

Three's a crowd: Killer whales attempt to evade few, but not several, boats

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ABSTRACT

Vessel traffic has been implicated as a potential contributing factor to the at-risk status of two killer whale populations in western Canada and the US. Relevant guidelines can be informed by conducting experimental impact assessments that allow animal response to guide vessel management. Two published experimental studies documented stereotyped avoidance responses by killer whales to boats, and opportunistic observations from these studies suggested a shift in avoidance behaviour when approximately 3 boats approached within 1000m. Our study was designed to test experimentally whether whales did, in fact, respond differently to approach by few (1-3) versus many (>3) vessels. Data were collected in summer 2004 in Johnstone Strait, British Columbia, using a theodolite to track positions of boats and individually identifiable whales. Experimental trials included 20-minute 'no-boat' and 20-minute 'boat' phases, with local whalewatching vessels serving as experimental boats, and during which data were collected continuously on the focal whale. Responses of the 16 adult male killer whales differed significantly between treatment levels (Wilcoxon's test $P = 0.0148$). Swimming paths became less direct when whales were approached by few boats, but whales increased their path directness when approached by many boats. Pooling both treatments would have masked these significant responses with strong statistical confidence (Wilcoxon's test $P > 0.999$), falsely suggesting that boat presence had no effect. Consistent with previous studies, inter-breath interval, swimming speed, angle between successive dives, and rates of surface-active behaviour did not differ significantly. The distinction between 'few' and 'many' boats was supported by opportunistic observations on 26 whales from the population of 216. Generalised Additive Models were used to control for effects of potentially confounding variables, and confirmed a non-linear relationship between the number of boats approaching within 1000m and whales' swimming path directness, with more detailed analyses confirming an inflection point at approximately 3 boats. We recommend that whalewatching guidelines be modified to address crowding, and urge caution when designing impact assessments that rely on a simple absence-presence framework, which can mask significant multivariate or non-linear responses. Interpreting biological significance of null findings from impact assessments is problematic and therefore statistical power, experimental design, and appropriateness of response variables must be considered.

INTRODUCTION

A growing number of studies have examined the impact of human activities on the behaviour, energetics and reproductive success of a variety of wildlife taxa. Generally, these studies demonstrate that there is an "observer effect" inherent in wildlife viewing, such that our simply being in the places where animals need to be may disrupt the natural activities of the animals we seek to watch. In some cases, these effects are striking, and obvious both to those who aim to watch wildlife, as well as researchers and managers aiming to reduce the impact of tourism. Birdwatchers know this, and curtail their movements in order to reduce their impact on the birds they watch, but also simply to prevent the birds from fleeing. In the coastal environment, an approaching boat can cause hauled out seals to enter the water (Suryan and Harvey 1999). Diving seals make for poorer viewing than those ashore, so it is in both the animals' and the tourists' best interests that guidelines be developed to reduce the probability of such a startle response. A much more subtle example is our disruption of wary predators, even when we are not physically there. The presence of human-built trails and roads in Alberta, Canada has been shown to cause an increase in the "tortuosity" of paths followed by wolves (*Canis lupus*) (Whittington *et al.* 2004). Thus even without posing absolute barriers, these trails and roads altered wolf movement across their territories. Such a finding may seem obvious in hindsight, but it is a very useful finding in that it gives a mechanism for: (1) defining what makes good wolf habitat to protect and how to protect it (*i.e.*, complete closure areas) and (2) linking short-term behavioural responses to fitness. In the bird, seal and wolf examples, we see that avoidance responses of wildlife to people range from the blatantly obvious to the incredibly subtle, but they are all real, and they can all carry costs to the animal. Human activities that make wolves adopt more circuitous routes to reach a destination can cost them energy if the increased travel time replaces time budgeted for resting. The effect is exacerbated if the increased travel time comes at the expense of time that otherwise would have been spent feeding, as is the case for grizzly bears (White *et al.* 1999).

Whalewatching is a growing industry worldwide (Miller 1993, Nelson 1994, Hoyt 1997) and managers have implemented whalewatching guidelines or regulations in dozens of countries (IWC 2005). Many of these recommendations came from the whalewatchers themselves, who identified early on in their industry that they had the potential to influence whale behaviour, and wanted to minimise that tendency (*e.g.*, JSKWC 1991, 1996). Indeed, anthropogenic activity has been cited as a causal factor in the at-risk status of two killer whale

populations in the northeast Pacific (Baird 2001). However, assessing the effects that our presence has on these cryptic species is difficult. We can expect avoidance responses of cetaceans to be subtle ones, because the animals have to surface to breathe, and because, unlike flying birds, whales can generally not move quickly enough to escape people in powerboats. Despite the difficulty in conducting impact assessments on free-ranging marine species, a number of studies have quantified the impacts of vessel traffic on cetacean behaviour (e.g., Kruse 1991, Lusseau 2003, Williams *et al.* 2002ab, Scheidat *et al.* 2004), activity and energetic budgets (Lusseau 2004, Williams 2003), and possibly reproductive success (Bejder 2005). One feature common to these studies is that they report subtle behavioural cues that cetaceans displayed when approached by boats. Taken as a whole, the literature in this field presents overwhelming evidence that behavioural responses of cetaceans to boats are generally small, but real.

Thus, the detection of avoidance responses by cetaceans will benefit from carefully designed control-exposure experiments and land-based observers. In Johnstone Strait, British Columbia (BC), Canada, the research and whalewatch communities work closely together to draft common-sense guidelines, to test their efficacy experimentally, and update them in an iterative process. Commercial whalewatch operators recommended that boaters approach whales slowly, from the side, and to parallel the animals' path at distances no closer than 100m (JSKWC 1996). In 1995, the whalewatch community began cooperating with land-based researchers to provide experimental control periods to quantify the effects of their industry on whales.

Two experimental studies were carried out in 1995 to test the relevance of local whalewatching guidelines. The first experimental study measured how whales responded to one boat following the guideline by approaching the animal slowly, from the side, and paralleling the whale's path at a 100m distance (Williams *et al.* 2002a). The second measured how whales responded to one boat violating local whalewatching guidelines by 'leapfrogging' (Williams *et al.* 2002b). Both studies revealed that whales, on average, attempted to increase their distance from boats by deviating from the smoother, more predictable path that they were following immediately before the experimental approaches. Opportunistic observations revealed that the magnitude of this horizontal avoidance response increased as boats approached closer than 100m.

The consistency between these experimental and opportunistic studies adds to a mounting body of evidence suggesting that northern resident killer whales display stereotyped responses to evade boats. Therefore, summary statistics describing the tortuosity of whale swimming paths can be considered a response variable that may be used as a proxy representing a killer whale's tolerance for various approach distances and activities. Both studies though, measured how whales responded to activities of a single boat. Killer whales, however, are often approached by more than one boat – in the case of southern residents, the whales occasionally attract as many as 120 vessels at once (Koski 2004). How should we manage boat *number*, as opposed to *proximity*, around whales? How do we inform that process by designing experiments to measure how whales respond to simultaneous approach by many vessels?

We can start by examining the correlations between boat number and whale behaviour that are reported from opportunistic observations. Opportunistic observations allow insights into killer whale behaviour under traffic conditions that would have been difficult to replicate experimentally, provided that we use statistical models to account for confounding effects. Studies that use a dual nature of data collection allow the causal relationships identified by experimental approaches to be compared with trends in whale behaviour across a wide range of traffic conditions. One such comparison was reported from the opportunistic component of the first study (Williams *et al.* 2002a), which found an apparent inconsistency between whale response to boat number and proximity. Whales in that study showed a decrease in path directness as boats approached more closely, but an increase in path directness as boat number increased. The findings suggested that killer whales responded differently to a few boats than to many, with the difference between few and many occurring when approximately 2-3 boats approached within 1000m of the whale. The authors speculated that:

"An irregular path may be a useful avoidance tactic with a single boat, but ineffective with more than one. In a multiple-vessel scenario, a dive that takes a whale farther from one boat may bring it closer to another. ... This compromise deserves further attention in the form of a multiple-vessel experiment."

Recent studies on other species support this hypothesis. These studies show a shift in avoidance strategies depending on number of boats present (Bejder 2005, Lusseau 2005); from short-term evasion strategies to long-term area avoidance. The at-risk status of killer whales (Baird 2001), and the increasing level of boat traffic around them (Foote *et al.* 2004) create an urgent need for science to inform management of boat number around whales. In other words, it is time to conduct that proposed multiple-vessel experiment to address the whales' "compromise" between responding to boat proximity and boat number (Williams *et al.* 2002a).

The primary goal of our study was to test whether resident killer whales responded differently to experimental approach by few boats than to many boats. Our experiment was designed to test the hypothesis raised in the previous study (Williams *et al.* 2002a): that whale responses to experimental approach by few boats (1, 2 or 3)

would differ from their response to experimental approach by more than three boats. Our secondary goal was to provide a quantitative description of whale behavioural data collected opportunistically – that is, across a wider range of traffic conditions (boat number) and with a larger sample of individuals than could be obtained practically using experimental approaches.

METHODS

Field Methods

We collected data from a cliff opposite Robson Bight-Michael Bigg Ecological Reserve (Johnstone Strait, British Columbia, Canada, 50°30'N, 126°30'W), from 1 July through 11 September 2004. We used an electronic theodolite to record locations of boats and whales via a laptop computer equipped with custom software (THEOPROG: see Williams 2002a, 2002b) in real time to reduce the likelihood of transcription errors, and to allow us to calculate real-time estimation distances.

As whales entered the study area, we selected a focal whale. We identified individual whales by referencing photo-identification catalogues (Bigg *et al.* 1990, Hammond *et al.* 1990, Ford *et al.* 2000), and chose a distinctive focal animal that could be tracked continuously for at least 15 minutes. Once a focal whale was chosen, we used the theodolite to ‘mark’, or record the position of the focal at each surfacing (*i.e.*, the time and position of each breath was recorded). We recorded vessel positions as frequently as possible to plot its trajectory with respect to that of the focal animal.

We subsequently refer to each record of at least 20-minutes of observation of a focal animal as a ‘track’. We summarised boat and whale data for each track such that each track represented only one observation in the analyses.

Experimental trials versus opportunistic observations

The local whalewatchers had agreed to participate in experiments prior to the beginning of our study. At the outset of an experimental treatment, we requested whalewatchers’ cooperation in creating an experimental control phase by avoiding the focal whale for approximately 20 minutes, and an experimental treatment phase by approaching the focal whale for a typical whalewatching session for an additional 20 minutes. No attempt was made to direct traffic other than the experimental vessel(s) during the experimental trial. However, all vessel traffic within 1000m of the focal was recorded by the theodolite at least twice, even those boats not engaged in whalewatching activity. The experimental control section referred to the data collected over a continuous 20-minute period during which no boat approached the focal whale within 1000m. The treatment section referred to the 20-minute minimum period immediately following this control period in which at least one experimental vessel approached within 1000m of the same focal for the duration of the treatment phase. In all cases, each animal served as its own control and the control section always preceded the treatment section.

We summarised boat traffic variables for every experimental trial for *post-hoc* categorization into one of two treatment levels based on the number of boats that approached the whale during the experimental section: that is, approach by *few* (1-3) versus *many* (>3) boats. Note that a LOWESS curve fitted to scatterplots of opportunistic data that correlated path directness with boat number (Williams 2002a) suggested that whale behaviour changed when more than two to three boats approached whales within 1000m. Consequently, we used that previous study to design the treatment levels in the present experiment.

On most occasions, however, we collected behavioural data under conditions that offered no ability to manipulate traffic around the focal animal. We called these “opportunistic observations.” In such cases, whales either entered the study area already accompanied by boats, or were observed in the absence of boats on occasions when no boat was available to serve as the experimental treatment. During these occasions, we selected focal animals and measured whale behaviour and boat activity in precisely the same way as described for the experimental trials.

Analysis Methods

For each track, we interpolated the number of boats within 100, 400 and 1000m of the focal whale at the time of each surfacing. Then, we recorded the maximum, inclusive count for each of these radii for each track. We refer to these candidate explanatory variables subsequently as MAX100, MAX400 and MAX1000, which represent the maximum number of boats ever to approach the whale during an observation session to within 100, 400 and 1000m, respectively. Similarly, we recorded the minimum distance ever observed between the focal whale and any boat, referred to subsequently as the point of closest approach (PCA).

We calculated the response variable, the *directness index*, using methods described in earlier experimental studies published on this population (Williams et al. 2002ab). The directness index measures path predictability, and is generated by dividing the distance between end-points of a path by the cumulative surface distance covered during all dives and multiplying by 100. The directness index is a ratio of the diameter of a circuitous path to its perimeter, and ranges from zero (a circular path, where the whale ends up exactly where it started) to 100 (a straight line).

We analysed our data in three ways:

1. Categorical analysis of experimental data with two treatment levels: few vs. many boats

We used paired, non-parametric tests to assess significance of behavioural responses of whales to our experimental treatments. For each trial, we calculated the whale's response to experimental approach. The response data were then placed in two categories: a sample of responses of whales exposed to approach by few (1-3) boats, and a second sample of responses of whales to many (>3) boats approaching. We used, the Mann-Whitney U test (Zar 1996) to assess whether the treatment variable influenced the ranking of each observation of whale behavioural response.

2. Analysis of experimental trials with one treatment level: boat presence

We then reran the analyses ignoring the boat number grouping variable, thereby pooling data in an absence-presence framework. We used the Wilcoxon matched-pairs signed-ranks test.

3. Descriptive analysis of opportunistic data

We used Generalised Additive Models (GAMs) to describe variation in whale swimming path directness as linear or non-linear functions of both traffic-related and non-traffic-related candidate predictor variables, using a set of objective criteria for adding or dropping terms. GAMs use splines for interpolating between observed data by placing "knots" automatically at observed data. We used package mgcv for R to determine the shape of the relationships between predictors and response using thin-plate regression splines (Wood 2003). The optimal degree of flexibility that can be justified by the data is estimated in a maximum likelihood framework, while penalizing the model for overparameterization or oversmoothing (Wood 2000). The higher the estimated degrees of freedom (edf) given to any term in the model, the more flexible that relationship was estimated to be.

We considered the following potential explanatory variables for inclusion in the model: Month, Time, Age, Sex, PCA, MAX100, MAX400, and MAX1000. Factor variables were entered as linear or grouping terms. Continuous variables were entered as candidates for smoothing (s(x)) by mgcv.

The following summarises our backward stepwise model specification procedure, which followed the framework proposed by Wood (2001):

1. We fitted a fully saturated model to the data: $\{DI \sim \text{Month} + \text{Time} + s(\text{Age}) + \text{Sex} + s(\text{PCA}) + s(\text{MAX100}) + s(\text{MAX400}) + s(\text{MAX1000})\}$ with the default degree of smoothing (10 knots, 9 edf).
2. We assessed model fit using the summary.gam and plot.gam functions in mgcv, to show coefficients, GCV score, explanatory power (deviance explained) and fit (residual plots). Term-wise P-values for each coefficient presuppose that the other terms are in the model.
3. For each linear term, we assessed whether the parameter coefficient (slope) was near 0 and the significance term near 1. If so, we removed the term to see if the GCV score decreased and the explanatory power of the model increased. If so, we dropped the term from the model. If no marked improvement was detected by removing the term, then we retained the term in the model.
4. For each smooth model term, we examined whether the estimated number of degrees of freedom was near 1. We evaluated whether the 95% confidence intervals for that term included zero across the range of observations. If so, we dropped the term temporarily, to see if the GCV score dropped and the explanatory power of the model increased.
5. We dropped a term from the final model if it satisfied all three of the conditions in step 4. If the first criterion was met ($\text{edf} \approx 1$), but not the other two, then we replaced the smooth term with a linear one. If replacing the smooth term with a linear one reduced the model's explanatory power, then we retained it as a smooth term with $\text{edf} \approx 1$.

Finally, we looked at the relationship between the animals' path directness and the number of boats. We conditioned on the selected model describing variation in path directness index, but placed knots

manually along the x-axis describing variation in boat number. This allowed us to assess objectively where the curve changed shape. We used AIC to determine objectively where a node ought to be placed.

RESULTS

Effort and sample size

We collected data on 72 of the 73 days between 1 July and 11 September 2004, representing 792 hours of search effort. Whales were present for at least part of the day on 60 of these 73 days. In total, our study represents 73.8 hours of continuous observation (*i.e.*, of tracks >20 minutes in length) of focal animals using the theodolite. We conducted a total of 16 experimental trials. Despite good cooperation from whalewatchers, non-whale-oriented traffic (particularly during a 10-day commercial fishing opening) disrupted dozens of otherwise-suitable control data collection periods. Opportunities to conduct experimental approaches became rare enough that we decided to restrict our trials to males. We excluded one experimental approach of a female from the analyses. We collected a total of 140 opportunistic tracks on 26 focal whales. Details on sample size by age, sex, and individual identity are provided in Appendix 1. Of the 16 experimental trials, focal animals were approached (within 1000m) on 8 occasions by 1-3 boats, and by 4-17 boats on the remaining 8 occasions.

Responses of whales to experimental approach by few versus many boats

Figure 1 shows the distribution of the response variable during the experimental trials. This figure should be interpreted with caution, because our statistical analyses used non-parametric tests based on ranks, rather than raw data. Whales approached within 1000m by few (1-3) boats decreased their index of path directness by 14.0 points relative to that observed during preceding, control conditions (from a mean of 82 to a mean of 67.9). Those approached within 1000m by many (4-17) boats actually increased the directness of their paths on average by 12.9 points than during control conditions (from a mean of 70.9 to a mean of 83.8). These responses differed significantly between treatment levels (Mann-Whitney U test, two-tailed $p=0.0148$).

We used these data and rearranged the formula for calculating directness index to estimate the distance a whale would have to travel along a circuitous route to cover 100m of straight-line distance. In the absence of boats (*i.e.*, a whale displaying a path directness index of 82), a whale would have to swim 122m (*i.e.*, $100/0.82$) along a circuitous path to cover the straight-line distance of 100m. The same whale adopting a mean swimming path directness index of 67.9 would have to swim 147m along a circuitous path to cover the 100m straight-line distance. The total distance a whale would have to travel in the presence of 1-3 boats represents a 20.7% increase over the total distance it would swim to cover the same ground in the absence of boats. Whales approached by 4-17 boats would actually increase the efficiency of their paths: from 141 (control) to 119m (treatment) required to make 100m of headway.

Responses of whales to experimental approach by any number of boats (*i.e.*, contrasting whale behaviour during absence and presence of boats)

Rather than looking at how cetacean behaviour varies across a number of boats, vessel impact studies often contrast behaviour in a presence-versus-absence framework. The following summarises what we would have found had we ignored the confounding effect of boat number, in a simple absence-presence contrast of the behaviour of whales that were approached by any number of boats (from 1-17 boats approaching the whale within 1000m) and the behaviour of whales in the absence of boats. In the absence of boats, the mean path directness index of focal whales was 76.4. When approached by boats, the mean directness index was 75.8. This difference was not statistically significant (Wilcoxon matched-pairs signed-ranks test, two-tailed $p>0.9999$).

Results of generalised additive model (GAM) fitting

The model performed reasonably well at explaining the variability in the whale behaviour as a combined function of boat traffic-related (*MAX400* and *MAX1000*) and unrelated (*MONTH* and *AGE*) variables (Table 1). Explanatory power of the selected model was good: the model explained 67.7% of the deviance (*i.e.*, residual sums of squares, Wood 2001). In general, paths became more direct as the season progressed. Paths also became more direct as number of boats within 400m increased. Age and number of boats within 1000m entered the selected model as smooth terms, and Figure 2 shows how directness index related to the smoothed components of these two variables.

The P-values listed are approximate, and should be interpreted with caution (Wood 2001), because they are conditional on all other terms being in the model, the accuracy of the smoothing parameters, and any scale parameter estimate. As a result, some model terms were retained even though their P-values were >0.05 , for the

reasons outlined in the description of the GAM-fitting process in the methods section. Note that poor model fit was found (as evidenced by patterns in residual plots) when specifying a quasi or Gamma distribution in path directness data on a 0-100 scale. This was addressed by standardising the data – expressing them as a proportion by dividing by 100, so that 0 represented a circular path and 1 a straight line. The response was then modeled in a quasibinomial (maximum likelihood) framework with a logit (1/0) link, which alleviated the patterns that were previously evident in the residual plots.

Table 1. Summary of the selected GAM describing variation in directness index.

Family: quasibinomial

Link function: logit

Formula:

DI ~ MONTH + s(AGE) + MAX400 + s(MAX1000) - 1

Parametric coefficients:

	Estimate	std. err.	t ratio	Pr(> t)
MONTH	0.15381	0.01857	8.281	1.1382e-13
MAX400	0.084724	0.09028	0.9384	0.34972

Approximate significance of smooth terms:

	edf	chi.sq	p-value
s(AGE)	2.04	3.0592	0.22666
s(MAX1000)	2.784	5.1208	0.14729

Deviance explained = 67.7%

n = 140

The terms we dropped from the model included SEX, TIME, PCA and MAX100, as well as the intercept term.

Using bsplines to verify the node between ‘few’ and ‘many’ boats

We conditioned on the selected model describing variation in path directness index, but manually placed knots at every observed value of MAX1000. A plot of AIC of the model describing path directness index against the knot position is shown (Figure 3). AIC was lowest when the knot was placed at MAX1000=3. This provides an objective identification of the optimal placement of the node separating few from many boats.

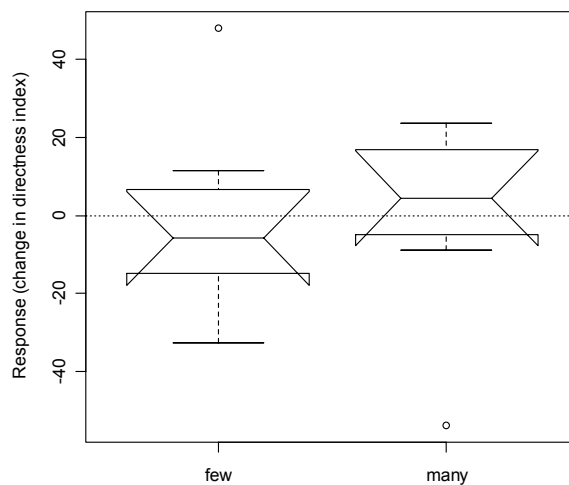


Figure 1. Response by focal male killer whales to experimental approach by few (1-3) versus many (4-17) boats within 1000m. Response represents the difference in path directness index between the treatment and control phase.

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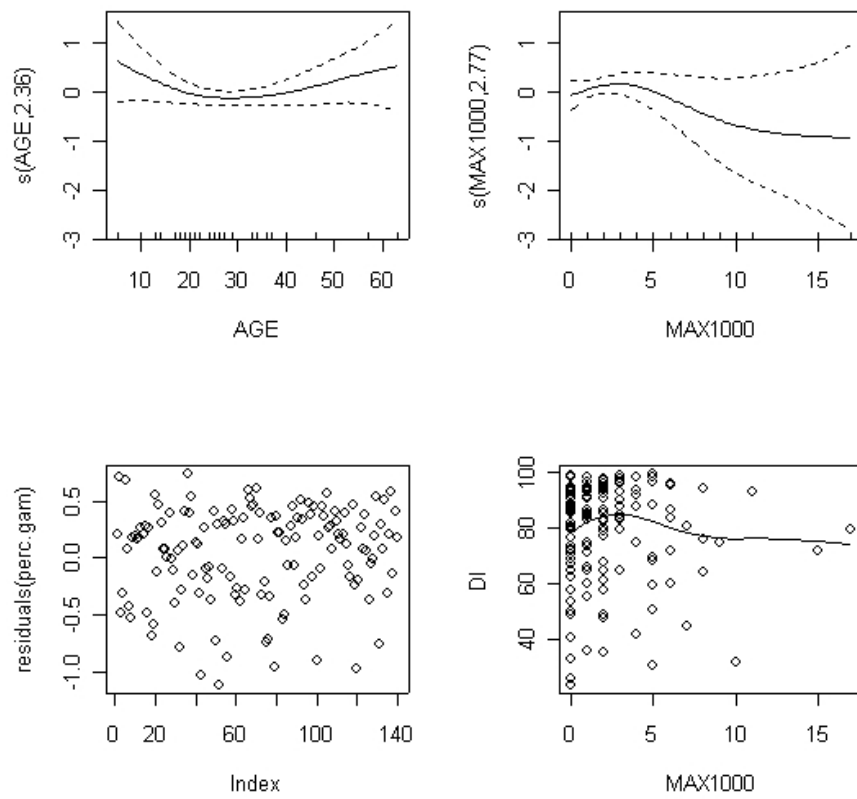


Figure 2. Smooth terms in the selected GAM (top), residuals (lower left) and the raw data for boat number and directness index (lower right).

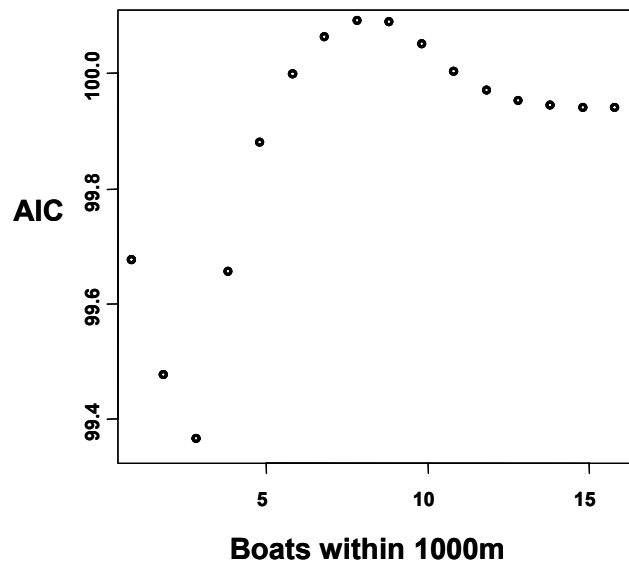


Figure 3. The x-axis represents the point at which a knot was placed manually, when fitting a bspline through the opportunistic data, conditional on the selected model (Table 1 and Figure 2). The y-axis represents the AIC value of that model. AIC was lowest when a knot was placed manually at MAX1000=3.

Discussion

Our results add to a growing body of experimental and opportunistic studies that suggest that northern resident killer whales show a suite of stereotyped responses to boat traffic, but our new findings indicate that these stereotyped avoidance responses began to diminish as the number of boats exceeded a threshold of three boats within 1000m of the whale. Northern resident killer whales approached by experimental boats tended to adopt paths that were less predictable (*i.e.*, less direct on the scale of a 20-min observation period) than that observed during the preceding control phase during which no boats were present within 1000m. This pattern has been demonstrated experimentally now for whales responding to: one boat following whalewatching guidelines (*i.e.*, paralleling at 100m; Williams *et al.*, 2002a); one boat violating whalewatching guidelines (*i.e.*, leapfrogging within 100m; Williams *et al.*, 2002b); and to a few (1-3) boats approaching simultaneously within 1000m (Figure 3). But whales that were approached by multiple boats (4-17) adopted paths that were actually *more* direct than they were during preceding control conditions.

A cautionary note for other control-impact experimental design studies emerged from our binary analyses that ignored this confounding effect of boat number. When we contrasted the path directness under all no-boat conditions with those under all boat conditions (that is, by pooling the treatment levels), the average response did not differ significantly from zero (Wilcoxon matched-pairs signed-ranks test, two-tailed $p > 0.9999$). Ignoring the multivariate nature of responses might have led a researcher to falsely conclude, with a great deal of statistical confidence, that boats do not alter the behaviour of killer whales, when in fact, boats altered behaviour in two opposing ways. A classic flaw in ecological studies is an attempt to describe a non-linear relationship using a linear model. The model will fit the data poorly, and one will falsely conclude, with a high degree of statistical certainty, that there is no effect. In fact there is one, but the effect is just not a linear one. We believe that something similar may occur in overly simplistic vessel impact studies, where presence/absence analyses may mask effects when the average of a multivariate or non-linear response may look like no response. Simple presence-absence studies that combine a variety of multivariate responses may produce misleading results.

We believe that carefully designed studies are needed to provide management advice for what whales can tolerate in terms of boat number, saturation of daytime hours with boat presence, or vessel proximity. While both populations of resident, fish-eating killer whales are listed under Canada's Species at Risk Act, boats are of particular concern for the *Endangered* status of the southern resident population (Baird 2001). Our study offers important guidance for experimental design of appropriate vessel impact assessments for southern resident killer whales, and provides, for the first time, an unbiased estimate of the number of boats at which northern resident killer whales shift avoidance strategies. Here we show that this effect starts at only 3 boats within 1000m of killer whales.

Unfortunately, the more critically endangered southern resident killer whales are rarely surrounded by fewer than 3 boats during the spring and summer months (when they return to Washington State and southern BC waters to forage on salmon). The flotilla of vessels following southern resident killer whales at any one time routinely numbers 18-26 boats, but the whales sometimes attract as many as 120 vessels at once (Koski 2004). Our study demonstrated a threshold at 3 boats, beyond which point the whales shifted or abandoned their avoidance responses. *What are the implications for an endangered population that is surrounded by six times that threshold number over its entire home range for several months each year?*

At this point, no one can say. Assessing the role of boat traffic on southern residents directly is problematic, primarily because there are too many boats to allow experimental control over studies¹. Efforts to control traffic via time-area closures along the west side of San Juan Island during have been unsuccessful and provided only brief 'no-boat' situations. In addition, the unpredictability of thousands of recreational boaters unaware of the exclusion zone significantly limits opportunities for controlled experiments with the southern resident population. We reasoned that working with the parapatric northern resident population would allow us to conduct an experimental impact assessment by proxy that could be used to inform the design of future experiments to be conducted on southern residents.

It remains to be tested whether our experiment did serve as an appropriate proxy for the southern resident case study, both in terms of the comparability in tolerance levels of the whales that were studied, and in terms of the boat traffic to which the whales were exposed. We address the first component first: were northern residents a suitable proxy for southern residents? Certainly, we do not advocate the direct extrapolation of our results to management of human activities around southern residents. Concerns have been raised about the potential for

¹ Fisheries and Oceans Canada noted that "[i]n 2001, the M3 program observed an average of 18 vessels (commercial and private) around whales at any time in the Victoria/Haro Strait area from dawn to dusk" and that "[u]p to 50 vessels actively viewing whales have been observed in the Victoria/Haro Strait area at any one time. (DFO 2002)"

southern residents to have habituated to whalewatching activity after exposure to such high boat traffic levels. But the fact that the avoidance responses we report here have persisted in northern residents after 25 years of whalewatching suggests that habituation, if it is occurring, is incomplete. The second component is whether our many-boats experimental treatment level contained sufficient numbers of boats to simulate the traffic that southern resident killer whales experience. Answering this question will require some alteration in the way that data are currently collected on southern resident killer whales and the boats that watch them. As it stands now, reports include the maximum number of boats seen within an observer's field of view, rather than within a measured radius around the whale. This makes it difficult to compare between northern and southern resident killer whale impact assessments, or even between reports made on southern resident killer whales at different locations (compare for example the DFO (2002) estimate of an average of 18 boats (maximum 50) "around whales" in Victoria/Haro Strait, and the Soundwatch (Koski 2004) estimate of an "average" of 18-26 boats (maximum 120)). Studies funded by the US National Marine Fisheries Service are currently examining these concerns, and are measuring boat traffic and southern resident killer whale behaviour in a way that will allow direct, quantitative comparisons between the two populations.

Until such a comparison can be made, we are left to speculate on what may happen when we exceed the threshold of disturbance that the animals are willing to tolerate. Trites *et al.* (this meeting) examined tens of thousands of observations of northern resident killer whales in Robson Bight (Michael Bigg) Ecological Reserve between 1991 and 1994. They found that as the number of boats entering a particular zone of the reserve increased, so too did the probability that whales would leave that zone. Displacement may be ultimate sign that anthropogenic stressors have passed some point beyond which it is no longer advantageous for whales to tolerate the disturbance. We know that killer whales are susceptible to increased ambient noise levels (Bain and Dahlheim 1994, Erbe 2002, Foote *et al.* 2004). On a very fine scale, they make demonstrable efforts to evade experimental boats that approach them (Williams *et al.* 2002ab, this study). When acoustic harassment devices were introduced to a peripheral part of the northern resident killer whales' range, they abandoned the area for a period of years (Morton and Symonds 2002), and have yet to recolonise the area (A.M. Morton, pers. comm.). It remains to be seen whether killer whales would be displaced from critical, as opposed to peripheral, habitat, but the resident killer whales continue to return to their core areas of Johnstone and Haro Straits each year, in spite of growing numbers of boats. If there is a tipping point of disturbance beyond which whales abandon their core habitat, we have not yet reached it for either population. A new interpretation has been proposed by Bejder (2005) though, which urges us to consider these short-term avoidance responses in a longitudinal perspective: it could well be that disturbance acts as an evolutionary selection force, through which less tolerant animals (or matriline in the case of resident killer whales) are excluded from core areas, and consequently, excluded from our vessel impact studies. Indeed, in both the southern and northern resident killer whale populations, some matriline spend much more time in the core whalewatching areas than others. It is not known whether these reflect differences in tolerance of disturbance, or if they relate to territoriality or maternally linked cultural specialisations in foraging behaviour. This may not be knowable, because the whalewatching industry and the research began at approximately the same time in both locations. In any event, the examples presented here of habitat displacement represent situations in which killer whales could avoid the disturbance by changing their residency patterns on a small scale, a result that has been seen with bottlenose dolphins in Doubtful Sound, New Zealand (Lusseau 2005). Southern resident killer whales are followed by a source of disturbance where changing their residency pattern would separate the population from a primary food source. Therefore, we must create situations in which killer whales can avoid disturbance and carry out activities essential to population survival and recovery such as mating and foraging. One way to mitigate some of these effects might be to identify areas of demonstrated importance to the whales, and to close them to boats altogether (Lusseau and Higham 2004).

We will also need sensible guidelines though for managing whalewatching outside those protected areas. Previous studies have provided some guidance on the need to set minimum approach distances no closer than 100m (Williams *et al.* 2002a) and to avoid leapfrogging (Williams *et al.* 2002b). This study suggests that on average, we should also aim to reduce crowding of whales by keeping the number of boats within 1000m of northern resident killer whales to a maximum of three. But perhaps we could combine recommendations on approach distance and boat number in the same guideline. Beale and Monaghan (2004) present a model in which cliff-nesting birds perceive people as "predation-free predators", where the perception of predation risk is a function of the number and proximity of observers. The authors suggest that setting one approach distance is insufficient when the numbers of visitors is variable. They recommend that large groups of people be kept farther from nesting areas of vulnerable species, or that approach distances be determined for the largest group of people that are likely to visit the site.

In the killer whale case, this would lead whalewatching guidelines to have different approach distances for different numbers of boats, or to setting the overarching set-back distance to that distance we would wish all boaters to keep when 120 boats are trying to approach a whale at once.

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LITERATURE CITED

- Bain, D. E. and Dahlheim, M. E. 1994. Effects of masking noise on detection thresholds of killer whales. In *Marine mammals and the Exxon Valdez*. Edited by T. Loughlin. San Diego, Academic Press: 243-256.
- Baird, R. W. 2001. Status of killer whales in Canada. *Canadian Field Naturalist* **115**(4): 676-701.
- Beale, C. M. and Monaghan, P. 2004. Human disturbance: people as predation-free predators? *Journal of Applied Ecology* **41**:335-343.
- Bejder, L. 2005. Linking short and long-term effects of nature-based tourism on cetaceans. PhD Thesis, Dalhousie University, Halifax, Nova Scotia, Canada.
- Bigg, M. A., Olesiuk, P. F., Ellis, G. M., Ford, J.K.B. and Balcomb, K.C. 1990. Social organization and genealogy of resident killer whales (*Orcinus orca*) in the coastal waters of British Columbia and Washington State. In Hammond, P.S., Mizroch, S. A. and Donovan, G. P. 1990. Individual recognition of cetaceans: Use of photo-identification and other techniques to estimate population parameters. Cambridge: International Whaling Commission (Special Issue 12).
- DFO. 2002. Protecting Canada's Marine Mammals: Proposed Regulatory Amendments. Marine Mammal Bulletin December 2002. Available from: http://www-comm.pac.dfo-mpo.gc.ca/pages/consultations/marinemammals/documents/bulletin-dec02_e.htm.
- Erbe, C. 2002. Underwater noise of whale-watching boats and potential effects on killer whales (*Orcinus orca*), based on an acoustic impact model. *Marine Mammal Science* **18**(2): 394-418.
- Foote, A. D., R. W. Osborne, and A. R. Hoelzel. 2004. Whale-call response to masking boat noise. *Nature* **428**:910.
- Ford, J. K. B., Ellis, G. M. and Balcomb, K. C. 2000. Killer whales. 2nd Ed. Vancouver, UBC Press.
- Hammond, P. S., Mizroch, S. A. and Donovan, G. P. 1990. Individual recognition of cetaceans: Use of photo-identification and other techniques to estimate population parameters. Cambridge: International Whaling Commission (Special Issue 12).
- Hoyt, E. 1997. The potential of whale watching in Europe. Bath, Whale and Dolphin Conservation Society.
- International Whaling Commission (IWC). 2005. *Journal of Cetacean Research and Management* **7** (Supplement).
- Johnstone Strait Killer Whale Committee (cited as JSKWC). 1996. Johnstone Strait Whale Watching Guide. BC Min. of Environment, Land and Parks and Dept. of Fisheries and Oceans.
- Johnstone Strait Killer Whale Committee (cited as JSKWC). 1991. Background report. BC Min. of Environment, Land and Parks and Dept. of Fisheries and Oceans.
- Koski, K. 2004. The Soundwatch Boater Education Program: trends in vessel traffic with southern resident killer whales. The Whale Museum, Friday Harbor, Washington.
- Kruse, S. 1991. The interactions between killer whales and boats in Johnstone Strait, B.C. In *Dolphin Societies: discoveries and puzzles*. Edited by K. Pryor and K. S. Norris. Berkeley, University of California Press: 149-159.
- Lusseau, D. 2003. Male and female bottlenose dolphins *Tursiops* sp. have different strategies to avoid interactions with tour boats in Doubtful Sound, New Zealand. *Marine Ecology-Progress Series* **257**:267-274.
- Lusseau, D. 2004. The hidden cost of tourism: detecting long-term effects of tourism using behavioral information. *Ecology and Society* **9**:2.
- Lusseau, D. 2005. The residency pattern of bottlenose dolphins (*Tursiops* spp.) in Milford Sound, New Zealand, is related to boat traffic. *Marine Ecology Progress Series* **295**:265-272.
- Lusseau, D. and Higham, J.E.S. 2004. Managing the impacts of dolphin-based tourism through the definition of critical habitats: the case of bottlenose dolphins (*Tursiops* spp.) in Doubtful Sound, New Zealand. *Tourism Management*. **25**(6):657-667.
- Miller, M. L. 1993. The rise of coastal and marine tourism. *Ocean and Coastal Management* **20**(3): 181-199.
- Morton, A. B. and Symonds, H. K. 2002. Displacement of *Orcinus orca* (L.) by high amplitude sound in British Columbia, Canada. *ICES Journal of Marine Science* **59**: 71-80.
- Nelson, J. G. 1994. The spread of ecotourism - some planning implications. *Environmental Conservation* **21**(3): 248-255.
- Scheidat, M., C. Castro, J. Gonzales & R. Williams. 2004. Behavioural responses of humpback whales (*Megaptera novaeangliae*) to whalewatching boats near Isla de la Plata, Machalilla National Park, Ecuador. *Journal of Cetacean Research and Management* **6**(1): 63-68.

- 517 Suryan, RM, and JT Harvey. 1999. Variability in reactions of Pacific harbor seals, *Phoca vitulina richardsi*, to disturbance. *Fish. Bull.* **97**:
518 332-339.
- 519 Trites, A.W., Hochachka, W.M., Carter, S.K., Wong, M.M. and Williams, R. This meeting. Boats displace killer whales from a marine
520 protected area.
- 521 White, D., Kendall, K. C. and Picton, H. D. 1999. Potential energetic effects of mountain climbers on foraging grizzly bears. *Wildlife Society*
522 *Bulletin* **27**(1): 146-151.
- 523 Whittington, J., C. C. St. Clair, and G. Mercer. 2004. Path tortuosity and the permeability of roads and trails to wolf movement. *Ecology and*
524 *Society* **9**(1): 4. [online] URL: <http://www.ecologyandsociety.org/vol9/iss1/>
- 525 Williams, R. 2003. Cetacean studies using platforms of opportunity. PhD Thesis. University of St Andrews, St Andrews, Scotland, UK.
- 526 Williams, R., Trites, A. W. and Bain, D. E. 2002a. Behavioural responses of killer whales to whale-watching traffic: opportunistic
527 observations and experimental approaches. *Journal of Zoology* **256**: 255-270.
- 528 Williams, R., Bain, D.E., Ford, J.K.B. and Trites, A.W. Behavioural responses of male killer whales to a 'leapfrogging' vessel. 2002b.
529 *Journal of Cetacean Research and Management* **4**(3): 305-310.
- 530 Wood, S. N. 2003. Thin-plate regression splines. *J. R. Statist. Soc. B* **65**(1): 95-114.
- 531 Wood, S. N. 2001. mgcv: GAMs and Generalized Ridge Regression for R. *R News* **1**(2): 20-25.
- 532 Wood, S. N. 2000. Modelling and Smoothing Parameter Estimation with Multiple Quadratic Penalties. *Journal of the Royal Statistical*
533 *Society, Series B.* **62**: 413-428.
- 534 Zar, J. H. 1996. *Biostatistical Analysis*. New Jersey, Prentice Hall.