Passive acoustic monitoring of the decline of Mexico's critically endangered vaquita

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Abstract

The vaquita (*Phocoena sinus*) is the world's most endangered marine mammal with ≈245 individuals remaining in 2008. This species of porpoise is endemic to the northern Gulf of California, Mexico, and has historically suffered population declines from unsustainable bycatch in gillnets. An illegal gillnet fishery for an endangered fish, the totoaba (*Totoaba macdonaldi*), has recently resurged throughout the vaquita's range. The secretive but lucrative wildlife trade with China for totoaba swim bladders has probably increased vaquita bycatch mortality, but by an unknown amount. Precise population monitoring by visual surveys is difficult because vaquitas are inherently hard to see and have now become so rare that sighting rates are very low. However, their echolocation clicks can be identified readily on specialized acoustic detectors. Acoustic detections on an array of 46 moored detectors indicate that vaquita acoustic activity declined by 80% between 2011 and 2015 in the central part of the species' range. Statistical models estimate an annual rate of decline of 34% (95% Bayesian Credible Interval -48% to -21%). Based on preliminary acoustic monitoring results from 2011-2014 the Government of Mexico enacted and is enforcing an emergency 2-year ban of gillnets throughout the species' range to prevent extinction, at a cost of $74 million USD to compensate fishers. Developing precise acoustic monitoring methods proved critical to exposing the severity of vaquitas' decline and emphasizes the need for continual monitoring to effectively manage critically endangered species.

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Introduction

The global bycatch of marine mammals in fishing nets is estimated to be over 600,000 animals per year and is a significant conservation threat for many species (Read et al. 2006; Reeves et al. 2013). Fishery bycatch contributed substantially to the extinction of the Yangtze River Dolphin in China (Turvey et al. 2007) and is an extinction risk for several species and subspecies of dolphins and porpoises (Reeves et al. 2013). This unsustainable bycatch occurs often in developing countries, even with sophisticated fisheries management programs, given that small scale fisheries receive little attention from managers (Reeves et al. 2013). Solutions that could prevent extinctions are not being implemented in many countries due to a lack of political will, a lack of convincing evidence of unsustainable bycatch rates, and sometimes a lack of viable alternative fishing gear (Reeves et al. 2013).

The vaquita (*Phocoena sinus*) is a small porpoise species endemic to the northern Gulf of California, Mexico (Rojas-Bracho & Reeves 2013). It has a very small geographic distribution entirely within an area with high fish and shrimp productivity and with easy access for small-scale fisheries. Vaquitas have been subject to a long history of unsustainable bycatch (Rojas-Bracho & Reeves 2013). The species was listed as Endangered under the U.S. Endangered Species Act in 1985 and Mexico's equivalent law in 1994, and as Critically Endangered by the International Union for Conservation of Nature and Natural Resources (IUCN) in 1996. Vaquita bycatch occurs in entangling nets used by small fishing vessels for shrimp and a wide variety of finfish (D'Agrosa et al. 2000). Fishing income is critical to the economy for the two villages that are adjacent to the range of the species (Blanco-Orozco 1998; Fig. 1). A long-standing illegal fishery for an endangered fish species, the totoaba (*Totoaba macdonaldi*; Fig. 2), has reportedly boomed in this area within the past 3 years, fueled by a

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new and lucrative illegal trade with China for totoaba swim bladders (Valenzuela-Quiñonez et al. 2015; Anonymous 2016). Fishers have reported receiving up to US$5,000 on the black market in recent years for the swim bladder of a large totoaba (Valenzuela-Quiñonez et al. 2015). Of the documented 128 vaquitas caught in gillnets between 1985 and 1992, 65% were caught in totoaba nets, which are characterized by large mesh size between 20.0 – 30.5 cm (Vidal 1995). The reported boom in illegal totoaba fishing clearly increased risks to vaquitas but the secretive nature of the fishery made direct estimation of the impact impossible.

Initially, the decline in vaquita abundance was inferred from the high estimated bycatch (D'Agrosa et al. 2000) relative to the low estimated population size (Jaramillo-Legorreta et al. 1999). Later, it was measured directly from visual surveys in 1997 (567 individuals, 95% CI 177-1073) and 2008 (245 individuals, 95% CI 68-884), resulting in a total decrease of 57%, corresponding to an average annual rate of change of -7.6% (Gerrodette et al. 2011). Vaquitas echolocate nearly continuously to locate prey, which makes acoustic signals a viable metric to detect trends in abundance. Anchored boat-based acoustic detectors deployed at random sites in the period 1997 – 2007, throughout the range of vaquitas, revealed a -7.6%/year rate of change in acoustic encounters, for a total decline of 58% (Jaramillo-Legorreta 2008). The congruence of the visual and acoustic inference, along the same period, supports an assumption of our current analysis that acoustic detection rates are proportional to population abundance.

In response to the estimated decline (Jaramillo-Legorreta 2008), the Government of Mexico implemented in 2008 a Recovery Plan (described in Rojas-Bracho & Reeves 2013), including a fishing ban inside the Vaquita Refuge (Fig. 1) and a reduction of fishing with gillnets based on buy-out and rent-out programs. The Refuge accounted for 50% of the total population estimate

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(Gerrodette & Rojas-Bracho 2011). Although enforcement appeared effective within the Refuge during the 2008 survey, the pilot study for the acoustic monitoring program reported here lost 9 of 15 acoustic detectors within the Refuge between October 30, 2010 and March 3, 2011, during the shrimp season. Few detectors were lost when monitoring occurred during summer months when there was little fishing. Therefore, it is likely that illegal fishing within the Refuge was common prior to the recent 2-year ban. Based on the preliminary results of this monitoring study, the Government of Mexico implemented an emergency ban on gillnet fishing throughout the known range of the vaquita in May, 2015 (Fig. 1). Enforcement of the temporary two-year ban is being coordinated by the Mexican Navy.

Infrequent and expensive visual sighting surveys (in 1997 and 2008) proved to be an ineffective monitoring tool for the increasingly rare vaquita. In contrast, an early acoustic monitoring program (in 1997 - 2007) was cost-effective and capable of estimating trends in acoustic detections with reasonable precision (Rojas-Bracho et al. 2010). Therefore, we designed a greatly expanded, five-year passive acoustic monitoring program, implemented throughout the Vaquita Refuge starting in 2011. Initial examination of the acoustic monitoring data from 2011 to 2013 indicated a severe decline, pointing the need for strong management actions; however, the loss of some detectors meant that survey effort was not uniform over space and time within or between seasons, meaning that simplistic interpretation of the data could be biased. An expert statistical panel was formed in 2014 to provide analytical expertise and generate robust inferences from the data collected between 2011 and 2015. Here we provide evidence that the already depleted population of vaquita declined by an additional 80% over this short time period.

Methods

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Acoustic Data Collection and Processing

The goal of the acoustic monitoring program was to detect a potential 4%/year growth rate over 5 years (Rojas-Bracho et al. 2010), where 4% was the expected intrinsic growth rate for vaquitas based on estimates for closely related species (Woodley & Read 1991; Barlow & Boveng 1991; Reilly & Barlow 1991; Slooten & Lad 1991). Variance in detection rates from earlier acoustic monitoring in 2008 indicated that approximately 5,000 sampling days per year would yield the precision needed to meet the monitoring goal (Rojas-Bracho et al. 2010). To achieve this, we designed a systematic array of 48 detectors in the Refuge, along with 14 buoys around the perimeter, recording for 3 months per season. Detectors in all perimeter buoys and two deployed in the 48 sites inside Refuge were lost each year, and their deployment ended in 2013, resulting in data from only 46 sites (Fig. 3). Results from 2008 indicated that, after testing several autonomous acoustic detectors, C-PODs (Tregenza et al. 2016; www.chelonia.co.uk) had the best detection distance (approximately 200 m) and the necessary battery life to obtain the quantity of data needed to meet the objectives (Rojas-Brach et al. 2010). C-PODs store temporal sequences of marine ultrasound (clicks) between 20 and 160 kHz, which are subsequently processed by pattern recognition software (Castellote et al. 2012; Roberts & Read 2014; Robbins et al. 2015). This classifier (KERNO version 2.044) looks for coherent sequences of more or less regularly spaced clicks (termed click trains). In the monitoring program reported here all click trains, identified as likely belonging to vaquita, were visually inspected on screen and validated by experienced analysts.

Although the sampling grid was designed to give equal coverage in space and time, in practice sampling was uneven due to shifts in the annual deployment of detectors and loss of data toward the

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end of deployment. To increase the robustness of the trend analysis (see below), we truncated the dataset to a core sampling period within which at least 50% of the detectors were operating across all 5 years (see Supporting Information for details). Vaquita acoustic behavior may vary according to many factors, thereby confounding comparisons of detection rates between years. We considered three factors related to sampling effort: season, time of day and state of the tide (the northern Gulf has a tidal range of over 10m). Using the same core period for all years helped reduce any seasonal effects, while using total acoustic counts per day averaged out diurnal effects. Tidal regimes among years were compared by the number of hours sampled at each tide speed, in intervals of 0.1m/hour using a Kruskal-Wallis ANOVA test.

Acoustic Metric

Click trains have been quantified in various ways, including attempts to identify encounters of animals by breaks in click sequences and counts of clicks over defined time periods such as Detection Positive Minutes (DPMs) (Carlström 2005; Carstensen et al. 2006; Todd et al. 2009; Scheidat et al. 2011; Roberts & Read 2014; Robbins et al. 2015). We elected to base our analysis on daily click rates (total number of identified clicks on a whole calendar day) for two reasons. First, this is the most direct form of detection data, not requiring any aggregation over time intervals such as minutes. Second, click rates are more likely to be proportional to animal density than metrics such as DPMs, which saturate at high density. For this reason, click rates are frequently used as input to passive acoustic density estimates (Marques et al. 2013; see also Discussion).

Relating trends in click rates to trends in abundance assumes there is no systematic change in animal vocal behavior. Checking this directly, e.g., through focal follows or animal-borne tags is not feasible

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for vaquita. However, we did undertake two simple exploratory diagnostics. First, we compared the distribution of number of identified clicks per DPM between years to assess if click rates may have changed over time. Second, we grouped clicks into encounters (periods of detected clicks with gaps of no longer than 30 minutes) and compared the relationship between encounter length, and number of DPMs, for each year to see if there was any indication of changes in the patterns of clicking.

Trend Analysis

Statistical modeling was used to make inferences about trends in click rate over time, where the models accounted for non-uniform sampling over space and time. We explored several models, including generalized additive models (GAMs, e.g. Ruppert et al. 2003), where the covariates were the spatial coordinates; time series models (e.g. Brockwell & Davis 2013); spatial statistical models (geostatistical models, e.g. Cressie 1993); spatio-temporal models (Cressie & Wikle 2011), and post-stratification mixture models (Little & Rubin 2014). Initial fits for estimating annual trend among all sampling sites showed that all models gave similar results. We chose to concentrate on models that were highly responsive to annual changes, so we discarded all time-series and spatio-temporal models, which smooth over time. The spatial GAM was substantially similar to the geostatistical model. The post-stratification mixture model had connections to classical sampling methods and differed substantially in assumptions from the geostatistical model. We performed detailed analyses using the geostatistical and post-stratification mixture models (full description of both models in Supporting Information). Rather than choose between these two models, we used a simple average of their predictions. No formal weighting between models was used (e.g., Hoeting et al. 1999; Burnham & Anderson 2002) in part because the spatial model was based on a log transformation making likelihood-based weighting difficult, and in part because we wanted the results to be robust to model

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choice, and not dominated by either model. Both models draw on the strength of the systematic sampling design, which is well-known to be efficient for spatially-dispersed natural populations (Thompson 1992).

In what follows, W_{ti} denotes the mean clicks/day for i^{th} site (i=1,…, 46) in the tth year (t=2011, … ,2015). Let S_{ti} be estimates of W_{ti}; the S_{ti} are predictions when acoustic detector data are missing, or smoothed spatial estimates or stratum means depending on the method. Then, an annual index of acoustic activity based on the click data is:

$$B_t = \frac{1}{K} \sum_{i=1}^{K} S_{ti}$$

for K= 46 sampling sites. Annual changes in acoustic activity were computed as B_{t+1}/B_t and for the five year trend we used the geometric mean of the annual changes which simplifies to $\lambda = (B_{2015}/B_{2011})^{1/4}$. Both geostatistical and post-stratification mixture analyses used Bayesian models and employed Markov Chain Monte Carlo sampling (MCMC, Gelfand & Smith 1990) from posterior distributions using the WinBugs/OpenBugs software (Lunn et al. 2000). For both models, MCMC chains used a burn-in of 10,000 samples with 1,000,000 further iterations, where every 100th sample was retained for posterior distribution summaries. Model and MCMC diagnostics are provided in Supporting Information.

Spatial Model

Spatial statistical models are used increasingly for wildlife data (Ver Hoef 2008). They are often used in hierarchical models (e.g., Cressie et al. 2009) that separate the noisy observation process from a

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smoother spatial surface (Royle et al. 2007). The number of clicks encountered at a sampling site were considered to be a noisy sample of an underlying use-surface, due to stochastic movements of vaquitas. There was also unequal effort at sites, with some sites completely un-sampled in some years. The hierarchical spatial model partitioned variability into a spatially smooth surface plus independent random error, where the variance of the independent part decreased proportionally to effort (number of sampling days). The estimated surface of vaquita use, then, was the predicted spatial surface. Each year was treated independently for predictions, but autocorrelation parameters were estimated by pooling across years.

The spatial model was a lognormal mixed model (i.e., data assumed normal on log scale) with a spatially autocorrelated random effect. The data were transformed by adding 1 and taking the log of the values, i.e. $Y_{ti} = \log(W_{ti} + 1)$, because some $W_{ti} = 0$. The transformed data had reasonable variance:mean properties for a Gaussian model (details in Supporting Information). The transformed data were modeled as

$$Y_{ti}|\mu_t, Z_{ti}, \sigma_\varepsilon^2, n_{ti} \sim \text{Normal}(\mu_t + Z_{ti}, \sigma_\varepsilon^2/n_{ti}),$$

where μ_t was the expected mean number of clicks per day across sites in year t, Z_{ti} was a spatially autocorrelated random effect allowing the number of clicks per day at each site, within a year, to depart from the overall mean (with sites in closer proximity to each other expected to have more similar departures from the overall mean), and σ_ε^2 was the variance for spatially independent random error, weighted by variable sampling effort (number of site-days, n_{ti}) across sites. Spatial autocorrelation among Z_{ti} within year used an exponential model (Chiles & Delfiner 2009). All parameters had non-informative priors (Supporting Information). MCMC samples from the posterior

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distribution of $\mu_t + Z_{ti}$ formed the estimates for $S_{ti} = \exp(\Box_t + Z_{ti})$ -1 and subsequently the posterior distributions for B_t and $\Box$, our primary quantities of interest.

Post-stratification Mixture Model

Mixture models have been applied to diverse types of ecological analyses for dealing with over-dispersed data (Martin et al. 2005; Rhodes 2015). The basic concept is to treat the data as having arisen from a mixture of processes rather than a single process. Here, the values of W_{ti} within a year were assumed to arise from a mixture of negative binomial processes. Within a generalized linear mixed model framework, individual sites were assigned probabilistically to one of $V = 3$ strata based on the level of detections they received across the 5 years of sampling (i.e., each site is assumed to be in a consistently high click-rate, medium click-rate, or low click-rate stratum; the data provided strong evidence for these consistencies). Estimates of S_{ti} for individual sites within each year were given by the estimated means for the strata to which the sites are assigned.

The parameter for basing inference was $\theta_{v[i],t}$ the mean click rate (clicks/day) in year t for each of the V strata to which site i is attributed. Let X_{ti} be the total number of clicks recorded at i in year t, i.e., W_{ti} * n_{ti}. Because X_{ti} were over-dispersed for a Poisson model, they were treated with a negative binomial distribution with expectation given by the product of the estimated $\theta_{v[i],t}$ and effort n_{it}, i.e.,

$$X_{it} \sim \text{negative binomial } (p_{it}, r_{v[i],t}),$$

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where p_{it} and $r_{v[i],t}$ are negative binomial parameters, and where $\theta_{v[i],t} n_{it} = r_{v[i],t} (1-p_{it})/p_{it}$ is the expectation for X_{it}. Thus, variable sampling effort across sampling sites is handled through its effect on the expectation and variance for X_{it}.

Individual sampling sites were probabilistically assigned to the 3 click-rate strata v based on the data recorded at i across the years during which site i had data, which were given a multinomial distribution,

$$v[i] \sim \text{multinomial}(\mathbf{s}_{vi}),$$

where $\mathbf{s}_{vi}$ is the vector of estimated probabilities for i being in stratum v. The vector $\mathbf{s}_{vi}$ was assumed to have a Dirichlet prior distribution (Supporting Information). The degree of certainty in assigning a site to a particular stratum depended on how correlated detections were at that site through time; sites with consistently low or high levels of detections (relative to others within a year) were assigned to a stratum with greater confidence, and all else being equal, sites with 5 years of data were assigned more confidently to a stratum than sites with fewer years of data. $S_{it} = \theta_{v[i]}$ for purposes of calculating B_t. Thus, uncertainty in group assignment was propagated through to estimates of B_t and $\square$. Information across years was shared for the purpose of assigning each sampling site to a particular group v, but the means and variances for each $\theta_{v[i],t}$ were independent. Predicted estimates for sites in years with missing data were based on the probability of belonging to group v, and the conditional mean and variance for $\theta_{v[i],t}$. Further prior distributions for the model are given in Supporting Information.

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Results

Acoustic Data and Metric

After truncating days in which less than 50% of the detectors were operating, the remaining core sampling period was 62 days, from June 19 through August 19. The number of sample days per year per site is shown in Fig. 3. The distribution of tide speeds in the core sampling period did not differ significantly between years (Kruskal-Wallis $H_{d.f 4, n=7440}$=1.425, p=0.84). We were therefore confident in proceeding without accounting for any effect of tide on detections.

We found no evidence that acoustic behavior has changed over time. The distribution of clicks per Detection Positive Minute (DPM) was very similar between years, while the relationship between encounter duration and number of DPMs was approximately linear with a very similar slope in each year. Detailed results are presented in Supporting Information, and the entire acoustic dataset is also available there.

Trend Analysis

Model-averaged estimates of annual rates of change in acoustic activity were: -17%, -3%, -46%, and -35% between successive years from 2011 to 2015. The geometric mean for the annual rate of decline from 2011 to 2015 was -34% per year with a 95% posterior credibility interval (CI) from -48% to -21% per year (Fig. 4). If acoustic activity is proportional to abundance, it is certain that the

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population declined (100% of the posterior distribution for the 2011-2015 average rate of change was < 0), and there is a 98% probability that it declined faster than 20% per year. The estimated total decline for the 2011-2015 study period was 80% (95% CI: 62% - 93%). Site-specific estimates of mean click rate (clicks per site per day) from the spatial model are illustrated in Fig. 3.

Discussion

A robust acoustic monitoring program has revealed a catastrophic decline of the vaquita population over a four-year period, following decades of already steep decline (Fig 7A in Gerrodette & Rojas-Bracho 2011). The accelerated collapse – with vaquita numbers decreasing now by 34% per year compared to 7.6% per year between 1997 and 2008 – illuminates the drastic impact of the illegal totoaba fishery (Valenzuela- Quiñonez et al. 2014, Anonymous 2016). Our preliminary analyses of these data in 2014 (CIRVA 2015) motivated the Mexican government to enact and enforce a 2-year gillnet ban and to compensate fishers and processors for US $74 million in lost income. As noted repeatedly by the international recovery team (CIRVA 2015), alternative fishing gears need to replace gillnets to allow both fishing and vaquitas to coexist.

Our interpretation that the observed decline in acoustic activity represents a population decline is broadly plausible, given that trends from independent visual and acoustic surveys have been nearly identical in the past (see Introduction). This interpretation depends on 2 key assumptions: (1) acoustic activity is proportional to abundance within the monitored area (i.e., the Vaquita Refuge); and (2) that the proportion of the population within the monitored area during the sampling period shows no systematic trend. Acoustic monitoring was only possible in areas and at times with limited gillnet and trawl fishing, which in practice meant inside the Vaquita Refuge within summer months.

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Experiments with replica acoustic detectors outside the Refuge resulted in loss of nearly all equipment. Regarding the first assumption, the acoustic measure we used, click rate, is proportional to abundance if three factors remain constant: the average animal click production rate, the effective click detection range and the false positive rate (Marques et al. 2013). Click production rate is an aspect of animal behavior that we could not measure; however, it is essential for foraging and navigation and we have no reason to suppose it varied systematically between years. We found no pattern in the acoustic metrics we examined, such as the number of clicks per detection-positive minute (although we recognize that this gives only a partial view). Factors that may affect acoustic behavior such as moon phase were balanced between years, and the same 62 calendar days were used in each year. Effective detection ranges were similarly not measured, but again there is no reason to expect systematic changes in click source level, propagation or background noise between years. For false positive rate, manual screening of detections ensured that this was negligible in all years. Regarding the second assumption (constant proportion of the population monitored), the spatial pattern of acoustic detections remained relatively constant within the Refuge across years (Fig. 3), giving some indication that spatial distribution did not change. Moreover, the spatial distribution of vaquitas was remarkably constant both within and outside the Refuge between the 1997 and 2008 visual surveys, with vaquita density outside the Refuge being low (Rojas-Bracho et al. 2010). Further evidence comes from a 2015 combined visual (waters between 20-50 meters deep) and acoustic survey of vaquitas throughout their plausible range (Supporting Information). Nearly all sightings and all but a few acoustic detections during that survey were within the Vaquita Refuge (see map in Supporting Information). No sightings were made to the south or east of the Refuge indicating the distribution of vaquitas has not shifted to that area to account for the large decline observed within the Refuge/monitoring area. However, visual data from 1997, 2008 and 2015 show a slight contraction of distribution from the southern portion of the range. Hence, the rate of decline could be slightly underestimated.

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Our work is an example of the importance of conducting well-designed, cost-effective, and precise monitoring to provide undeniable evidence of population decline and thereby prompt government action. Our approach may be useful as a template for monitoring other critically endangered species, which are often difficult to precisely monitor through visual or other conventional means. The use of an external statistical panel comprised of international leaders in their field proved valuable for expediting data analysis and minimizing politically-motivated debate concerning the credibility of results.

Given that we estimate the population to have declined by 80% in four years and that vaquitas numbered around only 200 at the start of this period, the emergency gillnet ban was clearly needed to save vaquitas. Should gillnetting resume at recent levels that resulted in an annual loss of 34%, vaquitas could quickly go extinct, and Mexico will lose its largest endemic mammal. If the vaquitas are lost, it will not be the last cetacean species to go extinct in the near future. The most endangered populations of porpoises and dolphins in the world suffer from similar gillnet threats, plus additional threats of habitat loss and degradation (Reeves et al. 2013). Mexico has taken a progressive step by temporarily banning gillnets and moving toward replacing shrimp gillnets with alternative "vaquita-friendly" gear (Anonymous 2013). But long-term actions are urgently needed if the vaquita is to be saved. Mexico can shine as a world leader by solving the cetacean-gillnet crisis in its waters or follow China in being the next to lose an endemic cetacean species.

Acknowledgments

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Figure 1. Distribution of vaquitas (hatched area) in the Upper Gulf of California. The gillnet exclusion zone was given straight boundaries described by single latitude and longitude to facilitate enforcement. The Biosphere Reserve was created in part because of vaquitas, but has not been implemented to reduce risk for the species.
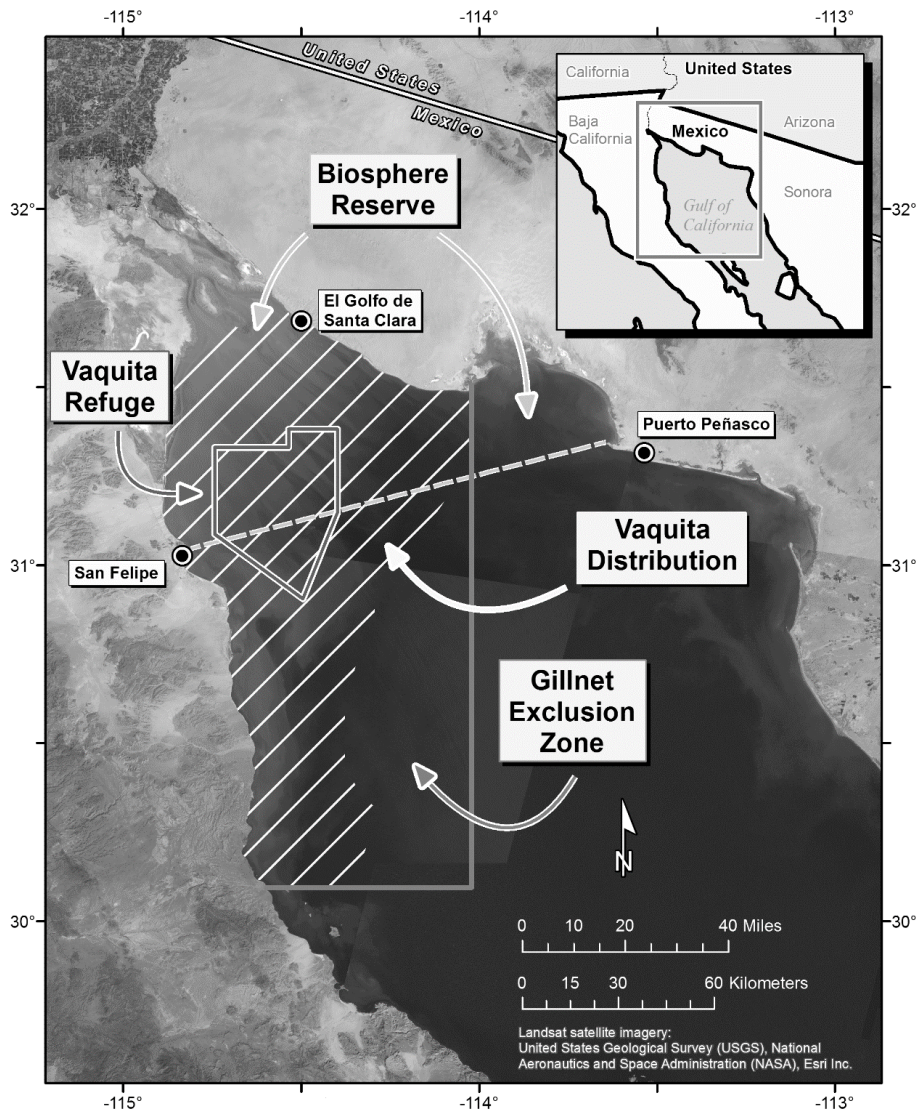

Figure 2. Totoaba (being held) and vaquita atop gillnet (photo credit: Omar Vidal).

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Fig. 2

Figure 3. Estimated mean number of clicks per day predicted by the spatial model for the 46

numbered sampling sites with data for at least one year. Values in legend are posterior medians (note

log scale). Some sites, ⊗, were missing in the indicated year. Size of circles indicate the number of

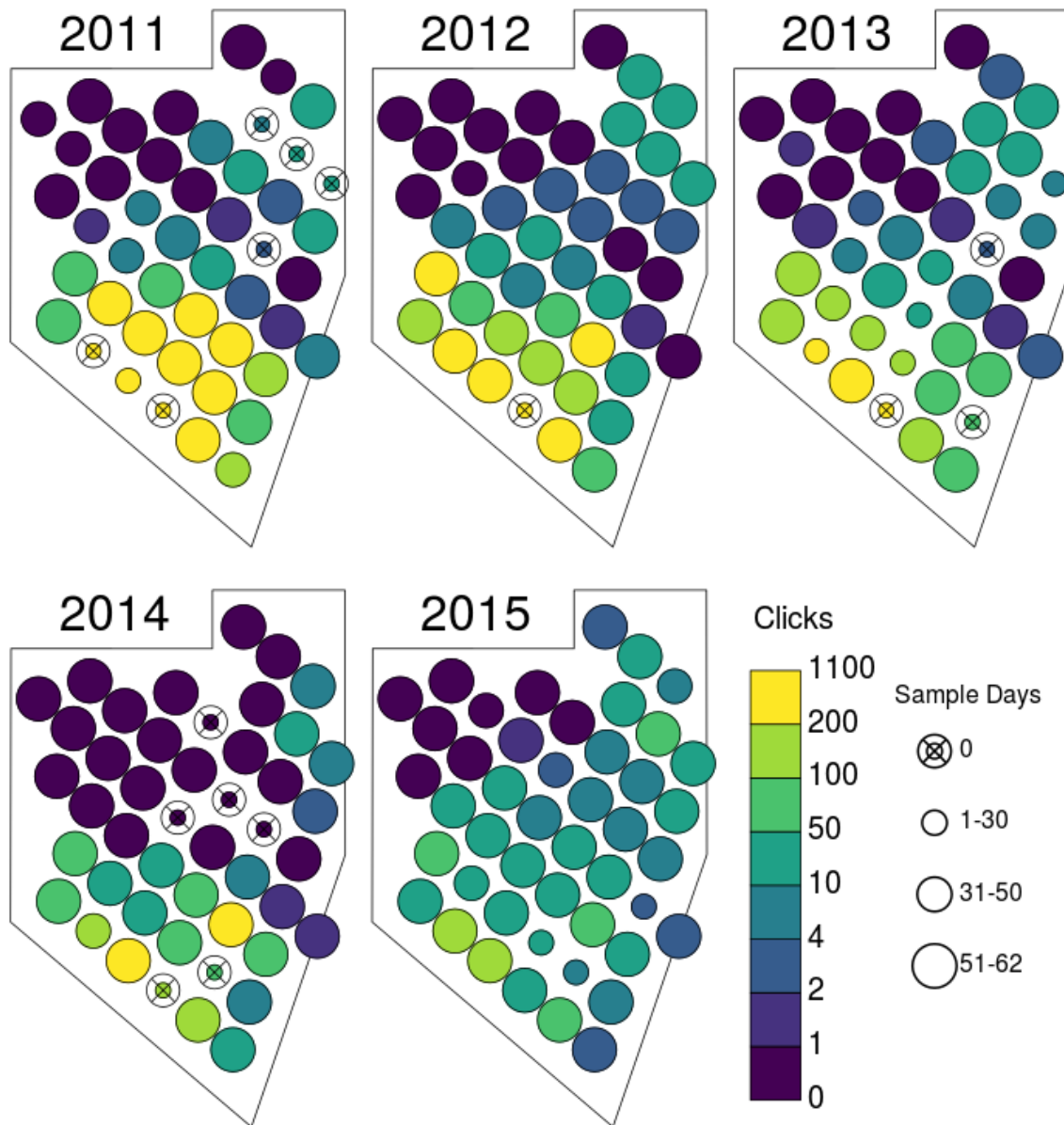sampling days on each year (see legend).

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Fig 3.

Figure 4. Posterior probability distribution for annual rate of change in mean clicks-per-day from the pooled spatial and non-spatial mixture models. The mean is a -34% change (decline) per year. 100% of the distribution is to the left of the vertical broken line, having density only in negative values.