are (unavoidably) left unconsidered. When multiple blows occurred within 1 s, the observers could only record one. Hence, in favor of doubt, all automatic thermal detections within that single second were counted as observed by the MMO.

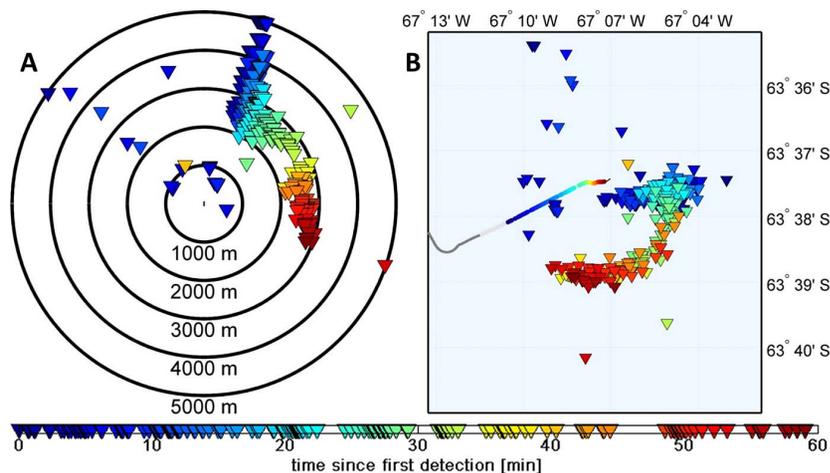


Figure 1. Mapping of blows by a pod of humpback whales. A: Whale blow locations (triangles) relative to the (moving) ship (ship is at center of concentric circles). B: Map of corresponding geo-referenced ship positions (dots) and blow locations (triangles). Color indicates time after first detection. D.

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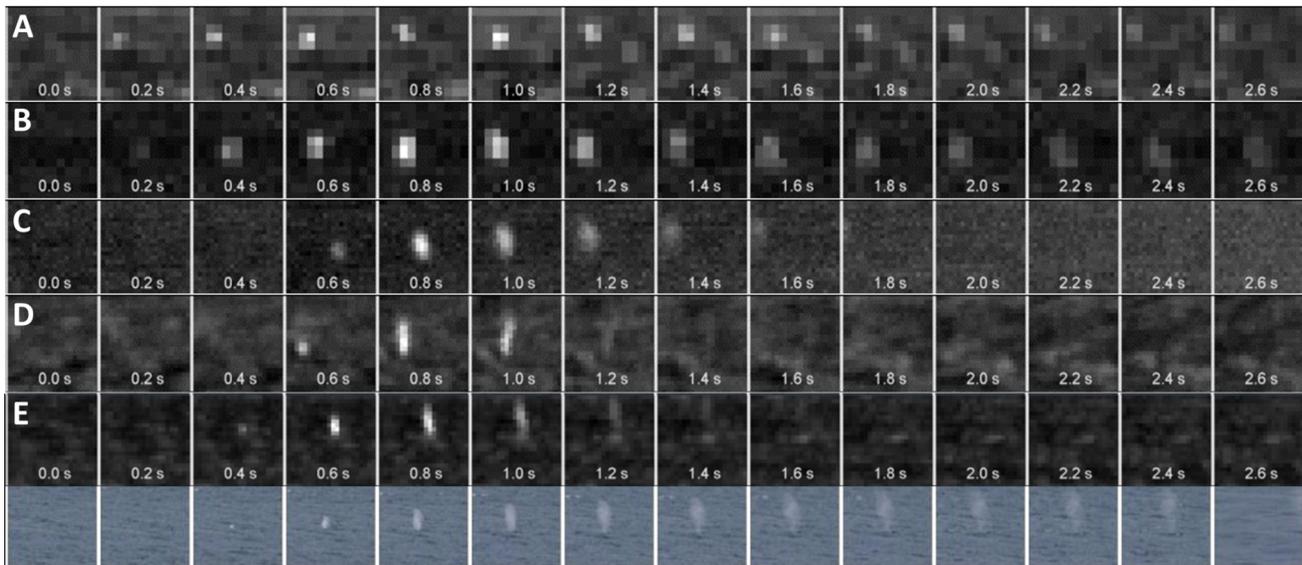


Figure 2. Night-time thermographic video snippets (at 0.2 s resolution) of whale blows: A) 24.03.2012 00:07; 61.11°S 56.36°W; $T_{\text{water}} = 1.3^{\circ}\text{C}$; $T_{\text{air}} = -1.7^{\circ}\text{C}$; $r = 3608$ m; B) 28.03.2012 03:27; 61.88°S 60.29°W; $T_{\text{water}} = 1.4^{\circ}\text{C}$; $T_{\text{air}} = 2.3^{\circ}\text{C}$; $r = 3608$ m; C) 29.12.2011 01:06; 56.49°S 00.00°E; $T_{\text{water}} = -0.8^{\circ}\text{C}$; $T_{\text{air}} = -0.5^{\circ}\text{C}$; $r = 1116$ m; D) 01.01.2012 02:38; 43.96°S 07.44°E; $T_{\text{water}} = 8.8^{\circ}\text{C}$; $T_{\text{air}} = 8.7^{\circ}\text{C}$; $r = 879$ m; Figure 2E: Day-time thermographic (top) and visual (bottom) video snippets of an automatically detected whale blow: 28.12.2011 14:41; 58.65°S 0.02°E; $T_{\text{water}} = -1.5^{\circ}\text{C}$; $T_{\text{air}} = -0.6^{\circ}\text{C}$; $r = 1072$ m; doi:10.1371/journal.pone.0071217.g002

Visual (including IR) detection of whales depends on them being at the sea surface, a factor known as availability bias. A simple numerical model was developed to estimate this bias for a set of detection radii, average dive/surface times and ship speeds. The model assumes randomly distributed, horizontally stationary animats (animal agents, Figure S3) which are presumed to surface and dive according to a binary dive function extracted from surface and subsurface periods as published in the literature (Table S1). The initial vertical position of each animat is based on its dive state (*at surface/subsurface*) at a randomly chosen point in time t_0 during its dive cycle. Model time progresses in 1-minute steps, Δt , with each animat's vertical position (*at surface/subsurface*) being updated according to its dive function at $t_0 + \Delta t$.

Concurrently, the ship transects the model space diagonally at a speed of 4.5 knots, a value typical for seismic surveys. The animat is considered detectable from the moving ship if it is at the surface and inside an assumed detection radius $r_{\text{detection}}$. It is considered undetectable if it is diving or outside $r_{\text{detection}}$. To estimate, in the context of marine mammal mitigation, the likelihood of detecting an animal before it is within the exclusion zone, which is moving with the seismic source (here $r_{\text{exclusion}} = 500$ m, centered 500 m behind the ship), the model algorithm applies the following classification: An animat is considered

detected timeously, if it surfaced within the detection zone before being within the exclusion zone;
missed, if it is within the (moving) exclusion zone before having surfaced inside the detection zone.

The probability for detecting an animat *timeously* is then calculated by dividing the number of animats *detected timeously* over the total number of animats blanketed by the (moving) exclusion zone.

Results

The automatic thermographic whale detection system introduced in this study continuously scans for whale blows in the environs of a ship operating offshore. By human screening, several hundred whale blows were unambiguously identified within a range of 8 km, with most of the blows originating from a distance of less than 4 km range (Figure 2). Using the automatic detection system on data from 7 expeditions, we identified more than 4500 whale blows at distances of up to 5500 m. These blows occurred over the course of more than 300 ship-whale encounters, during both night and day (defined as period between civil twilight), and for a wide range of environmental conditions, with sea surface temperatures ranging between -1.8 and $+22.7^{\circ}\text{C}$ and wind speeds between 0 and 7 Bft.

A key component of the automatic detection system is the *classifier*, which selects probable whale blows from a multitude of thermal anomalies provided by the *detector*. The *classifier's* efficiency is described quantitatively by Receiver-Operator-Characteristic (ROC) curves for false positive and false negative detection events (Figure 3A, green and red curves). The resulting Area Under the Curve (AUC) value, which is an integral measure of the reliability of the *classifier*, was 0.99 for the training data set, and 0.98 for the test data set. These AUC values however likely overestimate *classifier* performance; although the test data set and the training data set do not overlap, both are drawn from the same ship-whale encounters (collected during expedition ANT-27.2) and therefore represent similar environmental conditions and encounter ranges. To avoid this bias, we compiled an independent validation data set of 1074 manually classified thermal recordings, including data from a different expedition (ANT-28.4) [19], and evaluated recordings separately for day and night. *Classifier* performance was better at night (AUC = 0.98) than during day (AUC = 0.90), probably due to the lack of glare in the night-time images (Figure 3A, black and blue curves).

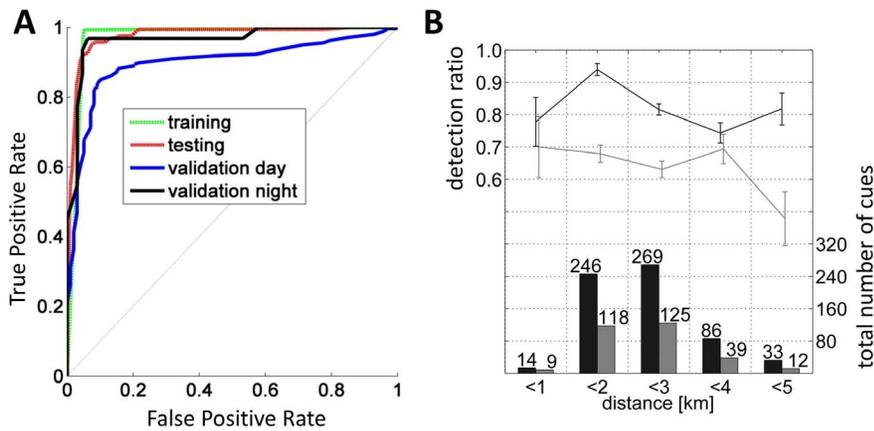


Figure 3. Detection performance and efficiency. A: Cue-based classifier ROC curves for training, testing and two (day and night) validation datasets. AUC values: Training: 0.99; Testing: 0.98; Validation day: 0.90; Validation night: 0.98. B: Lines with error bars: Proportion of successful automatic detections of all visually detected blows (black), and proportion of visually detected blows of all automatic detections (grey) versus distance (bin width of 1 km). Errorbars give the standard error. Bar plot: Number of automatic (black) and visual (grey) detections versus distance. doi:10.1371/journal.pone.0071217.g003

To evaluate the overall performance of the automatic detection system, comparisons with the “industry standard” of dedicated, trained visual observers were performed. Analyses were conducted using independent data sets at two different levels: a) at the encounter level (using ANT-28.2 data) and b) at the cue (i.e. blow) level, using ANT-27.2 data.

Of 26 visually recorded encounters during ANT-28.2, the IR system automatically detected 24. One of the two *missed encounters* occurred during high wind speed (11.5 ms^{-1}) and presence of an unusually high number of growlers (floating blocks of ice), which generated intense changes in contrast throughout the image. This probably led to a high STA/LTA threshold, leaving the blow undetected. Alternatively, the blow might have been distant, as indicated by the visual observers being unable to identify the animal’s species. The second *missed encounter* was that of a blue whale which surfaced right in front of the ship (pers. comm. D. Verbelen) and probably was too close (<110 m) to be within the field of view of the IR camera.

During the same expedition, the total number of infrared based encounters amounts to 85. Of these, 45 IR-based encounters occurred when the MMOs were on-effort, logging a total of 24 concurrent sightings (53%). The remaining 40 IR-based encounters occurred when the visual observers were off-effort. For 45% of (virtual) 2-hour watches, no false positives occurred; for more than 90% of the 2-hour watches less than 30 *false positives* occurred.

To determine range dependent detector efficiency, cue-based comparisons were performed. Each analysis period commenced with the first visual spotting of a blow, ensuring that the MMOs were alerted. Within the 0–5 km range, the algorithm detected 82% of all blows (303) sighted by the alerted observers, exhibiting a rather range-unspecific detection efficiency between 75 and 95% (Figure 3B, black data). The remaining blows were discernible in the thermographic recordings, but were too faint or unspecific to be picked up by the automatic detection algorithm. Conversely, the observers spotted about 63% of all events that were detected automatically within a 5 km range (Figure 3B, grey data). The average *false positive* (false alert) rate of the IR system was about 6 per hour, with false alerts frequently being caused by nearby birds.

Discussion

This study introduces a ship-based implementation of thermal imaging for automated marine mammal detection, consisting of a

spinning IR camera and an algorithm that detects whale blows on the basis of their thermal signature. The system detected 92% of all visually logged ship-whale encounters during expedition ANT-28.2 and 82% of cues recorded by a team of visual observers during ANT-27.2 (Figure 3). During these expeditions, the system detected about twice (2.5- and 2.1-fold) as many encounters and cues, respectively, as recorded by the MMOs, with false positive rates of less than 1 per 4 minutes occurring for the majority (>90%) of virtual 2-hour watches. On occasion, false positive rates exceeded 1 per minute, due to flocks of birds or presence of growlers during high sea states. However, the system *per se* is conceived as a “bell-ringer” rather than to operate completely unsupervised, alerting the MMO to likely whale blows in the ship’s environs while providing instant playback and documentation of the thermographic recordings. This allows the operator to easily verify the event and quickly determine whether a shut-down request should be issued or not.

Night and day detections rates are comparable. On expedition ANT-28.2 during 76 hours of nighttime observations, 7 encounters were detected by the IR system, resulting in a detection rate of 0.09 encounters/hour while the daytime encounter rate was 0.13 encounters/hour, based on 583 daytime hours. Generally, nighttime performance exceeds daytime performance due to the lack of glare and diffuse reflections, as indicated in the increased nighttime AUC values of Figure 3A (black vs. blue curve).

As with any optical detection system, its performance varies with environmental conditions such as fog, precipitation, sea state, glare, water- and air-temperatures and ambient brightness (insolation). Due to the currently available limited number of visual (i.e. reference) sightings, a statistically significant analysis of system performance in relation to these parameters cannot yet be performed. However, some general trends are already discernible. During ANT-28.2, the number of detections did not degrade up to wind speeds of 7 Bft (corresponding to sea state 6); Wind speeds higher than 7 Bft occurred for only a brief period (<12 h) during which no ship-whale encounter was detected. Detections also occurred at water temperatures of up to 23°C , yet sampling effort was heavily biased towards polar water temperatures with only 5 encounters having occurred in waters warmer than 15°C . Air temperatures are irrelevant to system performance as (dry) air is quasi-transparent in the LWIR (8–12 μm) band used. Contrastingly, fog may significantly compromise system performance.

Depending on droplet size, visibility in the LWIR band has been noted to be equal or better than in the visual (0.3–0.7 μm) band. Fog, rain and snow occurred rarely (visibility was less than 500 and 1000 m for only 0.5 and 2% of the expedition, respectively), representative of typical Southern Ocean conditions during austral summer. Glare resembles a clutter of warm anomalies in thermographic images, resulting in high local contrasts. This raises the STA/LTA threshold, rendering the detection of blows less likely. However, the field of glare in the IR image is significantly narrower than for visual observers, as the detector only considers the local contrast of each analyzed tile (between 1 and 3° horizontal field of view, FOV), rather than that of the human (i.e. binocular's) field of view (8°FOV).

As yet, the upper limit of sea surface temperatures and the lower limit of mammal sizes allowing reliable blow detection remain unknown. Our results were obtained for a limited range of environmental conditions and species, with sea surface temperatures predominantly between -1.8 to $+10^\circ\text{C}$, wind speeds below 7 Bft, and species consisting mainly of humpback (*Megaptera novaeangliae*), minke (*Balaenoptera bonaerensis*) and fin whales (*Balaenoptera physalus*). A comprehensive evaluation of the algorithm's efficiency for other whale species and pinnipeds, and for higher sea surface temperature and wind speeds, including their cross-dependencies, requires further studies which are in planning. Meanwhile, our results demonstrate that the IR systems works well for large whales in the subpolar and polar oceans, and provides a major breakthrough for night time detection.

The reliability of visual or thermographic observations for cetacean mitigation is strongly dependent on the ratio of a whale's surface versus dive times, a factor known as availability bias. Modeled availability biases range from 99% for whales exhibiting dive times of 10 min (baleen whales), over 65% for dive times of 40 min (sperm whale), to 45% for dive times of 60 min (beaked whales). The degree to which availability bias impedes a *timeous detection*, that is, a whale's detection before it enters the exclusion zone, depends further on the radius and relative position of the detection and exclusion zone and the ship's speed. Longer dive times in combination with a detection radius below 3 km lead to a dramatic decrease in the probability that a whale can be detected *timeously* (Figure 4, Figure S3). Baleen whales, for example, are very likely (>90%) to have surfaced within the IR system's detection range before entering the exclusion zone, whereas whales with long dive times (odontocetes in particularly) stand a reduced chance for being detected *timeously* (40–70%). The modeled values likely represent conservative (low) estimates, as the possibility of avoidance responses of the animals to loud sounds [20–23] was disregarded in the model. In addition, whale pods with asynchronous diving patterns present multiple detection opportunities, increasing the likelihood and of being detected before entering the exclusion zone.

Further protection might be achieved by concurrent passive acoustic monitoring of animal sounds [24]. However, quantitative and comprehensive studies of the detection probability of passive acoustic monitoring are largely lacking. With the exception of odontocetes, which emit clicks during foraging dives, the vocalization behavior of most species at the gender and contextual level is insufficiently understood for quantitative estimates of acoustic detectability. By contrast, whales need to respire regularly, rendering visual or thermographic detection methods reliable once the detection bias is minimized, as achieved by the system described herein. Further progress in sensor technology, such as the availability of multiple band (far and mid wavelength IR and visual) sensors and higher image resolution can be

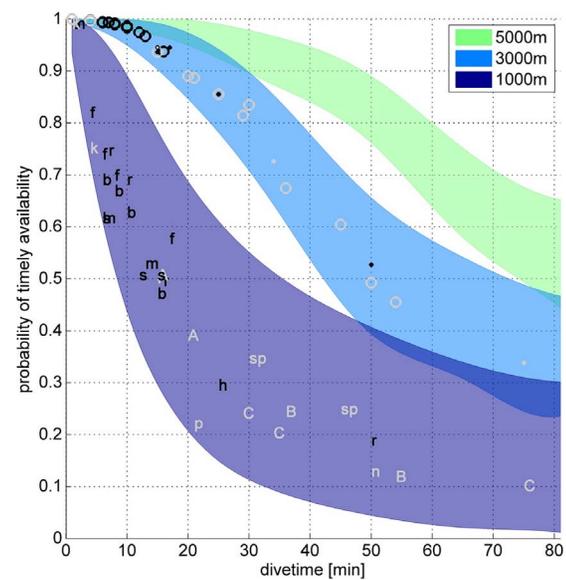


Figure 4. Timeous availability. Probability of a whale being at the surface within a detection radius r before it is within the exclusion zone (radius 500 m, acoustic source towed 500 m behind the ship, see Figure S3) as a function of diving time. Ship speed is assumed to be 4.5 knots. Colored areas indicate different maximum radii over which whales can be reliably detected (1, 3 and 5 km). Upper limits of filled areas correspond to the maximum, lower limits to the minimum known surface times. Mean values for various whale species are indicated by letters: A=Arnoux's beaked whale, B=Blainville's beaked whale, b=blue whale, C=Cuvier's beaked whale, f=fin whale, h=humpback whale, k=killer whale, m=minke whale, n=northern bottlenose whale, p=pilot whale, r=right whale, s=sei whale, sp=sperm whale. Letters are only displayed for a detection radius of 1 km. Within in the blue (3 km) area, circles and dots vertically aligned with letters indicate whether the underlying data represents single measurements (dots) or group averages (circles). Light grey symbols represent odontocetes, black symbols mysticetes.

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expected to further increase detection reliability and therefore whale protection.

The IR system presented here has additional benefits. It provides precise and reproducible distance and bearing information which can be used to study the response of whales to acoustic exposure with regard to locomotive behavior, respiration rates, and dive cycles (Figure 1). Automated blow detection can be coupled with acquisition of additional visual imagery for species identification and morphometric analyses, an approach currently under development. The increased use of such systems will eventually result in a large number of well documented encounters, providing urgently needed, statistically robust data resolved at the species and contextual levels [23]. With regard to marine mammal mitigation applications, the real-time detection and tracking capability of thermal imaging methods allows for fast and correct decisions, day and night, throughout seismic surveys or naval activities. In particular, the IR system's ability to concurrently detect multiple whales allows for full situation awareness, even in the presence of many whales.

Supporting Information

Figure S1 Absolute (A) and relative (B) error estimation of image and binocular based distance calculation. It is assumed that the vertical position of a whale in the thermal image is determined with an accuracy of ± 1 pixel and with $\frac{1}{4}$ reticule

accuracy using binoculars. Due to the spherical triangulation used to calculate the distance, this results in a distance dependent error. Red color indicates upper, blue color indicates lower error boundaries. For distances less than 5 km, relative errors are within 12%. (TIF)

Figure S2 Schematic of the STA/LTA algorithm. The example shows the V-shaped blow of a humpback whale. The black curve shows the short term contrast average (STA), the red curve the long term contrast average (LTA) computed from the sequence of snippets above. Blue and pink windows indicate the number of images used to calculate STA and LTA respectively. The blue curve indicates the adaptive threshold (AT) as computed from the right hand side of equation (1). (TIF)

Figure S3 Schematic of ship position, exclusion zone and detection zone. For the animat simulation model (see Figure 4), the sound source was assumed to trail 500 m behind the ship. (TIF)

Table S1 Surface and dive times used for animat model, as taken from the literature. (DOC)

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Text S1 Detailed description of detection and classification algorithms. (DOC)

Acknowledgments

The authors would like to extend many thanks to the Laboratory for Polar Ecology (PoE): coordination C.R. Joiris; observers D. Verbelen and J. Haelters for contributing the independent visual sighting data collected during *RV Polarstern* expedition ANT-XXVIII/2. We thank B. Fabry for his comments and continued support. A. Bombosch organized the recording of to-the-second whale blows during *RV Polarstern* Expedition ANT-XXVII/2. The support by crew and officers of *RV Polarstern* and Reederei Laisz during the expeditions and the installation of the thermal scanner is greatly appreciated. S. Richter operated the system on 3 out of the 6 expeditions and wrote the data storage module. We would like to thank W. John Richardson and an anonymous reviewer for their insightful comments which helped greatly improving the manuscript.

Author Contributions

Conceived and designed the experiments: DPZ LK OB. Performed the experiments: DPZ LK. Analyzed the data: DPZ. Wrote the paper: DPZ OB. Compiled whale diving data from literature: EB.



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Jolie Harrison

Chief, Permits and Conservation Division, Office of Protected Resources,
National Marine Fisheries Service.

ITP.Young@noaa.gov

RE: Document No. 2019-17634

Dear Ms. Harrison:

1) It is reasonable and prudent to conclude that if there is going to be around-the-clock seismic blasting, there needs to be around-the-clock monitoring; if effective monitoring at night cannot be accomplished, blasting should not occur at night. The claim that around-the-clock blasting without night observers will have a lower impact than a longer period of day-light-only blasting is unsubstantiated. The standard of reasonableness and prudence places the burden of proof on the side of non-harm; in other words, an opinion with this much weight must be justified by more than conjecture. The National Marine Fisheries Service must support their reasoning that round-the-clock blasting will have a lower impact on whales with evidence.

2) Hilcorp cannot not be treated with the same leniency given to other companies; having paid more than 3 million dollars in fines for violations in the past 18 years, their [track-record](#) warrants greater precaution. The accumulation of flagrant regulatory violations make it clear that an observer is needed on vessels conducting seismic surveys at all times. Hilcorp has shown that it cannot be trusted to comply with the spirit or letter of any regulation. As soon as the company started working in Alaska in April 2012, it began to accumulate violations. By October 2015, the Alaska Oil and Gas Conservation Commission (AOGCC), had [documented 25 instances](#) in which Hilcorp violated its regulations, prompting a reprimand that revealed the frustration of the commission: "The disregard for regulatory compliance is endemic to Hilcorp's approach to its Alaska operations and virtually assured the occurrence of this violation," the chair of the commission [wrote to the company in November 2015](#), "Hilcorp's conduct is inexcusable." Hilcorp again displayed bad-faith non-compliance in 2016, making clear that they must be held on a tight regulatory leash. In issuing a \$30,000 fine to Hilcorp, [AOGCC commented that](#) "Hilcorp's history of noncompliance and its failure to take the rudimentary measure of entering AOGCC's requirements in its regulatory tracking system preclude any claim that Hilcorp has acted in good faith." As Alaska Public Radio reported in 2016 ("As new companies enter Alaska oil fields, violations spike"), the Alaska Oil and Gas

The Kachemak Bay Conservation Society's mission is to protect the environment of the Kachemak Bay region and greater Alaska by encouraging sustainable use and stewardship of natural resources through advocacy, education, information, and collaboration.



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Conservation Commission proposed more than \$1 million fines for Hilcorp, with regulators writing that “disregard for regulatory compliance is endemic to Hilcorp's approach.” A company with this history needs observers working on board at all times that blasting is being conducted.

3) Additionally, in the wake of an extraordinarily warm and dry summer and fall, residents of Alaska's Kenai Peninsula are increasingly alarmed about the impacts of our warming climate on the future of our oceans, our fisheries, and our endangered species. Given the profound threats faced by our oceans, the National Marine Fisheries Service must provide a more in-depth analysis of the cumulative impacts of drought, warming waters, changing hydrology, and increased acidity. In other words, it may be that seismic blasting alone will have a negligible impact on the Beluga whale population in Cook Inlet, but it is possible that seismic blasting *at this time*—coincident with an extreme drought that threatens wild salmon, coincident with another Blob, with ocean waters that are so warm as to threaten prey species down to the zooplankton—are a threat to Steller Sea Lions and the remaining 328 Beluga Whales in Cook Inlet. An analysis along these lines is needed to meet the standard of reasonableness and prudence.

4) Lastly, a reasonable and prudent decision must include greater evidence and analysis of the impacts of blasting to the full spectrum of prey species of Beluga Whales and Steller Sea Lions, all the way down the food chain to the prey of the prey, the zooplankton. It is not enough to surmise that impacts will be insignificant due to tidal flushing. Prudence prohibits the use of conjecture and surmise as the basis to green light a project; prudence requires the demonstration of non-harm, especially given the fact that whale scientists and charter boat captains have recently documented large numbers of fin, humpback and orca whales feeding in the vicinity of the proposed seismic blasting, and fishermen note the southeast corner of the seismic testing zone is the hottest fishing in Lower Cook Inlet right now.

We sincerely hope that you will consider these pressing matters in your deliberations and you will take the responsibility of your office with the utmost seriousness. The residents of the Kenai Peninsula are watching and are counting on you to protect the natural resources we all love and depend on.

Sincerely,

Roberta Highland
President, Kachemak Bay Conservation Society

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MARINE MAMMAL COMMISSION

16 September 2019

Ms. Jolie Harrison, Chief
Permits and Conservation Division
Office of Protected Resources
National Marine Fisheries Service
1315 East-West Highway
Silver Spring, MD 20910-3225

Dear Ms. Harrison:

The Marine Mammal Commission (the Commission), in consultation with its Committee of Scientific Advisors on Marine Mammals, has reviewed the National Marine Fisheries Service's (NMFS) 16 August 2019 notice (84 Fed. Reg. 41957) proposing to modify the letter of authorization (LOA) issued to Hilcorp Alaska, LLC (Hilcorp) authorizing it to conduct oil and gas activities in Cook Inlet, Alaska, under section 101(a)(5)(A) of the Marine Mammal Protection Act (MMPA).

NMFS has proposed to modify Hilcorp's LOA to clarify that clearance of the entire 500-m exclusion zone during its 2D/3D survey is not necessary prior to initiating ramp up of airguns at night. Instead, protected species observers would be required to monitor the exclusion zone to the 'greatest extent possible' to verify that marine mammals are not present¹ prior to ramp up at night. NMFS previously required that operators in Cook Inlet clear the entire exclusion zone, and in some cases the entire Level B harassment zone, prior to initiating ramp-up procedures. Exceptions have been made to the clearance requirement during nighttime or low-visibility conditions if passive acoustic monitoring (PAM) was used to confirm no acoustic detections of marine mammals in the 30 minutes prior to ramp up². However, NMFS did not include PAM as a required mitigation or monitoring measure in the final rule based on past unsuccessful deployments of PAM devices during seismic surveys in Cook Inlet³. Nor did NMFS require any other type of night-vision device⁴ to aid in the detection of marine mammals at night, which are standard for other industry⁵ and research⁶ seismic surveys.

As part of its rationale to modify the LOA, NMFS indicated in the *Federal Register* notice that allowing ramp up at night would reduce the overall number of days that the acoustic source would be operating. Although that may be the case, the Commission is not convinced that NMFS adequately considered the use of both towed PAM and night-vision devices to aid in the detection of marine mammals at night, particularly since NMFS requires other operators to use both

¹ Based on a clearance time of 15 minutes for pinnipeds and porpoises and 30 minutes for other cetaceans.

² See, for example, 84 Fed. Reg. 35093, 83 Fed. Reg. 63312, and 83 Fed. Reg. 29306.

³ See section 13 of Hilcorp's application and the preamble to the final rule (84 Fed. Reg. 37456).

⁴ Such as night-vision binoculars, forward-looking infrared devices, or thermal imaging cameras.

⁵ See, for example, 83 Fed. Reg. 63359 and 83 Fed. Reg. 29286.

⁶ <https://www.fisheries.noaa.gov/webdam/download/93867938>.

monitoring methods during seismic surveys conducted in U.S. and international waters worldwide. As such, the Commission recommends that NMFS reconsider requiring Hilcorp to use towed PAM and night-vision devices to better assess whether the exclusion zone is clear prior to implementing ramp-up procedures at night and consult with other seismic operators regarding the standard use of these devices in other regions. NMFS also should consult with acousticians at the Alaska Fisheries Science Center and the University of St. Andrews regarding acoustically monitoring for the various species in Cook Inlet. In addition, the Commission recommends that NMFS require Hilcorp to limit ramp up at night and during low-visibility conditions to situations in which operational planning cannot reasonably avoid such circumstances, consistent with requirements in other recent authorizations⁷.

In its review of the Hilcorp LOA for year 1 activities, the Commission notes that the radial distances of the exclusion and safety zones⁸ were not specified in the LOA. NMFS's intent to include those distances in the LOA is indicated in section 217.164(f) of the final rule (84 Fed. Reg. 37503)—

- (1) For all relevant in-water activity, Hilcorp must implement shutdown zones/exclusion zones (EZ) with radial distances as identified in any LOA issued under §§ 216.106 of this chapter and 217.166. If a marine mammal is sighted within or entering the EZ, such operations must cease.
- (2) For all relevant in-water activity, Hilcorp must designate safety zones (SZ) for monitoring with radial distances as identified in any LOA issued under §§ 216.106 of this chapter and 217.166 and record and report occurrence of marine mammals within these zones.

Although the radial distances of the exclusion and safety zones were included in Table 20 of the preamble to the final rule (84 Fed. Reg. 37494), they also should have been included in the LOA⁹, consistent with other LOAs¹⁰. Additionally, the LOA referred to 'mitigation and monitoring zones' and 'Level B isopleths'. It is unclear whether the mitigation and monitoring zones are the same as the exclusion and safety zones or whether they are synonymous with the Level A and B harassment zones. In any case, all of the relevant zones should be specified in the LOA, including those associated with implementation of mitigation measures and those for which takes must be enumerated and reported. The Commission recommends that NMFS specify the radial distances of the exclusion and safety zones, as well as the Level A and B harassment zones, for all sound sources and remove all references to mitigation and monitoring zones in Hilcorp's modified and subsequent LOAs.

⁷ See, for example, 83 Fed. Reg. 63347 and 83 Fed. Reg. 29270.

⁸ In some instances, the exclusion and safety zones are much less than the Level A and B harassment zones.

⁹ Rather than specify the zones, NMFS inserted the same text in the LOA that was included in section 217.164(f)(1) and (2) of the final rule, resulting in circular statements that lack the necessary details.

¹⁰ NMFS may have decided not to include the relevant zones because Hilcorp is required to conduct sound source verification (SSV) during the 3D seismic survey and shallow-hazard survey (i.e., when sub-bottom profilers are used) and the zones may change. However, Hilcorp is authorized to conduct many more activities than those two activities, and NMFS included condition 5(a) in the LOA that allows for the zones to change based on the SSVs. Thus, all relevant zones should have been included in the LOA.

Ms. Jolie Harrison
16 September 2019
Page 3

I trust these comments will be helpful. Please let me know if you or your staff have questions with regard to the Commission's recommendations.

Sincerely,

A handwritten signature in blue ink that reads "Peter O. Thomas". The signature is written in a cursive style with a large initial "P" and "T".

Peter O. Thomas, Ph.D.,
Executive Director

NMFS received many copies of two form letters. One copy of each form letter is provided below.



ITP Young - NOAA Service Account <itp.young@noaa.gov>

Defend Belugas From Oil and Gas Drilling

1 message

Sat, Sep 14, 2019 at 11:58 PM

Reply-to: [REDACTED]
To: ITP.Young@noaa.gov

Dear Jolie Harrison,

I'm writing to urge you to do everything in your power to defend the population of endangered beluga whales in Cook Inlet, Alaska — down to just 328 individuals.

Rules allowing corporations to use sonic blasts to explore for oil and gas in whale habitat will not only hurt these whales, but also violate the Marine Mammal Protection Act and Endangered Species Act. We can't permit oil and gas exploration at the expense of endangered wildlife. So please — do everything you can to protect these belugas and keep companies like Hilcorp from blasting whales to hunt for oil.

Sincerely,

[REDACTED]



ITP Young - NOAA Service Account <itp.young@noaa.gov>

Marine Mammal Takes in Cook Inlet Alaska

1 message

Fri, Sep 13, 2019 at 4:33 PM

Reply-to: [REDACTED]
To: ITP.Young@noaa.gov

Dear Jolie Harrison,

I urge you to carefully consider the impact any decisions or actions regarding this issue will have.

Researchers have long recognized that seismic air guns can injure or even kill large whales roughly within 500 meters to 1 kilometer of the blasts, depending on offshore water depth and the number of guns firing. Vessels in US waters are required to cease firing when marine mammals are within an unsafe zone. We need to put the science first and require marine mammal observers during all times of testing to minimize this damage. (<https://academic.oup.com/bioscience/article/68/12/1024/5160052>)

The Cook Inlet marine environment is already under intense stress and the endangered Beluga whale and recent decline of Cook Inlet salmon fisheries are symptoms of this stressed system. We already know that seismic blasting is harmful to marine life, so ensuring that whales are not in direct proximity to the blasting is absolutely necessary to minimizing harm regardless of time of day. If there is going to be around the clock blasting there needs to be around the clock monitoring; if this cannot be accomplished, blasting should not occur at night.

Sincerely,

[REDACTED]

NMFS received several public comment letters from private citizens which are provided below.



ITP Young - NOAA Service Account <itp.young@noaa.gov>

Re: keep oil and gas profiteers out of cook - they are polluters and destroyers of fish and people

To: itp.young@noaa.gov, [REDACTED]

Sat, Aug 17, 2019 at 3:06 PM

public comment on federal register

theres should be no operations at night.if these profiteers cant see if they are killign and murdering sea creatures at night, THEN THEY SHOULD NOT OPERATE. WHY ARE YOU LETTING THIS PROFITEER DELIBERATELY KILL SEA CREATURES. THAT IS DISGUSTING, EVIL, WICKED, AND NEEDS TO BE STOPPED. THIS COMEMTN IS FOR THE PUBLCI RECORD PLEASE RECEIPT. [REDACTED]

On Fri, Aug 16, 2019 at 10:07 AM [REDACTED] wrote:

[Federal Register Volume 84, Number 159 (Friday, August 16, 2019)]
[Notices]
[Pages 41957-41958]
From the Federal Register Online via the Government Publishing Office [www.gpo.gov]
[FR Doc No: 2019-17634]

DEPARTMENT OF COMMERCE

National Oceanic and Atmospheric Administration

RIN 0648-XR036

Takes of Marine Mammals Incidental to Specified Activities;
Taking Marine Mammals Incidental to Oil and Gas Activities in Cook
Inlet, Alaska

AGENCY: National Marine Fisheries Service (NMFS), National Oceanic and
Atmospheric Administration (NOAA), Commerce.

ACTION: Notice; request for comments on modification of Letter of
Authorization.

SUMMARY: Pursuant to the Marine Mammal Protection Act (MMPA), as
amended, and implementing regulations, NMFS is requesting comments on
its proposal to modify a Letter of Authorization issued to Hilcorp
Alaska LLC (Hilcorp) to take marine mammals incidental to oil and gas
activities in Cook Inlet, Alaska.

DATES: Comments and information must be received no later than
September 16, 2019.

ADDRESSES: Comments should be addressed to Jolie Harrison, Chief,
Permits and Conservation Division, Office of Protected Resources,
National Marine Fisheries Service. Physical comments should be sent to
[1315 East-West Highway, Silver Spring, MD 20910](https://www.hilcorp.com) and electronic comments
should be sent to ITP.Young@noaa.gov.

Instructions: NMFS is not responsible for comments sent by any
other method, to any other address or individual, or received after the
end of the comment period. Comments received electronically, including
all attachments, must not exceed a 25-megabyte file size. Attachments

to electronic comments will be accepted in Microsoft Word or Excel or Adobe PDF file formats only. All comments received are a part of the public record and will generally be posted online at <https://www.fisheries.noaa.gov/action/incidental-take-authorization-hilcorp-alaska-llc-oil-and-gas-activities-cook-inlet-alaska> without change. All personal identifying information (e.g., name, address) voluntarily submitted by the commenter may be publicly accessible. Do not submit confidential business information or otherwise sensitive or protected information.

FOR FURTHER INFORMATION CONTACT: Sara Young, Office of Protected Resources, NMFS, (301) 427-8401.

SUPPLEMENTARY INFORMATION:

Background

Sections 101(a)(5)(A) and (D) of the MMPA (16 U.S.C. 1361 et seq.) direct the Secretary of Commerce (as delegated to NMFS) to allow, upon request, the incidental, but not intentional, taking of small numbers of marine mammals by U.S. citizens who engage in a specified activity (other than commercial fishing) within a specified geographical region if certain findings are made and either regulations are issued or, if the taking is limited to harassment, a notice of a proposed authorization is provided to the public for review.

An incidental take authorization shall be granted if NMFS finds that the taking will have a negligible impact on the species or stock(s), will not have an unmitigable adverse impact on the availability of the species or stock(s) for subsistence uses (where relevant), and if the permissible methods of taking and requirements pertaining to the mitigation, monitoring and reporting of such takings are set forth.

NMFS has defined ``negligible impact'' in 50 CFR 216.103 as an impact resulting from the specified activity that cannot be reasonably expected to, and is not reasonably likely to, adversely affect the species or stock through effects on annual rates of recruitment or survival.

The MMPA states that the term ``take'' means to harass, hunt, capture, kill or attempt to harass, hunt, capture, or kill any marine mammal.

Except with respect to certain activities not pertinent here, the MMPA defines ``harassment'' as: Any act of pursuit, torment, or annoyance, which (i) has the potential to injure a marine mammal or marine mammal stock in the wild (Level A harassment); or (ii) has the potential to disturb a marine mammal or marine mammal stock in the wild by causing disruption of behavioral patterns, including, but not limited to, migration, breathing, nursing, breeding, feeding, or sheltering (Level B harassment).

Summary of Request

NMFS issued regulations governing the take of eleven species of marine mammal, by Level A and Level B harassment, incidental to Hilcorp's oil and gas activities on July 31, 2019; 84 FR 37442). These regulations include mitigation, monitoring, and reporting requirements for the incidental take of marine mammals during the specified activities. As detailed in the regulations (50 CFR 217.167), adaptive management measures allow NMFS to modify or renew Letters of Authorization as necessary if doing so creates a reasonable likelihood of more effectively accomplishing the goals of mitigation and monitoring set forth in those regulations.

NMFS proposes to modify a mitigation measure pertaining to 3D seismic surveying during Year 1 of Hilcorp's activity. NMFS published a mitigation measure in error that stated before ramp up of seismic airguns during the 3D seismic survey, the entire exclusion zone (EZ) must be visually cleared by protected species observers (PSOs). This measure is correct for operations beginning in daylight hours. However, visually clearing the entirety of the EZ to ramp up airgun activity at night was not NMFS' intent. The intent was that PSOs should monitor the EZ to the greatest extent possible for 30 minutes prior to ramp-up of

nighttime operations, but with the understanding that it is not possible to observe the entirety of the EZ at night and that Hilcorp would still be allowed to initiate ramp-up as long as no marine mammals were seen during this time. If any marine mammal is observed in the EZ, during daylight hours or at night, ramp up would not commence until

[[Page 41958]]

either the animal has voluntarily left and been visually confirmed outside the EZ or the required amount of time (15 for porpoises and pinnipeds, 30 minutes for cetaceans) have passed without re-detection of the animal. The analysis and findings contained in the final rule were made under the premise that nighttime ramp up of airguns is allowable.

Ramping up airgun activity at night is essential to Hilcorp's survey design and minimizes the amount of days that active acoustic sources are emitting sound into the marine environment. As described in Hilcorp's application, acquisition of one line of 3D seismic takes approximately five hours. At the end of a line while the vessel turns to prepare for the next line acquisition, NMFS requires that airguns are turned off, to reduce the amount of unnecessary noise emitted into the marine environment. Turning the source vessel takes approximately one and a half hours, during which no noise is emitted from airguns. By allowing ramp up of airguns at night, the total number of 3D seismic survey days is notably reduced and marine mammal habitat noise will sooner be reduced to ambient noise levels.

Specifically, while there is a somewhat higher probability that a marine mammal might go unseen within the clearance zone when the airguns are initiated at night, the likelihood of injury is still low because of the ramp-up requirement, which ensures that any initial injury zone is small and allows animals time to move away from the source, and the fact that PSOs are on duty monitoring the exclusion zone to the degree possible at that time. Further, any potential slight increase in the probability of injury (in the form of a small degree of PTS, and not considered at all likely, or authorized, for beluga whales or other mid-frequency specialists) is offset by the reduced behavioral harassment and reduced potential for more serious energetic effects expected to result from the significant reduction in the overall number of days across which the area will be ensonified by the airgun operation.

Ramp up of airguns at night is also the most practicable survey design, which allows the survey to be completed as quickly as possible before weather conditions deteriorate and daylight decreases in Cook Inlet, and at less cost.

Of important note, this change in mitigation does not change either the predicted take numbers or the negligible impact analysis, as the predicted Level A harassment (injury) numbers conservatively do not include any sort of an adjustment to account for the effectiveness of any of the measures.

Request for Public Comments

In order to maintain a transparent process for issuance of incidental take authorizations and because the public was not able to comment on this mitigation measure, NMFS is requesting comment on amending the Year 1 Letter of Authorization to allow ramp up of airguns at night without requiring full clearance of the EZ by PSOs. Full clearance of the EZ by PSOs would still be required for all operations in daylight hours. A draft of the amended LOA is available at: <https://www.fisheries.noaa.gov/action/incidental-take-authorization-hilcorp-alaska-llc-oil-and-gas-activities-cook-inlet-alaska>.

Dated: August 13, 2019.

Cathryn E. Tortorici,
Acting Director, Office of Protected Resources, National Marine
Fisheries Service.

[FR Doc. 2019-17634 Filed 8-15-19; 8:45 am]

BILLING CODE 3510-22-P



ITP Young - NOAA Service Account <itp.young@noaa.gov>

Belugas need protection

1 message

Mon, Sep 16, 2019 at 8:03 PM

Reply-to: [REDACTED]
To: ITP.Young@noaa.gov

Dear Jolie Harrison,

I'm deeply concerned about the current administration's plan to allow the expansion of offshore oil and gas exploration in Alaska.

In 2013, I researched the effect of Navy sonar on whales and other marine mammals and wrote an article on the subject, "Navy Sonar Versus Marine Mammals," for Eric Chudler's Neuroscience for Kids website: <https://faculty.washington.edu/chudler/marinem.html>

I'd like to call out the particular recommendations made by the California Coastal Commission for protecting marine mammals against the harmful effects of sonar:

- Establishing larger areas (up to 2 km/1.25 miles) where no sonar is allowed when marine mammals are nearby.
- Avoiding use of mid-frequency sonar in biologically sensitive areas like marine sanctuaries.
- Reducing sonar sound levels when marine mammals are difficult to spot due to darkness or low visibility.

I implore you to use every lever at your disposal to protect these important creatures that play a critical role in the health of oceans and food chains and simply make the world a nicer place to live.

Sincerely,

[REDACTED]



ITP Young - NOAA Service Account <itp.young@noaa.gov>

Doc#2019-17634 Attn: Jolie Harrison

1 message

Wed, Sep 4, 2019 at 10:53 PM

[REDACTED]
To: "ITP.young@noaa.gov" <ITP.young@noaa.gov>

To Jolie Harrison, I am an Alaskan resident for 43 years and have commercial fished Cook Inlet for 25 years. Having spent numerous hours in Cook Inlet and Kachemak Bay, I am writing in protest of the modification or renewal of Letters of Authorization to Hilcorp's night ramp up for their seismic activity planned for September. How can the observers possibly see any marine activity at night? It is essential that they can see the marine life, especially whales that are so sensitive to such bombardment. I happen to be encountering several pods of Orca whales and Humpback whales this week, within the Bay and near the proposed site. There is no way whales can possibly flee the bombardment of these air guns with only a 30 minute warning, no less experiencing any such activity at all with the permanent damages done to them. Of course, I am totally opposed to any seismic activity but to not give the little bit of extra protection granted to the marine life by observers is critically unacceptable. Thank you for your consideration in this matter. [REDACTED]

[REDACTED]
Sent from my iPhone



ITP Young - NOAA Service Account <itp.young@noaa.gov>

Hilcorp night airguns

1 message

[Redacted]
To: "ITP.Young@noaa.gov" <ITP.Young@noaa.gov>

Tue, Sep 3, 2019 at 2:17 PM

Jolie Harrison
Chief, Permits and Conservation Division,
Office of Protected Resources, NMFS

September 2, 2019

Dear Jolie Harrison,

I have been conducting humpback whale surveys in Alaskan waters specifically in Kachemak bay, lower Cook Inlet, Kenai Fiords and Prince William Sound (PWS) since 1980 (presently under NOAA permit NO.21295 and the supervision of Amy Hapeman).

In the fall many whales congregate in Lower Cook Inlet and Kachemak Bay, some coming from the other areas described above. I am appalled to hear that Hilcorp Alaska LLC (Hilcorp) is planning seismic exploration that will start as soon as September 10. There are whales all over the place in the lower Cook Inlet right now.

Hilcorp has been authorized by NOAA to "take" marine mammals incidental to oil and gas activities in Lower Cook Inlet with the requirement that they stop the airguns if a marine mammal is within 5 miles of the ship. Now Hilcorp is requesting your authorization to operate at night. This will allow them to operate despite the presence of marine mammals. The reasoning is that the overall time they will be shooting the seismic airguns will be less. This is poor reasoning. Give the whales a rest. They can hear the airgun blasts alot farther away than the required 5 miles. It has been shown that marine mammals will change their behavior in response to seismic airguns up to 200 miles away (Risch et al 2012).

I urge you strongly to review Hilcorp's permit and their track record. They are considered disreputable here in Alaska. How well will the marine mammals really be monitored? We are facing great changes in the marine environment and this is a very bad time and place for this activity.

Operating at night with no ability to spot marine mammals is inexcusable. This is a pristine area and it is full of marine mammals, including the endangered Pacific Right whales and beluga whales, and the recently delisted humpback whales. In 1975 the State of Alaska bought back oil leases in Lower Cook Inlet. Why are you allowing this now?

Please do not authorize Hilcorp to operate at night (or at all) in Lower Cook Inlet.

Sincerely

[Redacted signature block]



ITP Young - NOAA Service Account <itp.young@noaa.gov>

Hilcorp testing

1 message

[REDACTED]
To: ITP.Young@noaa.gov

Sat, Aug 31, 2019 at 11:58 AM

Hello,

I would like to ask that you suspend Hillcorps " sound testing " at darkness due to the fact they will be unable to physically see any wildlife in the area of the testing.

[REDACTED]

[REDACTED]

Sent from my iPad



ITP Young - NOAA Service Account <itp.young@noaa.gov>

looking for marine mammals?1 message

Wed, Sep 11, 2019 at 2:03 PM

[REDACTED]
To: "ITP.Young@noaa.gov" <ITP.Young@noaa.gov>
Cc: Amy Hapeman - NOAA Federal <Amy.Hapeman@noaa.gov>

To whom it concerns,

I am deeply concerned...how could the observers on the oil exploration ship really be looking out for the many marine mammals of lower cook Inlet?

Hilcorp is very disreputable here in Alaska. Is anyone outside the industry monitoring them? I have spent my life studying and counting the humpback whales in this area under your permits.

I am also puzzled that the comment period ends on September 16th for Hilcorp being allowed to operate at night when they cannot be looking for marine mammals. However they started exploration today. Does that mean they can or cannot operate at night until the permit division reviews the comments and makes a decision?

These ships are huge and the waters of Cook Inlet can be dangerous. The citizens are depending on you.

sincerely looking forward to your response

[REDACTED]

[REDACTED]



ITP Young - NOAA Service Account <itp.young@noaa.gov>

Proposed amendment to allow night operations in Lower Cook Inlet by Hilcorp

Mon, Sep 2, 2019 at 2:53 PM

[REDACTED]
To: ITP.Young@noaa.gov

Jolie Harrison, Chief,
Permits and Conservation Division,
Office of Protected Resources,
National Marine Fisheries Service.

On August 30 2019 our principle investigator counted 72 sea otters, four humpback whales and 6 seals in a brief survey within a single track line between Seldovia and the Homer Spit. On Sept 1 we further counted and photo identified 5 different humpbacks in Kachemak Bay and a large pod of transient killer whales were spotted near Hesketh island. Obviously Lower Cook Inlet has abundant marine mammals feeding at this time.

Hilcorp is asking that they be allowed to conduct seismic blasting at night without observers. They are counting on these mammals moving away from the blast sound. I absolutely oppose this strategy and frankly don't care if it saves them money. We plan citizen monitoring of Hilcorp's seismic blasting during daylight hours because we have no faith in their permit agreements with NMFS based on 1. their previous record of ignoring environmental regulations 2. a rebuke issued by the State of Alaska and 3. the fact that at a recent public meeting in Homer, Hilcorp "wildlife specialist" and spokesperson and perennial oil company flak Beth Sharp lied about the effects of their proposed seismic testing on fish and marine mammals. We have no faith that they will follow NMFS guidelines requiring them to cease activities in the presence of marine mammals. We have no faith in their marine observers. And we have no faith in whatever their final assessment will be. As NMFS is perfectly aware a simple google search will turn up a minimum of 50 scientific papers linking damage from seismic testing to marine mammals and fish species. Hilcorp has a history of ignoring environmental rules. Now they want to conduct this seismic survey without oversight. I vehemently oppose granting a permit for night time blasting.

[REDACTED]
[REDACTED]
[REDACTED]
Sent from my iPad