

**An indicator-based decision framework for managing the northern California  
red abalone fishery**

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## **Executive summary**

Selection of a management strategy for the red abalone fishery is a process that requires objective and transparent evaluation of alternative approaches. Here we have built a multi-indicator decision tree that was originally designed at a public meeting in collaboration with fishery stakeholders and the Department of Fish and Wildlife. The Decision Tree has been refined over the past year and a half based on feedback from the CDFW and interested fishermen. Our model results provide an opportunity for Commissioners, members of the public and independent scientists to review the specification and performance of this approach. We recommend that all possible management strategies under consideration in the Fishery Management Plan be subjected to the same guidelines for transparency and evaluation of performance as the approach undertaken here.

The Decision Tree management strategy evaluated in this report incorporates landings data from each of 56 sites reported by fishermen as well as length frequency information collected by both CDFW and Reef Check, California at 15 sites. The decision tree can easily accommodate length frequency data from additional sites as they become available. The management strategy evaluation clearly shows that the decision tree can adjust catch to levels that reduce the possibility of stock collapse while continuing to maintain a fishery, even under extreme environmental scenarios. The decision tree can maintain long-term stock productivity levels that are slightly above the biomass associated with the production of maximum sustainable yield. The potential adoption of the harvest control rule presented here can lead to cost savings for the CDFW and meet the requirements of the MLMA. The authors welcome the opportunity to co-develop an approach with the CDFW and the Commission to overcome logistical challenges and ensure effective use of this harvest control rule.

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## 1. Introduction

Data-limited fisheries management typically proceeds in the absence of quantitative stock assessment, relying instead on simpler indicators derived from monitoring data to inform decision-making. Circumstances contributing to data limitations are varied, but can arise for example where fine-scale spatial stock structure is at odds with feasible scales of data collection or where an overwhelming number of fishers and landing sites prevents comprehensive monitoring (Butterworth et al., 2010; Dowling et al., 2015a; Prince et al., 2008). These data limitations are familiar circumstances facing management of the California recreational fishery for red abalone (*Haliotis rufescens*). Fishing occurs along the northern California coast between San Francisco and the Oregon border, is estimated to be worth US\$40 million and includes approximately 25,000 registered fishers (Reid et al., 2016). In addition, historical collapses of other abalone stocks has spurred interest in ensuring sustainability of the remaining fishery (Braje, 2016; Erlandson et al., 2005).

The red abalone fishery is regulated under the State of California Department of Fish and Wildlife (CDFW) Abalone Recovery and Management Plan (CDFW, 2005). But recently, a diver-based survey of red abalone density, which is heavily relied upon for regulatory decision-making, was the subject of scientific review. Convened by California's Ocean Science Trust (OST), this review recommended fundamental improvements to biological assessment of red abalone (OST, 2014). The CDFW has subsequently initiated development of a red abalone fishery management plan. In support of this initiative, non-governmental organizations have engaged with recreational fishers to expand data collection and to develop candidate management strategies. A protocol for cost-effective monitoring of red abalone length composition has been successfully implemented by the citizen scientist group Reef Check

California (Freiwald et al., 2016). In connection with advancements in resource monitoring, The Nature Conservancy has collaborated with independent fisheries scientists to develop and evaluate data-limited management strategies for the red abalone fishery. Management strategy development has been motivated by similar approaches utilized by Australian abalone fisheries (Prince et al., 2008).

Here, we report on formulation of management strategy options, specification of these options to overcome specific issues facing the California red abalone fishery, and evaluation of these strategies through simulation testing. Developing management strategies for the red abalone fishery is complicated by the prevalence of localized (e.g., sites < 1000 m apart) spatial differences in red abalone growth, survival, and reproductive success (Emmett and Jamieson, 1988; Geibel et al., 2010; Haaker et al., 1995; Leaf et al., 2007; McShane and Naylor, 1995; Nash, 1992; Sloan and Breen, 1988). Globally, it is well established that the small-scale meta-population dynamics of abalone species need to be accommodated in management strategy design (Bedford et al., 2013; Mayfield et al., 2012; Prince, 2005; Saunders et al., 2008; Shepherd and Brown, 1993). Problematically, within the California red abalone fishery, less than 50% of fishing sites along the coastline are monitored, aside from recording of catches, and many sites differ with respect to fishing pressure, oceanographic conditions and availability of habitat. Thus, in relying on existing data streams, management strategy design needed to accommodate site-specific signals about resource changes where this information is available, while also attempting to guide regulatory adjustments along the entire coast (Fig. 1). Because the northern California coastline consists of approximately 56 fishing sites, a pragmatic balance was also desirable between deploying fine-scale resource monitoring and implementing regulatory tactics at a regional scale. Regional regulatory tactics were a practical necessity for both fishers and fishery

enforcement. Currently, tactical regulation of red abalone catches includes a ban on scuba, a minimum shell length of 178 mm (7 inch) for possession, seasonal and area closures, and daily and annual bag limits that vary by specified management region. In addition, CDFW closed the red abalone fishery for the 2018 season, citing ecological and environmental conditions that have been detrimental to red abalone abundance, and thus, stock rebuilding considerations also entered into considerations for management strategy design.

During an initial phase of management strategy development, we recognized that design complications could not be adequately addressed through expert judgement alone. Management strategies are complicated decision-making frameworks because they emphasize the interconnections between resource monitoring, data analysis, and harvest control rules (HCRs). The functioning of each of these parts contributes to overall success in achieving fishery objectives (Butterworth et al., 2010; Harford and Babcock, 2016). Monitoring designs affect indicator reliability, and consequently, the scientific merit of management decisions (e.g., Smith et al., 2011). Data-limited analytical approaches also affect indicator reliability and can vary in complexity from summary statistics of relative abundance indices or of size composition data, to more involved statistical estimates of stock depletion, reproductive potential, or fishing mortality rates (Apostolaki and Hillary, 2009; Carruthers et al., 2015; Dick and MacCall, 2011; Gedamke and Hoenig, 2006; Geromont and Butterworth, 2015a; Hordyk et al., 2015c; Klaer et al., 2012; Martell and Froese, 2012). In amalgamating information from data analysis, HCRs must be designed to integrate this information and correctly guide regulatory changes towards beneficial outcomes (Dowling et al., 2015a).

Accordingly, management strategies were subjected to simulation testing using management strategy evaluation (MSE; Butterworth, 2007; Butterworth and Punt, 1999; Punt et al., 2016;

Sainsbury et al., 2000; Smith et al., 1999). MSE examines the collective performance of data collection, data analysis, and decision-making in the currency of whether fishery objectives are expected to be achieved over various time horizons. By comparison, treatment of any isolated aspect of a strategy, viewed independent from its intended use in decision-making, is merely an abstraction from the objective of sustainable management. The effects of uncertainty on decision-making are also explicitly addressed in MSE, for instance, by replicating the error structure or bias of a monitoring program and propagating this (potentially unreliable) information into application of a HCR. MSE is also well suited to examining management reactions to environmental changes that may affect stock status because MSE simulates recursive decision-making through time, where each decision is supplied with updated information, and thus, each decision is a reaction to a new set of conditions (A'mar et al., 2010; Punt et al., 2014).

Through MSE and through feedback from stakeholders and scientists with interest in the red abalone fishery, our initial phase of management strategy development highlighted the need for a variety of refinements (Harford et al., 2017). Initially, management strategies were formulated as multi-indicator frameworks that relied on fishery-independent density surveys, catch histories, length frequency distributions, and a climate index. These indicators collectively contributed to a HCR that was hierarchically structured as a decision tree. Like other data-limited harvest strategies that rely on small incremental regulatory changes through time, our decision tree determined the direction of total allowable catch (TAC) adjustments and iteratively modified TACs in small steps until catches stabilized around target reference points. The results of the initial MSE produced concerns that density surveys could be unreliable, resulting from patchily-distributed red abalone and modest sampling intensity (Kashiwada and Taniguchi, 2007). This concern was echoed in the OST review of the same diver-based survey of red abalone density

(OST, 2014). We ultimately decided to exclude these density surveys (given the current field sampling design) from use in further refinements of candidate management strategies. In the initial model specification, anomalies of the El Niño Southern Oscillation index were also used as an empirical indicator, reflecting shifts between desirable and undesirable environmental conditions; recognizing that red abalone growth and survival can vary dramatically in response to climate variation and its effects on kelp biomass (e.g., *Nereocystis luetkeana*), which is red abalone's main dietary constituent (Cavanaugh et al., 2011; Jiao et al., 2010; Rogers-Bennett et al., 2011; Tegner et al., 2001; Tegner and Dayton, 1987). However, this climate index was subsequently excluded because, in reality, mechanistic linkages between red abalone biology and environmental conditions are typically difficult to confirm and because environmental indices typically fail to improve management performance unless mechanistic relationships are well established (A'mar et al., 2010; Punt et al., 2014).

Based on our initial analyses, catch histories and length frequency distributions offered the most potential as useful inputs to HCRs for the red abalone fishery, given that our management strategy designs were constrained to existing data streams. Length frequency distributions were used to calculate spawning potential ratio (SPR), which is a measure of the state of reproductive potential of the stock and a proxy for spawning stock density (Goodyear, 1993; Hordyk et al., 2015c; Shepherd and Baker, 1998). Catch histories were used in a Monte Carlo method, known as catch-MSY, to estimate site-specific harvest rates relative to a harvest rate reference point (Froese et al., 2017; Martell and Froese, 2012). Initial development of management strategies also highlighted complexities in specifying reference points, which are essential inputs to HCRs and against which status indicators are compared. For indicators derived from catch histories and length frequency distributions, reference points could be established based on catch optimality

criteria or more precautionary management targets, both of which are commonly obtained from per-recruit analysis (Beverton and Holt, 1957). Conversely, the use of the diver-based density surveys would have required that a historical density reference point be chosen. In some circumstances, relative abundance-based HCRs have been demonstrably useful for fishery management (Hilborn, 2002; Little et al., 2011; Pomarede et al., 2010). Nevertheless, per-recruit based reference points can potentially reflect disparate fishery objectives than those that might have been selected based on a historical frame of reference, especially if the latter is established without connecting the historical relative stock size to stock exploitation status. While conflicting fishery objectives are the norm in fishery management, careful design multi-indicator decision frameworks is necessary to avoid selecting reference points that can work antagonistically to hinder achievement of any fishery objective.

In this study, we used MSE to evaluate refinements that were made to candidate management strategies for the red abalone fishery. Like our initial performance evaluation, these management strategies were specified as multi-indicator decision trees. In conducting MSE, we first measured the expected performance of management strategies under simulation scenarios reflecting uncertainty about the future severity of environmental conditions. We then evaluated considerations related to stock recovery in the context of the trade-off between reduction of catches and the expected pace of stock recovery. Finally, the open access nature of this recreational fishery motivated us to consider how our decision tree framework could accommodate regulatory tactics, like annual bag limits, which are more practical for recreational fisheries than TACs. Collectively, our MSE contributes guidance to specification of several central constituents of the red abalone fishery management plan.

## 2. Methods

### *Operating model*

#### *Spatial distribution of red abalone*

Stock dynamics were a spatially-explicit representation of red abalone inhabiting the northern California coastline. Abalone were distributed along a 1-dimensional array consisting of 56 sites, each of which corresponded to a recreational fishing location and spanning a total distance of approximately 540 km (334 miles) from San Francisco to the California-Oregon border (Tables 1 & 2). Given that each site corresponded to an area of one-to-tens of kilometers, we did not model site connectivity because larval dispersal and adult movement likely occur on much smaller spatial scales. Short larval durations of abalone species typically act to minimize dispersal distances from 10s to 100s of meters (Leighton, 2000; McShane et al., 1988; Prince et al., 1987; Shepherd and Brown, 1993). While potential for long distance larval dispersal has been suggested (Rogers-Bennett et al., 2016; Watson et al., 2010), most evidence demonstrates that nearly all new recruits come from parents located within several hundred meters (Gruenthal et al., 2007; Saunders et al., 2008; Temby et al., 2007). Adult movement over various time scales is also thought to be limited to 100s of meters (Ault and Demartini, 1987; Coates et al., 2013). In addition, we did not explicitly represent separation between deep water habitat that is inaccessible to free-diving fishers and shallow water areas where fishing occurs. The operating model and evaluation of management strategies, in its entirety, was implemented in the R statistical computing environment (R Development Core Team, 2012).

#### *Temporal dynamics of red abalone*

Temporal dynamics were formulated using length-structured population dynamics, which is an approach well-suited for modeling species that are difficult to age, like marine invertebrates (Breen et al., 2003; Haddon, 2011). Length-based models account for survival, growth, and reproduction through time by assigning individuals to length classes or length bins. Numbers-at-length matrices differ from numbers-at-age matrices because the latter tracks specific cohorts as they transition between age classes, while the former probabilistically tracks transitions between length classes where individuals from several cohorts are likely to be found in any given length bin (Haddon, 2011). The red abalone stock was initialized for the year 2002 and historical temporal dynamics were modeled for the time period of 2002 to 2016, using actual site-specific catches, before generating 25 years forward forecasts during which time a management strategy was implemented and modified fishery regulations.

Numbers of red abalone were assigned to length classes from 5 mm to 320 mm, with bin sizes increasing in 5 mm increments. For a given site  $l$  and simulation replicate  $k$ , the matrix algebra involved in calculating the progression of individuals between length bins, according to an annual time step,  $j$ , was (for brevity  $k$  and  $l$  subscripts are omitted):

$$\mathbf{N}_{j+1} = \mathbf{G}_j (\mathbf{S}_j \mathbf{N}_j) + \mathbf{R}_j, \quad (1)$$

where  $\mathbf{N}$  is the abundance vector of length classes,  $\mathbf{G}$  is the square growth transition matrix with upper triangle of zeros preventing negative growth in length,  $\mathbf{S}$  is the zero square matrix with only diagonal elements having non-zero values, and  $\mathbf{R}$  is the recruitment vector. The growth matrix specified how numbers-at-length would transition probabilistically into other length classes based on a Gaussian probability density function with expected growth increments obtained from a von Bertalanffy function (i.e., expected growth increment is

$\Delta L_{i,j,k,l} = (L_{\infty,j,k,l} - Lbin_i)(1 - \exp(-K_{k,l}))$ , where  $K$  is Brody growth coefficient,  $L_{\infty}$  is average maximum size, and  $Lbin$  is the lower bound of each length bin,  $i$ , ) and standard deviation of 8.5 mm (Rogers-Bennett et al., 2007). The subscript  $j$  indicates parameters that were time-varying. In the subsequent section (**Methods: *Operating Model: Environmentally-driven life history variation***), we describe our approach for generating environmentally-driven spatial and temporal variation in life history parameters.

Maturity schedules were time-invariant, but did reflect site-specific growth differences. At each site, a logistic maturity function ( $Mat_{i,k,l}$ ) was parameterized based on average maximum size ( $\bar{L}_{\infty,k,l}$ ) and the life history relationships:  $L50_{k,l} = \bar{L}_{\infty,k,l} \times 0.48$  and  $L95_{k,l} = L50_{k,l} \times 1.15$ , where  $L50$  and  $L95$  are the lengths associated with 50% and 95% probabilities of maturity, respectively. These relationships conform to established relationships that are known as Beverton-Holt life history invariants (Jensen, 1996; Prince et al., 2015), but which have been modified to reflect available life history information for California red abalone. Appendix A describes how the ratios  $L50/L_{\infty}$  and  $L95/L50$  were obtained from analysis of empirical growth and maturity patterns of red abalone from the northern California coastline, based on previous studies (Rogers-Bennett et al., 2007, 2004).

Maturity-at-length was linked to emergence-at-length. This linkage reflected an ontogenetic shift from cryptic juveniles, hidden in crevices, to mature adults that inhabit exposed substrates (Prince et al., 1988). Using actual information collected from the fishing site known as Van Damme (see Rogers-Bennett et al., 2004), this linkage was established by comparing the  $L50$  obtained from histological examination to the left-hand side of the observed length frequency distribution, which reflects emergence of red abalone. By aligning the cumulative probability of emergence-at-length with the histologically-derived  $L50$ , we were able to identify the cumulative

emergence probability associated with  $L50$ . For Van Damme, this quantity was 0.17. Logistic emergence mirrored the maturity curve, except that the emergence curve was shifted so it passed through the point  $(L50, 0.17)$ . Modeling emergence was necessary to reproduce simulated length frequency sampling (of emergent red abalone) and to incorporate any effect of emergence on fishery selectivity. Additional details are in Appendix A.

Eggs-per-female was an exponential function of length ( $fec_i = \exp(-10.434)Lmids_i^{4.701}$ ;  $Lmids$  is mid-point of each length bin), with parameter estimates obtained by fitting the exponential function to digitized length-fecundity data from Rogers-Bennett et al. (2004). Numbers of recruits at each site were calculated according to the Beverton-Holt stock-recruitment function that was re-parameterized using steepness ( $h$ ):

$$R_{j,k,l} = \left( \frac{0.8R_{0,k,l}hB_{j-1,k,l}}{0.2B_{0,k,l}(1-h) + (h-0.2)B_{j-1,k,l}} \right) \exp\left(d_{j,k,l} - \sigma^2/2\right), \quad (2)$$

where  $d$  is a recruitment deviation for each combination of year, site, and simulation replicate, which is specified to have a normal distribution with mean zero and with standard deviation  $\sigma$ .  $B_0$  is unfished egg production, and  $B$  is a measure of reproductive output summed across length bins,  $i$ , in year  $j-1$ :

$$B_{j-1,k,l} = \sum_i Mat_{i,k,l} \times fec_i \times N_{i,j-1,k,l} \quad (3)$$

Steepness was specified as 0.6, as abalone species tend to display weak compensatory recruitment at low stock size and this value is similar to those assumed in abalone stock assessments (Fu, 2014; Gorfine et al., 2005; Rossetto et al., 2013; Zhang et al., 2007). The Allee effect has been suggested as being an important limitation to reproduction at low density, although exact reproductive thresholds are difficult to identify (Catton et al., 2016; Shepherd and

Brown, 1993; Tegner et al., 1989a). In our stock-recruitment simulations, we forced complete recruitment failure to occur when reproductive output fell below 1% of unfished reproductive output. Biomass fell below this threshold in 13% of forecast across year-site combinations during our simulations of severe environmental conditions and low historical red abalone depletion levels (see **Methods: Baseline performance testing**). Age 1 recruits ( $R_{i,j}$ ) populated length bins of the recruitment matrix ( $\mathbf{R}_j$ ) according to a Gaussian probability density function with expected length calculated according to site-specific average von Bertalanffy parameters ( $\bar{L}_{\infty,k,l}$  and  $\bar{K}_{k,l}$ ) and standard deviation in length-at-age one of 8.5 mm (Rogers-Bennett et al., 2007).

Survival ( $S$ ) consisted of natural mortality ( $M$ ) and fishing mortality ( $F$ ) and was calculated at the beginning of each time step:

$$S_{i,j,k,l} = \exp(-M_{i,j,k,l} - sel_{i,j} F_{j,k,l}), \quad (4)$$

where  $sel$  is selectivity and is a function of red abalone emergence and specified minimum possession size. For a given  $l$  and  $k$ ,  $S_{i,j}$  populates the diagonal of the corresponding survival matrix ( $\mathbf{S}_j$ ). Average natural mortality-at-length was obtained from Leaf et al. (2008) who describe natural mortality as being  $0.65 \text{ year}^{-1}$  for shell lengths less than 50 mm,  $0.05 \text{ year}^{-1}$  for length greater than 245 mm, and a decreasing logistic function in between. Catch in numbers ( $C^N$ ) is calculated:

$$C_{i,j,k,l}^N = \frac{sel_{i,j} F_{j,k,l}}{(M_{i,j,k,l} + sel_{i,j} F_{j,k,l})} (1 - S_{i,j,k,l}) N_{i,j,k,l}, \quad (5)$$

And catches in weight ( $C^B$ ; kg) is:

$$C_{i,j,k,l}^B = C_{i,j,k,l}^N W_i \quad (6)$$

*Environmentally-driven life history variation*

Spatial variation was simulated by generating site-specific mean asymptotic length ( $\bar{L}_{\infty,k,l}$ ) and Brody growth coefficient ( $\bar{K}_{k,l}$ ) according to a multivariate Gaussian distribution ( $MVN(\mu, \Sigma)$ ) with  $\mu = (\bar{L}_{\infty} = 254, \bar{K} = 0.108)$  and using a standard deviation of 8.5 on  $\bar{L}_{\infty}$ , 0.006 on  $\bar{K}$ , and a correlation coefficient of 0.6 to obtain the variance-covariance matrix,  $\Sigma$  (Geibel et al., 2010; Rogers-Bennett et al., 2007). Maturity- and emergence-at-length functions were parameterized based on site-specific means the growth parameters, thus enabling growth and reproductive characteristics to co-vary at each site (Prince et al., 2015).

The life history parameters  $L_{\infty}$  and natural mortality were time-varying and were correlated with an index of the El Nino Southern Oscillation (ENSO) known as the Ocean Nino Index, which measures surface temperature anomalies (NOAA, 2017). This index was not considered to be an exhaustive environmental driver of red abalone dynamics, but we did consider this index to have reasonable statistical properties of prevailing climate fluctuations. Through laboratory experiment, water temperature has been shown to negatively affect red abalone gamete production, body condition, survival rates, and somatic growth (Moore et al., 2011; Perez, 2010; Vilchis et al., 2005). In an observational study, Jiao et al. (2010) reported a negative correlation between  $L_{\infty}$  and warm-phase temperature anomalies of the El Nino Southern Oscillation index. Likewise, trends in food availability, especially related to climate- and storm-induced variability in kelp biomass (e.g., *Nereocystis luetkeana*), have been implicated in changes to red abalone survival and growth (Cavanaugh et al., 2011; Rogers-Bennett et al., 2011; Tegner et al., 2001; Tegner and Dayton, 1987). During the time period of 2002 to 2016, actual ENSO autumn season means (i.e., the September through November average) were used in constructing historical stock dynamics. To produce forecasts, we randomly selected toroidal-like segments of the autumn

season ENSO index from the time period of 1950 to 2016 in an effort to preserve temporal autocorrelation.

Given generation of an ENSO index, time series of  $L^\infty$  and natural mortality were generated using a Cholesky transformation. We opted to link  $L^\infty_{j,k,l}$  with the ENSO index using a negative correlation of 0.5 and  $L^\infty_{j,k,l}$  varied in magnitude based on a Gaussian CV of 0.05 around the corresponding parameter  $\bar{L}^\infty_{k,l}$  (Jiao et al., 2010). Correlation strength reflected observational studies that have demonstrated statistically significant correlations between climate signals and red abalone growth parameters (Jiao et al., 2010) or kelp biomass (Cavanaugh et al., 2011), albeit, reported correlation strengths varied considerably among studies. To link natural mortality with the ENSO index, we specified a positive correlation of 0.5 and modeled this linkage to be Gaussian on a log scale. The log-normal variance was specified such that at the most extreme positive ENSO anomaly,  $M$ -at-length reflected a 40% reduction in survival (i.e., survival = exp(- $M$ )) and at the most extreme negative ENSO anomaly a corresponding increasing in survival occurred. Magnitudes of changes in natural mortality reflected experimental comparisons of red abalone survival between ambient conditions and those representing a severe el Nino warm event, which produced between 20% and 60% decreases in adult survival (Moore et al., 2011; Vilchis et al., 2005). Having both  $L^\infty$  and natural mortality co-vary with ENSO anomalies produced demographic changes that were more systematic in response to environmental conditions than would occur if life history parameters varied independently from one another. In addition, we found that previous studies were generally informative about variance in temporal fluctuations of life history parameters, but were less informative about linkages to sources influencing this variation (Geibel et al., 2010; Jiao et al., 2010; Leaf et al., 2007). Thus, in constructing these relationships with the ENSO index we conserved the total variance of life

history parameters, while assigning directional influence to a climate driver that reasonably reflected prevailing environmental conditions.

Recruitment deviations were lognormal with a standard deviation of 0.2. Deviations were independent of other environmental signals. We also simulated recruitment failures (generated independently for each site and simulation run) to reflect studies that have reported apparent absences of red abalone recruitment (Karpov et al., 1998; Rogers-Bennett et al., 2016; Tegner et al., 1989b). These events were generated as a Bernoulli random variable with recruitment failure probability of 0.25, or thus, occurring on average once per every four years.

Time-varying natural mortality increases caused by non-anthropogenic sources, like harmful algal blooms or starvation, were generated as regional signals (separate signals for each of two regions) affecting either the northern fishing sites (Mendocino, Humboldt, and Del Norte counties) or the southern fishing sites (Sonoma and Marin counties). This approach accommodated observations both about the formation of localized algal blooms and about the large-scale oceanographic conditions that initiate these events (Anderson et al., 2008; Rogers-Bennett et al., 2012; Trainer et al., 2000). A log-normal distribution representing the relative strength of harmful algal blooms was constructed based on the approximated severity of the 2011 event (Rogers-Bennett et al., 2012). It is suspected that the 2011 event may have caused a 3-fold increase in natural mortality relative to the average adult natural mortality rate. Our simulated sampling distribution had a log-normal mean of one and the occurrence of a three-fold natural mortality multiplier occurred at the 97.5<sup>th</sup> percentile. Thus, our approach recognizes the three-fold increase in natural mortality to be the previously estimated maximum recorded event strength, while also recognizing the possibility that more severe events could arise from sampling the tail of the log-normal distribution. Annual episodic events were multiplied against

$M$ -at-length for all lengths greater than the site-specific  $L50$ , reflecting non-anthropogenic changes in natural mortality dominantly affecting emergent red abalone. An exception to the probabilistic generation of red tide events was during 2011, where we imposed 3-fold multiplier to sites in Sonoma county and southward (in all simulation runs) to reflect reports of this severe event (Rogers-Bennett et al., 2012).

### *Simulated monitoring of red abalone*

Simulated observation of catches occurred without error. Observation of length frequency distributions was simulated at 15 sites that are routinely monitored by either CDFW or Reef Check California (Tables 1 & 2; CDFW, 2005; Freiwald et al., 2016). Lengths were observed as a multinomial process with an effective sample size of 100 individuals, which is a reasonable sampling variance assumption for collection of length composition data (Hulson et al., 2012). Availability of length classes to the simulated survey was affected by site-specific emergence. Both Reef Check and CDFW do not annually sample all 15 sites. During the time period of 2002 to 2016 the actual schedule of sampling events was imposed on simulation runs. During the forecast time period, 9 of 13 sites monitored by Reef Check were randomly selected annually and 3 of 10 sites monitored by CDFW were likewise randomly selected to reflect current sampling intensity. Site selection is not currently coordinated between these two organizations, and was not coordinated in our simulations. In connecting simulated data to the indicator-based decision tree, we also implemented a three-year delay between data collection and its application to decision-making as a caution against institutional delays that may occur.

### *Indicator-based decision tree*

The red abalone decision tree used catches (numbers of legal sized red abalone) and length frequency distributions to inform regulatory adjustments (Table 3). Regulatory adjustments refer to either setting annual TACs or modifying annual recreational bag limits. Our decision tree linked pre-specified regulatory adjustments to each combination of status indicators (Fig. 2). Magnitudes of regulatory adjustments, to TACs for instance, ranged between -20% and 20% from year to year based on observed quantities of status indicators. Decision tree design reflected the population biology of red abalone and past management experience with other abalone species (Prince, 2005; Prince et al., 2008). In instances where one data stream became unavailable, break-out rules were specified to cope with the remaining indicator (Table 4).

Catches at each site were used in conjunction with the catch-MSY approach to calculate the ratio of last year's exploitation rate ( $U$ ) to the exploitation rate associated with production of MSY-level catches (Martell and Froese, 2012). Catch-MSY is a numerical routine that identifies plausible combinations of intrinsic rate of increase  $r$  and unfished vulnerable stock size  $B0$ , given the site-specific input of a catch history. The estimation routine proceeds by drawing samples from specified prior distributions for  $r$  and  $B0$ . Using the Schaefer surplus production model, reconstructed stock size trends are compared against plausible benchmarks for depletion in the initial year and final year of the time series. Parameter combinations of  $r$  and  $B0$  that satisfy plausibility criteria about stock depletion are retained. Plausible parameter ranges for depletion in the initial year and final year were specified following the procedure outlined in Martell and Froese (2012). Retained  $r$  and  $B0$  combinations were used to estimate the median value of the ratio  $U/U_{MSY}$ , where  $U_{MSY}=r/2$  and current  $U$  is the catch in the final year divided by  $B0$  times median depletion. Given that red abalone catches were available for 56 sites, we leveraged information across sites to develop an informative prior for  $r$ , which occurred in two steps. First,

100,000 draws of  $r$  from a diffuse prior (Uniform[0.05, 0.15]) were made and identically applied to each site. Second, the subset of those 100,000 draws that satisfied the plausibility criteria for at least 25% of sites were retained and the remaining  $r$  values were discarded. The retained  $r$  values were used as an informative prior and re-applied to each site, producing final estimates of  $U/U_{MSY}$ . This approach gleans information about  $r$  from sites where catch histories are informative about this quantity, and then leverages this information to produce derived quantities for each site. Throughout, priors for  $B0$  were diffuse, delineating using the default approach from Martell and Froese (2012), and independently specified for each site to reflect site-specific scaling. At each site, the estimated harvest rate ratio was used to indicate whether catches were considered high (i.e.,  $U/U_{MSY}$  was greater than 1.0), low (i.e.,  $U/U_{MSY}$  was less than 0.75), or stable ( $0.75 \leq U/U_{MSY} \leq 1.0$ ).

Length frequency data was used to calculate spawning potential ratio (SPR) using the LB-SPR library in R (Hordyk et al., 2015b, 2015c, 2015a, 2016). The SPR describes the reproductive potential of an exploited stock relative to its reproductive potential in an unexploited state (Goodyear, 1993; Restrepo and Powers, 1999). The theoretical basis for the LB-SPR method is that total mortality ( $Z$ ) will affect the length frequency distribution of the stock and accordingly affect SPR. Thus, in the absence of a direct measure of total mortality, sampling of length frequency distributions can be used to infer current SPR, given a few additional life history parameters (Hordyk et al., 2015b). The maximum likelihood LB-SPR estimation routine requires input parameters of  $M/K$ , asymptotic length, coefficient of variation of asymptotic length, and a logistic maturity curve (Hordyk et al., 2015c). For all sites,  $M/K$  was specified as 0.9, which conformed to  $M$  and  $K$  expectations for red abalone. We specified the default coefficient of variation of asymptotic length to be 0.1, but allowed this parameter to

increase up to 0.3 in instances where statistical convergence could not be obtained using the default value. Because emergence is thought to reflect site-specific maturation trends (e.g., Prince et al., 1988), logistic maturity parameters ( $L50$  and  $L95$ ) were obtained from the emergence trends captured in the left-hand side of the length frequency distribution (see Appendix A). By identifying cumulative emergence probability-at-length using all length bins less than or equal to the mode of the length frequency distribution, we were also able to delineate the corresponding length at 50% maturity (e.g.,  $L50$ ). Having site-specific  $L50$  was useful so that spatial variability in growth could be incorporated into the LB-SPR fitting routine. From the  $L50$  estimate, we calculated site-specific asymptotic length according to the ratio  $L50/L\infty=0.48$  and the length at 95% maturity according to the ratio  $L95/L50=1.15$  (both ratios are based on empirical analysis of red abalone data, see Appendix A).

Because length frequency data was only collected at several sites of the 56 sites in any given year, any site where length frequency sampling occurred within the previous three years was utilized in calculating the current site-specific SPR. If a site was sampled more than once during the previous three years, the most recent sampling event was used. The site-specific length-based indicator was the ratio of current SPR to the SPR expected to produce MSY in the long-term. The quantity  $SPR_{MSY}$  was uniquely obtained for each of the sampled sites, as inter-site life history differences often do not support the use of a regional assumption about SPR that will ensure optimal recruitment (Shepherd and Baker, 1998). Site-specific  $SPR_{MSY}$  was calculated using numerical routine built into the LB-SPR library in R (Hordyk et al., 2016). For sites where monitoring did not occur, the median of site-specific SPR ratios was taken separately for each of two regions (Sonoma county and southward or Mendocino county and northward). At each site, the median SPR ratio or the site-specific SPR ratio (whichever was available) was used to

determine indicator status of high ( $SPR/SPR_{MSY} > 1.1$ ), stable ( $0.9 < SPR/SPR_{MSY} \leq 1.1$ ), low ( $0.5 < SPR/SPR_{MSY} \leq 0.9$ ), or extremely low ( $SPR/SPR_{MSY} \leq 0.5$ ).

It is worth noting how our chosen status indicators function cohesively. Catch histories are available for nearly all sites (no-take MPAs are excluded for obvious reasons) and were used to estimate harvest rate ratios via the catch-MSY approach. But this indicator only reflects current harvest intensity. It does not reveal whether the cumulative effects of past harvesting and environmental conditions have rendered the stock in a state where recruitment overfishing is likely to be occurring. The SPR ratio provides an indication of recruitment overfishing. When SPR levels trigger TAC reductions because recruitment overfishing is occurring, the harvest rate ratio works as a mitigating factor that recognizes when harvesting has been sufficiently reduced to theoretically induce stock rebuilding. Thus, since rebuilding is a slow process, the harvest rate ratio prevents ad nauseam TAC reductions (while SPR is low) and instead recognizes when reductions should be sufficient for rebuilding. Additionally, our chosen SPR and fishery exploitation indicators work non-antagonistically and reflect common reference benchmarks aimed at guiding the stock to a biomass level associated with production of MSY.

### ***Spatial allocation of fishing and regional regulatory adjustments***

Given the practical challenges associated with utilizing site-specific TACs for a regional fishery with wide-spread interest from fishers, two regional TACs were implemented. Using the Sonoma-Mendocino county line, one region consisted of Mendocino and northward (i.e., Mendocino, Humboldt, and Del Norte counties) and the other consisted of Sonoma and southward (i.e., Sonoma and Marin counties). Depending on practicality, monitoring and enforcement capabilities, regions can be further broken into smaller sizes, if desired. Using the decision tree, site-specific TAC adjustments were made. These site-specific TACs were then

summed across sites within a region to produce the regional TAC. In applying this approach, TAC adjustments to sites with the largest catches will have the greatest affect, or weighting, on the regional TAC. Thus, changes to the entire coastline are most predominantly affected by fishing sites with the highest catches, which also tend to be sites that are subject to length-frequency monitoring and are believed to be experiencing the highest relative levels of exploitation.

Regional TACs were removed (harvested) without error; however, implementation error occurred at the level of site-specific removals. We utilized a spatial effort allocation model that increased or decreased regional effort as necessary to achieve removal of the regional TAC, while maintaining the relative spatial distribution of effort commensurate with the simulated 2016 effort distribution. This effort allocation model reflected the idea that each site would continue to maintain its relative popularity with fishers into the foreseeable future, despite local abundance changes. Consequently, site-level implementation error was an emergent property of the simulations, not a pre-specified level of precision. In initial model development we also considered alternative effort allocation models; however, resulting simulation results did not vary dramatically between model formulations and thus we opted to utilize only a single effort allocation framework (Harford et al., 2017).

### ***Baseline performance testing***

Management strategies were examined against a factorial combination of operating models reflecting two historical abundance scenarios and two forecasting scenarios about future environmental conditions. To construct these operating model combinations, model tuning was carried out using the actual datasets and their corresponding indicator quantities for the time

period of 2002 to 2016 (Appendix B). Uncertainty associated with re-constructing historical conditions led us to consider two historical scenarios:

Scenario 1: *Historical natural mortality baseline.*

Scenario 2: *Historical natural mortality anomaly.*

The baseline natural mortality scenario reflected visual tuning without recognition of possible natural mortality increases during the final years of the historical time period. Conversely, the second scenario considered concerns about high natural mortality during the final years of the historical time period (i.e., Rogers-Bennett et al., 2012) by including a 20% increase in natural mortality during the final 5 years of the historical time period. This 20% natural mortality increase was arbitrary, but was intended to reflect the possibility of recent anomalous losses of red abalone. In producing forecasts under different management strategies, two scenarios about environmental conditions involved simulating (1) future ENSO anomalies and (2) future ENSO anomalies plus severe episodic natural mortality fluctuations (i.e., harmful algal blooms) and episodic recruitment failures. Note that natural recruitment fluctuations (not to be conflated with recruitment failures) occurred during all historical and forecast scenarios. Collectively simulating historical scenarios, followed by forecast scenarios, reflected a variety of environmental conditions to which the red abalone are, at times, simultaneously subjected.

Management strategies were two variants of our decision tree, which differed principally in magnitude of TAC adjustment increments when the stock was thought to be at extremely low sizes (Table 3). For comparison, a perfect-information reference HCR was also implemented. The reference HCR applied a constant fishing mortality rate of  $0.048 \text{ year}^{-1}$ , which was  $F_{MSY}$  as determined from equilibrium stock characteristics. Forecasts of management strategies were conducted for 25-year durations. All time- and space- varying stochastic parameter values were

generated ahead of simulation runs and applied in parallel against each management strategy to ensure that all evaluations occurred against the same sequences of events, thus avoiding chance differences inherent in a sample of random draws from affecting performance outcomes (Punt et al., 2016). Like current regulations, general performance characteristics were evaluated under the actual minimum harvest length of seven inches (178 mm).

Four performance metrics were calculated as medians and measures of dispersion across 56 sites times 250 simulation runs. First, we measured spawning biomass by calculating the ratio of biomass in the 25<sup>th</sup> year of the forecasts to the biomass in first year of the forecasts. Second, we calculated this ratio using catches in weight. Third, we calculated the ratio of biomass in the 25<sup>th</sup> year to  $B_{MSY}$ . Fourth, we calculated the ratio of catches in the 25<sup>th</sup> year to  $MSY$  in weight. These performance metrics were also calculated for the 10<sup>th</sup> year of the forecasts. We also parsed performance measure calculations according to sites with and without length frequency monitoring. We then calculated probabilities related to spawning biomass status and achievement of catches as the counts of a given event occurrence for each site-simulation run combination divided by total site-simulation run combinations (e.g., 1000 events / (56 sites x 250 simulation runs) = 0.07). We calculated the probability that forecast biomass in the 25<sup>th</sup> year would be below  $1/2B_{MSY}$  and the probability that forecast catches in weight in the 25<sup>th</sup> year would be below  $1/2MSY$ . We also calculated the propensity for the stock to be below  $1/2B_{MSY}$  during 25-year forecasts as the fraction of site-simulation run combination where biomass was below  $1/2B_{MSY}$  for at least half of the 25-year forecast duration. This metric can help reveal the volatility associated with severe environmental conditions and the challenges faced in stock rebuilding under these conditions.

### ***Rebuilding considerations***

Given the fishery closure that occurred for the 2018 season, we aimed to provide general guidance on stock rebuilding using our two decision trees (Table 3). These two decision trees differed principally in the magnitude of TAC reductions imposed during extremely low stock sizes. For reference, we also report rebuilding trends associated continued fishery closure and a constant catch policy proposed by a few interested fishers involving a constant TAC of 45,000 red abalone for the entire coastline. For comparison, preliminary reports of total catches for the fishery in 2015 and 2016 were, 155,196 and 159,002, respectively. The constant TAC of 45,000 was divided between Mendocino region and Sonoma region in the ratio of 80% to 20%, reflecting the actual distribution of catches. For brevity, we initialized rebuilding following our historical time period of 2002 to 2016, thus rebuilding begins in 2017 in our simulations.

### ***Implementing annual bag limits***

In this demonstration, we used annual bag limits to reflect a type of fishery control that is used for open access fisheries. The decision tree variant named faster rebuild was modified for this task (Table 3). The decision tree was implemented in its original configuration, generating a regional TAC. Then, this regional TAC was compared against the previous year's regional TAC to identify the proportional change in target catch. Accordingly, the previous year's annual bag limit in a region was adjusted upward or downward using the corresponding proportional change in target catch, rounding down to the nearest integer. This decision tree was initialized using annual bag limits of 12 for Mendocino county and 9 for Sonoma county, similar to actual regulations in 2016 and 2017. For simplicity, simulated fishers were constrained to fish in one of the two regions, where in reality fishers can harvest red abalone from both regions if they are so inclined. As a demonstration, we specified a lower annual bag limit of three to simulate a fishery

that remained open even at low stock sizes. After specifying an indirect control of total catches like an annual bag limit, it was also necessary to calculate the total regional removals of red abalone that would correspondingly occur and apply this catch against the simulated red abalone stock. Removals were calculated as the annual bag limit times the number of fishers entering the fishery in a given year. The number of fishers was specified in relative terms and forecasts specified that fishers entering the fishery would remain constant during the simulated 25-year time horizon.

### **3. Results**

#### ***Baseline performance testing***

Under typical ENSO-driven survival and growth patterns, our decision trees produced 25 year forecasts with biomass tending to be at or above  $B_{MSY}$ , and with catches correspondingly below  $MSY$  (Fig. 3; Tables 1 & 2). Our results suggest that similar performance of each respective management strategy occurred in the face of severe events, given baseline natural mortality (less-depleted) conditions at the outset of forecasts (compare Figs. 3A & 3C). Likewise, diminished performance is shared between scenarios with anomalous natural mortality increases (more-depleted) at the outset of forecasts, regardless of the specified scenario about environmental severity (compare Figs. 3B & 3D). Thus, our decision tree approaches demonstrate robustness in performance to the environmental conditions that we simulated; however, rebuilding trajectories will differ considerably based on initial depletion levels. Tracking relative changes to stock biomass over 10 and 20 year horizons illustrated increased biomass in the shorter term, with continued biomass maintenance over the longer-term (Fig. 4).

Catches tended to be diminished in the shorter term, facilitating biomass increases, but longer catches were similar to catch levels that existed at the outset of forecasts (Fig. 5).

Comparisons between sites that were subject to length-based sampling and those that were not suggested that sampled sites had catches closer to MSY than did non-sampled sites, across all operating model scenarios (Table 5). Also, biomass levels at non-sampled sites tended to exceed those at sampled sites. The reason for this performance pattern is twofold. First, selection of sampling sites by Reef Check and CDFW tends to be non-random and favor sites that have historically maintained the highest catches (Tables 1 & 2). These sites are also most likely to require large catch reductions. The corresponding regional TAC reductions likely affected non-sampled sites in a manner that reduced catches at these sites more than was necessary, especially when non-sampled sites generally were experiencing less intense fishing pressure. When viewed from the perspective of the functioning of the entire management strategy, our results suggest that non-sampled sites are treated, perhaps inadvertently, in a precautionary manner that sacrifices optimal catches for increased biomass. Second, in conducting additional simulation testing, we found that in instances of erroneous estimates of the harvest rate ratio, errors tended to be made in the direction that led to cautionary catch reductions (Appendix D). Catch-MSY enabled site-specific indicators for all sites; however, additional simulations revealed that 41% of harvest rate ratios were correctly assigned to a catch-MSY indicator category, 39% were erroneously assigned to a higher exploitation rate category, and 21% were erroneously assigned to a lower category (Appendix D). This effect propagated into overall performance of management strategies, and can be seen in results that demonstrate infrequent achievement of MSY-level catches (Fig. 3).

### ***Rebuilding considerations***

Rebuilding to  $B_{MSY}$  from the less-depleted stock condition at the outset of forecast (i.e., baseline natural mortality scenario) had a median rebuilding time during fishery closure of 9 years for both typical ENSO conditions and severe environmental conditions, respectively. In these same scenarios, both decision tree variants had median rebuild times of 14 years and 18 years, for typical ENSO conditions and severe environmental conditions, respectively (Fig. 6). When the stock was substantially depleted at the outset of forecasts (historical natural mortality anomaly) recovery time was considerably longer than the 25-year time horizon that we simulated. Rebuilding is also reported in probabilistic terms (Tables 6 & 7), which emphasizes the volatility of red abalone stock dynamics, owing to future recruitment events and changes in environmental conditions (e.g., see Fig. 3 for variation in 25-year stock status).

Our slower rebuild and faster rebuild scenarios were designed to achieve just that, different rebuild trajectories. However, this was not the case for the operating model scenarios that we considered, as rebuilding trends were quite similar between the two decision trees (Table 8). In each of these decision trees, only two combinations of status indicators (i.e., branches of the tree) differed in their corresponding percent TAC change (Table 3). Accordingly, differences in rebuilding performance between the two decision trees could have been made more pronounced in the faster rebuild strategy either by introducing different percent changes across a broader array of status indicator combinations or by introducing larger magnitude TAC reductions. It is worth noting that rebuilding plots also illustrate the relative responsiveness of the decision trees in making different magnitude TAC reductions under different levels of stock depletion. Magnitudes of TAC reductions differ in response to initial state of stock depletion (compare catch plots in Figs. 6A and 6B). Further, TACs tended to rebound faster when stock biomass

quickly rebuilt, while TACs rebounded more slowly when stock biomass also rebuilt slowly (compare catch plots in Figs. 6C and 6D).

### ***Implementing annual bag limits***

Our demonstration of annual bag limits highlights one approach to utilizing the decision trees in conjunction with regulatory tactics that are associated with recreational fisheries. Not surprisingly, MSY-based performance measures were consistent between management strategies implemented using annual bag limits or TACs (Fig. 7A). In plots that provide an example simulation run (1 simulation out of 250 total runs) using annual bag limits, we have demonstrated two important characteristics of the decision tree. The first characteristic is observed via following the biomass trend and bag limit trend for Mendocino region (Figs. 7A & 7B; noting that these results are plotted separately for each spatial region). The initially depleted state of the Mendocino region causes bag limit reductions. As biomass rebounds slightly after 2025, bag limits increase, but importantly bag limits stay below the initial 12 year<sup>-1</sup> as the biomass in the region has not reached  $B_{MSY}$ . For the Sonoma region, an initial bag limit reduction is followed by rapid biomass recovery that surpasses  $B_{MSY}$ , and accordingly, bag limits continue to increase beyond the initial 9 year<sup>-1</sup>, allowing fishers to take advantage of favorable stock sizes.

## **4. Discussion**

Our management strategies were aimed at addressing five policy considerations for the red abalone fishery. First, site-specific indicators were desirable to inform decision-making, while also recognizing that guidance was needed for adjusting catches (or other management measures) at any spatial scale, including the entire coastline. Second, indicators needed to reflect the most

reliable of existing data sources, rather than exploring alternate or new data streams. Third, flexibility needed to be maintained to accommodate additional monitoring sites, should monitoring programs expand (e.g., Freiwald et al., 2016). Fourth, flexibility needed to be maintained in HCR specification to accommodate tactical regulations associated with managing this open-access recreational fishery. Finally, HCRs needed to enable managers to implement regional regulations (perhaps using different spatial boundaries that we assumed herein).

In addressing these policy considerations, we used MSE and needed to make a variety of assumptions about operating model specifications. Our operating models reflected current estimates of life history parameters (Kashiwada and Taniguchi, 2007; Leaf et al., 2008; Rogers-Bennett et al., 2007, 2004) and we incorporated life history variation in space and time in a manner that was consistent with empirical and experimental evidence. Temporal variation in growth and survival were not simply stochastic independent variables, but were together systematically linked to a shared environmental signal (Cavanaugh et al., 2011; Jiao et al., 2010; Leaf et al., 2007). We also simulated each site as a spatially-explicit component of a larger red abalone stock, which is consistent with expectations related to larval dispersal, adult movement, and meta-population dynamics (Ault and Demartini, 1987; Coates et al., 2013; Gruenthal et al., 2007; Saunders et al., 2008; Shepherd and Brown, 1993; Temby et al., 2007). As a precaution against building reliance on larval exchange into management strategy performance, sites had no such exchange of red abalone.

The results of our MSE highlighted trade-offs that will require consideration by managers and stakeholders. First, our multi-indicator approach tended to maintain biomass levels that exceeded the biomass associated with maximum sustainable yield ( $B_{MSY}$ ), but at a cost to catches, which remained less than  $MSY$  over the 25-year time horizon. However, recent-past

(2002-2016) catches were also unlikely to be at or near MSY, given our historical reconstructions (Appendix B). Consequently, managers and stakeholders should consider whether recent-past catches were reasonable and whether forecasted catches, generated by the decision tree, would be considered similarly acceptable; especially, given the trade-off of increased protection to spawning biomass (compare Figs. 4 & 5). Second, the use of catch-MSY to calculate a harvest rate ratio enabled site-specific indicators for each site, but tended to err on the side of cautionary catch reductions in instances where estimated quantities were unreliable. Thus, the use of catch-MSY trades off the desirability of an indicator for each site, with lower catches than could be optimally obtained. Third, our candidate management strategies were conservative in terms of biomass protection at non-sampled sites. This is an example of the complex set of interactions that exist between components of a management strategy (i.e., sample site selection, data analysis, & HCR). The consequence of the limited spatial extent of length frequency sampling tended to be lower catches at non-sampled sites (whether these catch reductions were needed or not), but consequently this approach improved precautionary protection of spawning biomass at non-sampled sites.

We also found that the decision tree performed similarly between forecasting scenarios with and without extreme environmental fluctuations. Management strategy robustness to environmental conditions is central to developing candidate management strategies because red abalone are particularly vulnerable to environmental conditions in terms of their survival, growth, and reproductive success (Harley and Rogers-Bennett, 2004; Rogers-Bennett et al., 2012; Tegner et al., 2001). Median rebuilding timeframes, particularly if the actual red abalone stock is currently highly depleted, exceeded the 25-year time horizon of our simulated forecasts. This result is concerning, but reflects the slow-growth biology of abalone species, as similar

concerns have been raised about other highly depleted abalone species in California (Catton et al., 2016). Given our MSE, we provide two suggestions for viewing our results cohesively and in a manner that could assist in selecting a management strategy. Selection practices sometimes involve deciding whether candidate management strategies satisfy minimum performance standards across a sufficiently broad set of conditions, or the least across the most severe of plausible conditions. This decision-making procedure is known as satisficing (Miller and Shelton, 2010). A related consideration, known as the precautionary approach, would require deciding whether candidate management strategies pose sufficiently low risk of damage to the resource, including irreparable damage, across as many circumstances as possible (Darcy and Matlock, 1999; Restrepo and Powers, 1999).

In formulating candidate management strategies for the red abalone fishery, our use of site-specific length frequency data offers some practical solutions to on-going monitoring challenges. Measurement of site-specific indicators, especially given considerable variation in abundance between locations, is known to affect management success of abalone fisheries (Geibel et al., 2010; McShane and Naylor, 1995; Prince, 2005; Prince et al., 2008). Because diver-based observations of length frequency distributions can be expanded to accommodate additional sites, coverage of the coastline can be continually improved and new sites can be included in the decision tree as soon as they are added to monitoring programs. Importantly, the SPR indicator that is derived from length frequency data is compared to a biological reference point that is independent of historical conditions (i.e., an SPR reference point). Continued reliance on red abalone density surveys creates a scientific impasse to improving spatial coverage of monitoring. The existing suite of density survey sites have been criticized as not being indicative of red abalone abundance along the entire coastline, nor does averaging historical conditions across

sites constitute an acceptable coast-wide historical density reference point (OST, 2014). Additional density survey sites would appear desirable; however, at currently non-sampled sites there is no historical baseline for comparison. The status quo practice under the ARMP is to calculate a historical reference density using an average taken across three sites, which is then compared to current density estimates from up to 10 recently sampled sites (CDFW, 2005). The approach used in the ARMP appears to confound temporal changes in density with site-specific differences in density that may reflect spatial differences in fishing pressure, local habitat conditions, and local productivity. Setting aside related concerns about low precision of density surveys (OST, 2014), additional density survey sites would address spatial coverage concerns only if contemporary density estimates could be compared to a reference density at the same site. Of course, using contemporary estimates as a new baseline would result in a “shifting baseline” of reference conditions as new sites are added, which is not a desirable component of a fishery policy.

Arguably, reliance on length frequency data better addresses sampling design principles when it comes to red abalone ecology. Length frequency distributions measure relative changes in size structure, and are not dependent on the reliability of total counts. Therefore, length-frequency sampling appears to be less affected by depth-oriented movement or re-distribution of red abalone, so long as length sampling adequately covers the entire depth range of habitats and as long as post-exploitation sized individuals are not subject to size-based differences in detection probability. Density surveys appear to be more problematic in this regard, as unless specifically accounted for via statistical analysis, or some specialized sampling scheme, year-to-year movement of red abalone between deep and shallow habitats can be difficult to distinguish from changes in site relative abundance. Lastly, there remains an unresolved complication

pertaining to whether habitat conditions, including instances of low kelp density, affect the detection probability or ‘catchability’ during density surveys. Pronounced year-to-year changes in catchability will affect the reliability of density estimates. When detection probability is affected by environmental conditions, magnitude of bias in animal counts can co-vary with environmental conditions (Guillera-Arroita et al., 2010; Monk, 2014; Royle and Dorazio, 2009).

Our development of management strategies necessarily reflected the data-limited condition of the red abalone fishery. As it was applied here, MSE provided guidance on decision tree design, and in doing so, illustrated how pragmatism is required in designing an indicator-based approach. Pragmatism was reflected in our attempt to design a management strategy that reconciled capacity for achieving fishery objectives against practical impediments of data availability and data quality (Cadrin and Pastoors, 2008; Dowling et al., 2015b; Harford et al., 2016). We also demonstrated that data-limited methods should be examined cautiously and be subjected to simulation testing. Data-limited approaches often rely on simplified models of complex stock dynamics, and if implemented without adequate evaluation, these approaches can sometimes result in poor management performance (Carruthers et al., 2014; Fulton et al., 2016; Hordyk et al., 2015a). Conversely, in some circumstances, performance of data-limited methods has been shown to be on par with more complex approaches requiring the use of quantitative stock assessment (Geromont and Butterworth, 2015b). The application we have presented herein relates changes in indicator quantities to fish stock dynamics, and arguably, this type of approach has the potential to provide clarity to the process of developing fisheries policy and can lead to stakeholder buy-in (Campbell et al., 2007; Prince et al., 2008; Wilson et al., 2010). We recommend the results and conclusions drawn from this work be subjected to independent peer

review and evaluated against any alternative harvest control rule put forth by CDFW or other stakeholders for consideration in the red abalone fishery management plan.

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## 6. Figures

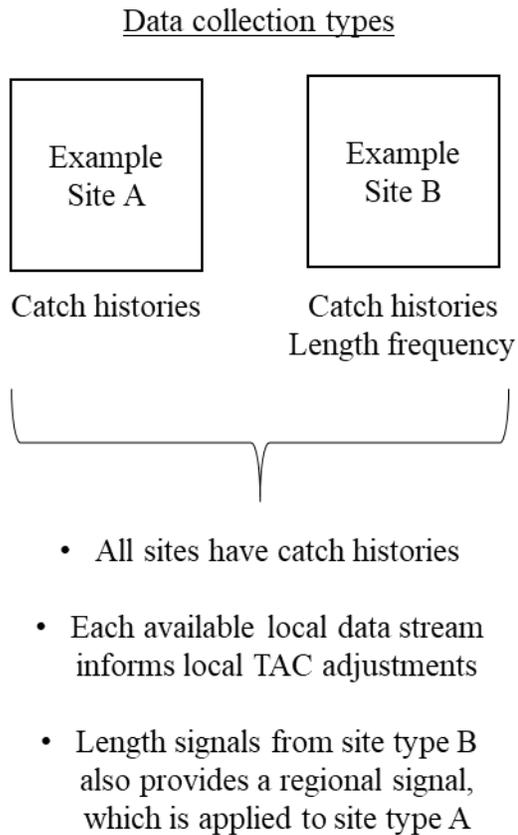


Figure 1. Data availability and its influence on harvest control rule design.

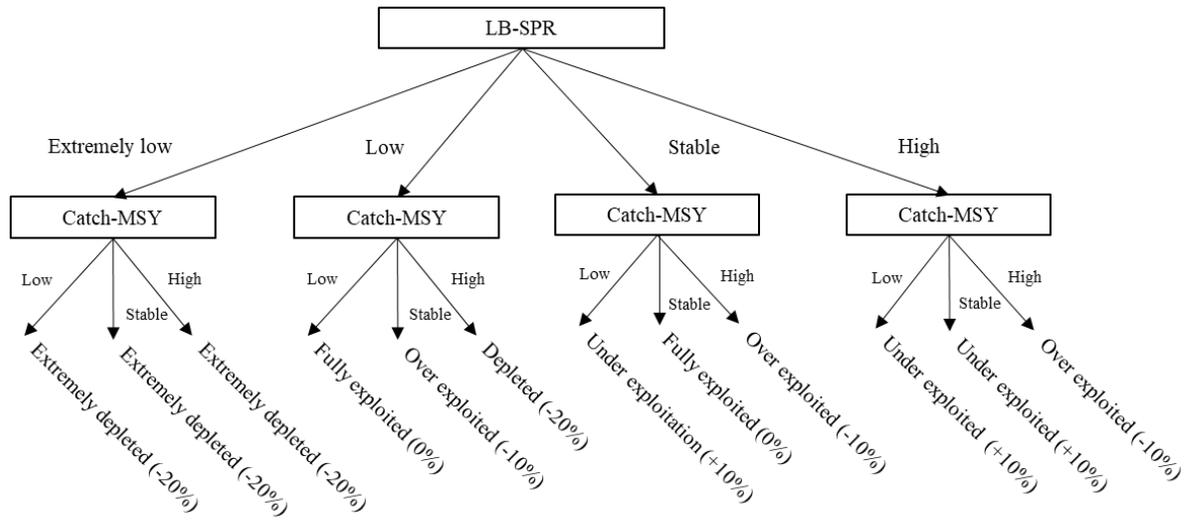


Figure 2. Faster rebuild variant of the red abalone decision tree.

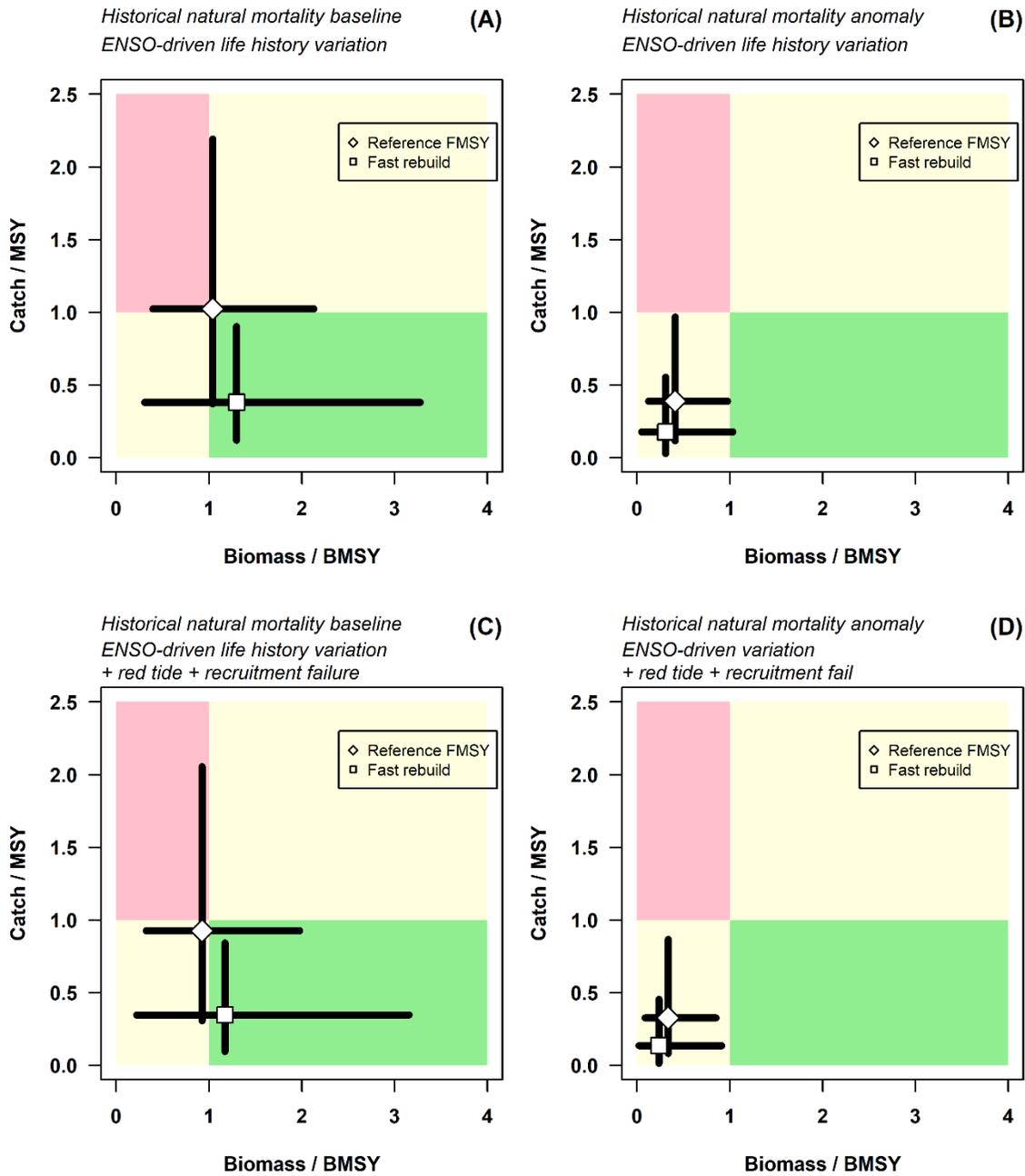


Figure 3. Trade-off plot illustrating performance at end of year 25 of forecasts. The fast rebuild scenario is contrasted against the reference  $F_{MSY}$  strategy. Slow re-build is not shown, as very similar performance of decision trees occurred over this time horizon. Plots indicate different historical conditions and different environmental conditions during forecasts. Points are medians, lines are centered 50% of simulation outcomes.

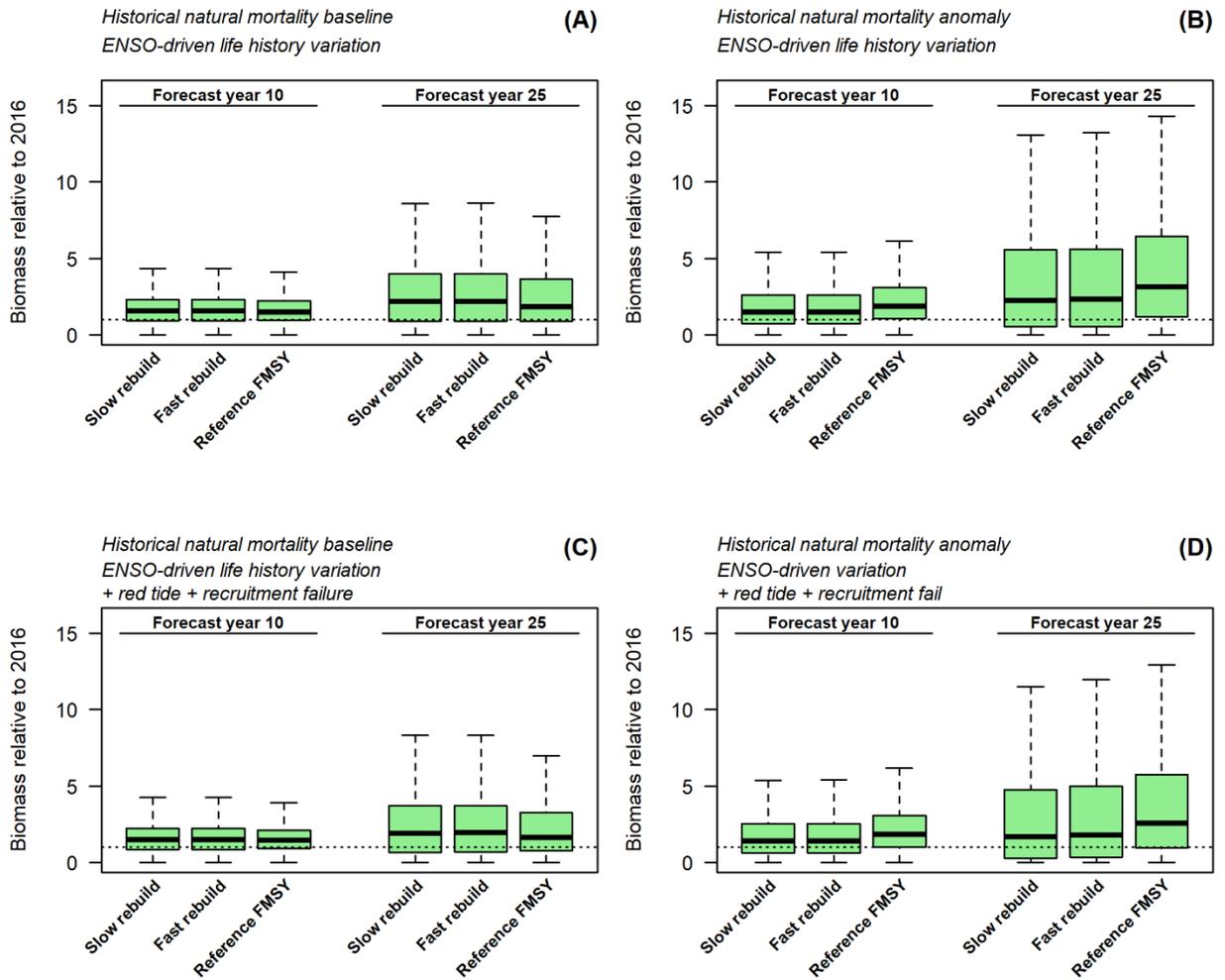


Figure 4. Box plots of relative change in spawning biomass relative to initial year of forecasts. Plots indicate different historical conditions and different environmental conditions during forecasts. Boxes show medians and interquartile ranges; whiskers are 1.5 times the interquartile range; horizontal dotted line at relative biomass of 1.0

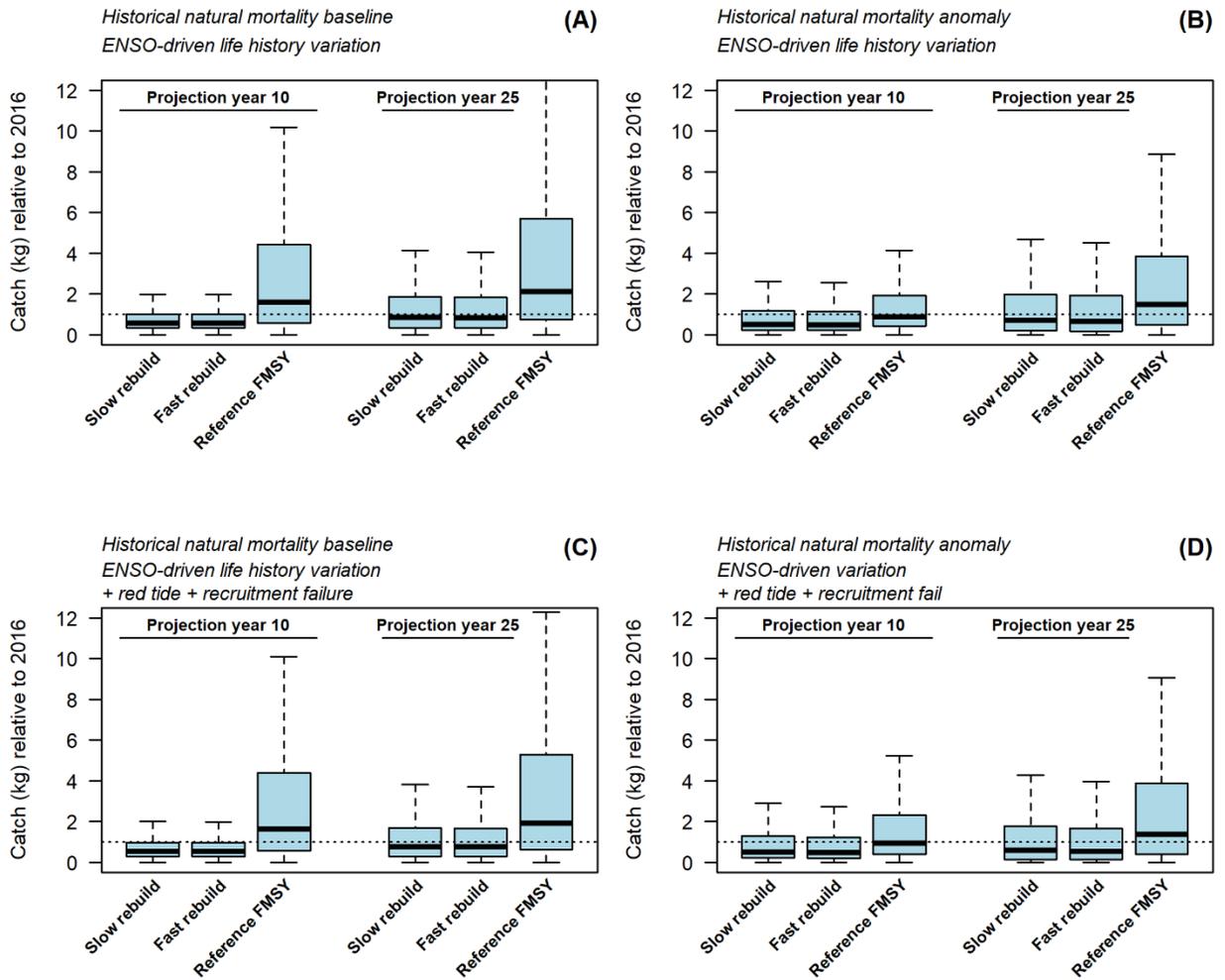


Figure 5. Box plots of relative change in catches (weight) relative to initial year of forecasts. Plots indicate different historical conditions and different environmental conditions during forecasts. Boxes show medians and interquartile ranges; whiskers are 1.5 times the interquartile range; horizontal dotted line at relative catches of 1.0

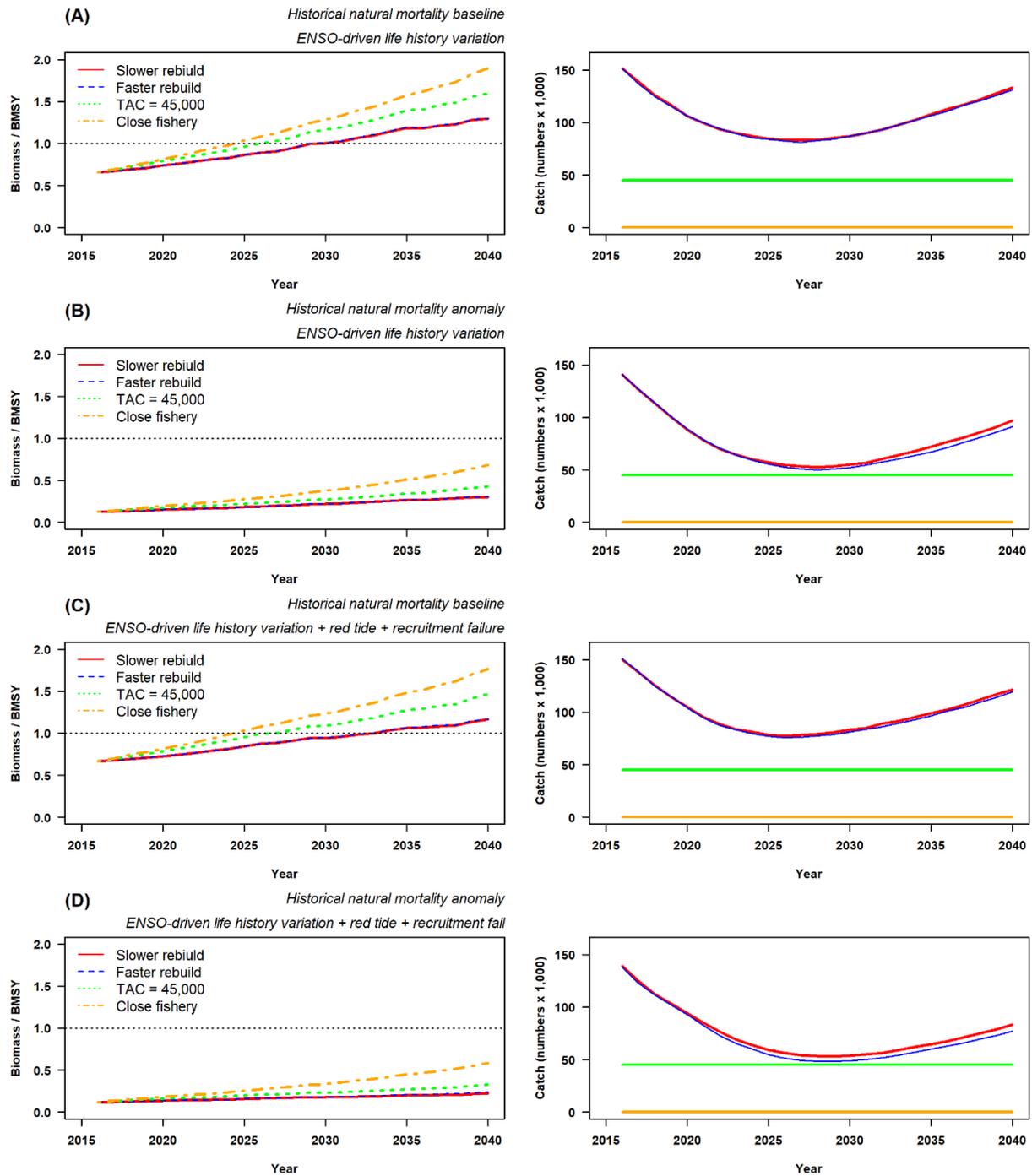


Figure 6. Median rebuilding expectations for two operating model configurations (rows), showing biomass relative to  $B_{MSY}$  and total catches in numbers.

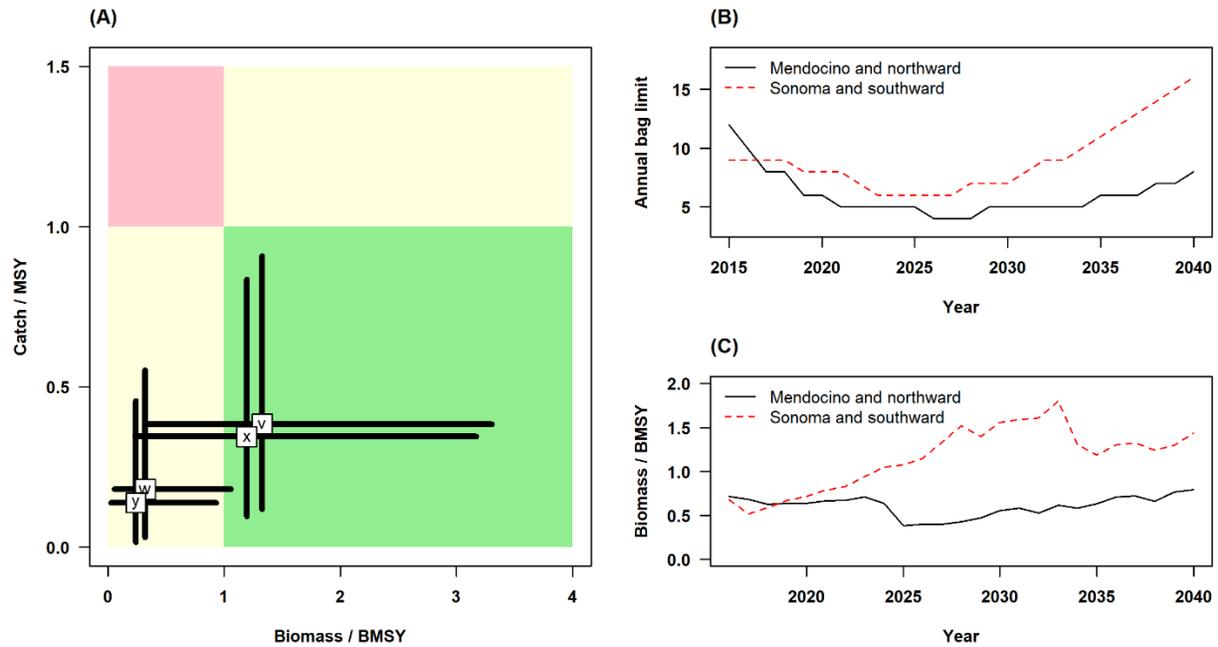


Figure 7. Demonstration of the faster rebuilding decision tree that specifies adjustments to annual bag limits. (A) 25-year performance for four operating model scenarios: v is historical natural mortality baseline, ENSO-driven life history variation; w is historical natural mortality anomaly, ENSO-driven life history variation; x is historical natural mortality baseline, ENSO-driven life history variation + red tide + recruitment failure; and, y is historical natural mortality anomaly, ENSO-driven life history variation + red tide + recruitment fail. (B) Example of 1 simulation run initialized at annual bag limits of 9 and 12 red abalone for Sonoma region and Mendocino regions, respectively, and (C) corresponding regional biomass trends for example simulation run.

## 7. Tables

Table 1. Summary of sites in Del Norte, Humboldt, and Mendocino counties. Catches are in numbers of abalone.

Site	Region	Mean Catch 2002-2016	Catch 2016	No-take Zone	Reef Check Sampling	CDFW Sampling
Crescent City	1	135	79			
Other Del Norte	1	45	6			
Patrick's Point	1	585	343			
Trinidad	1	326	198			
Punta Gorda	1	788	182			
Shelter Cove	1	3041	1557			
Other Humboldt	1	619	209			
Bear Harbor	1	386	282			
Usal	1	239	77			
Hardy Creek	1	1373	669			
Abalone Point	1	2871	1445			
Westport	1	1805	974			
Bruhel Point	1	645	188			
Kibesillah	1	572	0	✓		
MacKerricher	1	4690	3204			
Glass Beach	1	5475	5685		✓	
Georgia Pacific	1	7316	5627			
Todds Point	1	7259	6272			✓
Hare Creek	1	4605	2949			
Mitchell Creek	1	2685	2290			
Jughandle	1	5714	6464			
Caspar Cove	1	6597	6283		✓	✓
Russian Gulch	1	7097	8110		✓	✓
Jack Peters Gulch	1	3792	8404			
Mendocino Hdlns	1	10371	12222		✓	
Gordon Lane	1	3140	4424			
Van Damme	1	16525	17051		✓	✓
Dark Gulch	1	4636	5941			
Albion Cove	1	7688	6016			
Salmon Creek	1	1654	1449			
Navarro River	1	3306	2447			
Elk	1	8193	6506			
Point Arena Lighthouse	1	4387	1010		✓	
Arena Cove	1	8993	4040		✓	✓
Moat Creek	1	9592	5132			
Schooner Gulch	1	539	161			
Saunders Landing	1	701	0	✓		
Anchor Bay	1	4965	3785			
Robinson Point	1	1327	1414			

Table 2. Summary of sites in Sonoma and Marin counties. Catches are in numbers of abalone.

Site	Region	Mean Catch 2002-2016	Catch 2016	No-take Zone	Reef Check Sampling	CDFW Sampling
Gualala Point	2	850	321			
Sea Ranch	2	10803	5723		✓	✓
Black Point	2	244	26			
Stewarts Point	2	1098	153			
Rocky Point	2	232	39			
Horseshoe Cove	2	1038	0	✓		
Fisk_Mill Cove	2	5542	1415			
Salt_Point State Park	2	8555	4197		✓	✓
Ocean Cove	2	4293	2897		✓	✓
Stillwater Cove	2	3747	3147		✓	
Timber Cove	2	7625	3681			✓
Fort Ross	2	28672	2366		✓	✓
Jenner	2	2515	963			
Bodega Head	2	902	263		✓	
Tomaes Point	2	1968	561			
Point Reyes	2	281	31			
Other Marin	2	424	124			

Table 3. Rationale for the decision tree based on indicators of spawning potential ratio (SPR) and exploitation rate calculated via catch-MSY approach. Two decision trees are described that differ with respect to rebuilding red abalone abundance with it is at low levels.

<b>SPR indicator</b>	<b>Catch-MSY indicator</b>	<b>Exploitation status</b>	<b>TAC adjustment</b>	<b>Explanation</b>
<b>Slower rebuild</b>				
High	High	Over exploitation	-10%	Watch and wait
High	Stable	Under exploited	+10%	SPR high under stable catches
High	Low	Under exploited	+10%	Possibly restrictive management
Stable	High	Over exploitation	-10%	SPR stable, but fishing is increasing
Stable	Stable	Fully exploited	0%	SPR stable around reference
Stable	Low	Low exploitation	+10%	Possibly restrictive management
Low	High	Depleted	-20%	Recruitment overfishing possible
Low	Stable	Depleted	-10%	Recruitment overfishing possible
Low	Low	Depleted	0%	Recruitment overfishing possible
Extremely low	High	Very depleted	-20%	Rebuild abundance
Extremely low	Stable	Very depleted	-10%	Rebuild abundance
Extremely low	Low	Very depleted	-10%	Rebuild abundance
<b>Faster rebuild</b>				
High	High	Over exploitation	-10%	Watch and wait
High	Stable	Under exploited	+10%	SPR high under stable catches
High	Low	Under exploited	+10%	Possibly restrictive management
Stable	High	Over exploitation	-10%	SPR stable, but fishing is increasing
Stable	Stable	Fully exploited	0%	SPR stable around reference
Stable	Low	Low exploitation	+10%	Possibly restrictive management
Low	High	Depleted	-20%	Recruitment overfishing possible
Low	Stable	Depleted	-10%	Recruitment overfishing possible
Low	Low	Depleted	0%	Recruitment overfishing possible
Extremely low	High	Very depleted	-20%	Rebuild abundance
Extremely low	Stable	Very depleted	-20%	Rebuild abundance
Extremely low	Low	Very depleted	-20%	Rebuild abundance

Notes: The harvest rate ratio  $U/U_{MSY}$  indicated whether catches were considered high (i.e.,  $U/U_{MSY}$  was greater than 1.0), low (i.e.,  $U/U_{MSY}$  was less than 0.75), or stable ( $0.75 \leq U/U_{MSY} \leq 1.0$ ). The SPR ratio  $SPR/SPR_{MSY}$  indicated whether site status as high ( $SPR/SPR_{MSY} > 1.1$ ), stable ( $0.9 < SPR/SPR_{MSY} \leq 1.1$ ), low ( $0.5 < SPR/SPR_{MSY} \leq 0.9$ ), or extremely low ( $SPR/SPR_{MSY} \leq 0.5$ ).

Table 4. Break-out rules in instances where length frequency data or catch time series are not available or are not included in the analysis.

<b>Indicator</b>	<b>TAC adjustment</b>
Catch history	
High	-10%
Stable	0%
Low	+10%
Length frequency data (for either fast and slow rebuilding)	
High	+10%
Stable	0%
Low	-10%
Extremely low	-20%

Table 5. MSY-based performance measures for 25-year forecasts of two decision tree variants and a reference  $F_{MSY}$  rule. Column headings indicate operating model scenarios.  $B$  is spawning biomass and  $B_{MSY}$  is the spawning biomass that produces maximum sustainable yield, both in kg;  $C$  is catch and  $MSY$  is maximum sustainable yield, both in kg; ENSO is El nino Southern Oscillation index; episodic events include both increases in natural mortality and recruitment failures.

Management strategy	Baseline natural mortality		Anomalous natural mortality	
	ENSO-driven variation	ENSO + episodic events	ENSO-driven variation	ENSO + episodic events
<i>Performance metric: median <math>B / B_{MSY}</math> (standard error)</i>				
All sites (56)				
Slower rebuild	1.30 (2.81)	1.17 (2.81)	0.30 (1.24)	0.22 (1.28)
Faster rebuild	1.30 (2.81)	1.17 (2.81)	0.31 (1.24)	0.24 (1.29)
$F_{MSY}$	1.04 (1.54)	0.93 (1.50)	0.41 (0.87)	0.34 (0.86)
Sites with length sampling (15)				
Slower rebuild	1.20 (2.72)	1.06 (2.77)	0.27 (1.17)	0.19 (1.23)
Faster rebuild	1.20 (2.72)	1.07 (2.77)	0.27 (1.17)	0.20 (1.24)
Sites without length sampling (41)				
Slower rebuild	1.33 (2.84)	1.20 (2.82)	0.32 (1.26)	0.23 (1.30)
Faster rebuild	1.34 (2.84)	1.21 (2.82)	0.32 (1.27)	0.25 (1.31)
<i>Performance metric: median <math>C / MSY</math> (standard error)</i>				
All sites (56)				
Slower rebuild	0.39 (1.02)	0.35 (0.96)	0.18 (0.86)	0.15 (0.78)
Faster rebuild	0.38 (1.01)	0.35 (0.95)	0.18 (0.85)	0.13 (0.74)
$F_{MSY}$	1.02 (1.71)	0.92 (1.68)	0.39 (0.96)	0.33 (0.96)
Sites with length sampling (15)				
Slower rebuild	0.48 (1.09)	0.44 (1.01)	0.22 (0.89)	0.19 (0.76)
Faster rebuild	0.47 (1.10)	0.43 (1.00)	0.21 (0.88)	0.17 (0.72)
Sites without length sampling (41)				
Slower rebuild	0.36 (0.99)	0.33 (0.94)	0.17 (0.84)	0.14 (0.78)
Faster rebuild	0.35 (0.99)	0.32 (0.92)	0.16 (0.83)	0.12 (0.75)

Table 6. Probabilistic biomass performance for two decision tree variants. For reference,  $F_{MSY}$  rule, close-fishery, and constant TAC rules are reported, allowing rebuilding comparisons over 10 year and 25 year time horizons. Column headings indicate operating model scenarios;  $B$  is spawning biomass and  $B_{MSY}$  is the spawning biomass that produces maximum sustainable yield, both in kg; Pr is probability; ENSO is El nino Southern Oscillation index; episodic events include both increases in natural mortality and recruitment failures.

Management strategy	Baseline natural mortality		Anomalous natural mortality	
	ENSO-driven variation	ENSO + episodic events	ENSO-driven variation	ENSO + episodic events
<i>10-year performance metric: Pr [B &lt; 1/2B<sub>MSY</sub>]</i>				
All sites (56)				
Slower rebuild	0.36	0.38	0.76	0.77
Faster rebuild	0.36	0.38	0.76	0.77
$F_{MSY}$	0.36	0.37	0.74	0.75
Close fishery	0.30	0.31	0.69	0.70
Constant TAC = 45,000	0.33	0.34	0.73	0.90
Sites with length sampling (15)				
Slower rebuild	0.38	0.39	0.77	0.77
Faster rebuild	0.38	0.39	0.77	0.77
Sites without length sampling (41)				
Slower rebuild	0.36	0.37	0.76	0.77
Faster rebuild	0.36	0.37	0.76	0.76
<i>25-year performance metric: Pr [B &lt; 1/2B<sub>MSY</sub>]</i>				
All sites (56)				
Slower rebuild	0.32	0.35	0.60	0.64
Faster rebuild	0.32	0.35	0.60	0.63
$F_{MSY}$	0.30	0.34	0.55	0.60
Close fishery	0.19	0.22	0.42	0.46
Constant TAC = 45,000	0.26	0.29	0.53	0.58
Sites with length sampling (15)				
Slower rebuild	0.34	0.36	0.61	0.66
Faster rebuild	0.34	0.36	0.61	0.65
Sites without length sampling (41)				
Slower rebuild	0.31	0.34	0.59	0.64
Faster rebuild	0.31	0.34	0.59	0.63

Table 7. Probabilistic catch performance for two decision tree variants. For reference,  $F_{MSY}$  rule, close-fishery, and constant TAC rules are reported, allowing rebuilding comparisons over 10 year and 25 year time horizons. Column headings indicate operating model scenarios;  $C$  is catch and  $MSY$  is maximum sustainable yield, both in kg; Pr is probability; ENSO is El nino Southern Oscillation index; episodic events include both increases in natural mortality and recruitment failures.

Management strategy	Baseline natural mortality		Anomalous natural mortality	
	ENSO-driven variation	ENSO + episodic events	ENSO-driven variation	ENSO + episodic events
<i>10-year performance metric: Pr [C &lt; 1/2MSY]</i>				
All sites (56)				
Slower rebuild	0.74	0.75	0.87	0.87
Faster rebuild	0.74	0.75	0.87	0.87
$F_{MSY}$	0.36	0.37	0.74	0.73
Close fishery	1.00	1.00	1.00	1.00
Constant TAC = 45,000	0.88	0.88	0.89	0.90
Sites with length sampling (15)				
Slower rebuild	0.70	0.69	0.86	0.84
Faster rebuild	0.70	0.70	0.86	0.86
Sites without length sampling (41)				
Slower rebuild	0.75	0.77	0.87	0.87
Faster rebuild	0.75	0.77	0.88	0.88
<i>25-year performance metric: Pr [C &lt; 1/2MSY]</i>				
All sites (56)				
Slower rebuild	0.57	0.59	0.72	0.75
Faster rebuild	0.58	0.60	0.73	0.77
$F_{MSY}$	0.31	0.34	0.57	0.60
Close fishery	1.00	1.00	1.00	1.00
Constant TAC = 45,000	0.85	0.85	0.85	0.86
Sites with length sampling (15)				
Slower rebuild	0.51	0.53	0.69	0.73
Faster rebuild	0.52	0.54	0.70	0.74
Sites without length sampling (41)				
Slower rebuild	0.59	0.61	0.73	0.76
Faster rebuild	0.60	0.62	0.74	0.78

Table 8. Probabilities of stock biomass being below  $1/2B_{MSY}$  across the duration of 25 year simulations runs. This performance metric is calculated as the fraction of site-simulation run combinations where biomass was below  $1/2B_{MSY}$  for at least half of the 25-year forecast duration. ENSO is El nino Southern Oscillation index; episodic events include both increases in natural mortality and recruitment failures.

<b>Management strategy</b>	<b>Baseline natural mortality</b>		<b>Anomalous natural mortality</b>	
	<b>ENSO-driven variation</b>	<b>ENSO + episodic events</b>	<b>ENSO-driven variation</b>	<b>ENSO + episodic events</b>
All sites (56)				
Slower rebuild	0.35	0.37	0.74	0.75
Faster rebuild	0.35	0.37	0.74	0.75
Close fishery	0.27	0.29	0.64	0.65
Constant TAC = 45,000	0.31	0.33	0.70	0.71

## Appendix A. Red abalone life history parameters used in simulations

### Section headings:

*Section i: Growth, natural mortality, and fecundity*

*Section ii: Maturity-at-length and emergence-at-length*

*Section iii: Per-recruit analysis*

*Section i: Growth, natural mortality, and fecundity*

Simulating life history characteristics that were representative of red abalone was required in order to conduct management strategy evaluation. A summary of growth, natural mortality, and fecundity using average parameter values (noting that some parameter values were time- and space-varying in simulation runs) is presented (Fig. A1). Average life history parameters were obtained from several sources, namely:

- von Bertalanffy growth parameters  $L_{\infty}=254$ ,  $K=0.108$ ,  $t_0=0$  from Rogers-Bennett et al. (2007);
- Natural mortality-at-length from Leaf et. al (2007);
- Length-weight conversion ( $W=\alpha*L^{\beta}$ )  $\alpha=0.0001$  and  $\beta=3.03$ , with whole weight in grams and length in mm, from Rogers-Bennett et al. (2007);
- Fecundity-at-length was obtained by digitizing their Figure 4 from Rogers-Bennett et al. (2004) and fitting an exponential function of the form  $\text{fecundity} = a*L^b$ , where  $a=\exp(-10.434)$  and  $b=4.701$ .
- Beverton-Holt stock-recruitment function with steepness of 0.6, reflecting assumptions made in abalone stock assessments (Fu, 2014; Gorfine et al., 2005; Rossetto et al., 2013; Zhang et al., 2007);

## *Section ii: Maturity-at-length and emergence-at-length*

A logistic function was fit to maturity-at-length dataset that was produced through histologic examination (Data from their Table 1; Rogers-Bennett et al., 2004), which suggested that length associated with 50% of individuals in the population being mature was  $L50=120.8$  mm and  $L95=142.2$  (Fig. A2). The ratio of  $L50$  to  $L\infty$  is one of the Beverton-Holt life history invariants. Prince et al. (2015) report variation in this ratio as it relates to life history types. We calculated this ratio for northern California red abalone as  $L50 / L\infty = 122/254 = 0.48$ . We were also interested in whether observed emergence trends that are reflected in the left-hand side of length frequency distributions could be informative about maturation schedules, rather than needing to rely on continued collection of histological data (Fig. A3; e.g., Prince et al., 1988). This question is also relevant to the problem that growth characteristics and corresponding maturity schedules of red abalone may vary along the coastline, and thus, it is desirable to have site-specific information about maturity. Histology data were available at the site known as Van Damme, as were length frequency distributions. For Van Damme, we characterized the emergence trend captured in the left-hand side of the length frequency distribution by binning length by 5 mm bins and using all length bins less than or equal to the mode of the length frequency distribution to construct a cumulative distribution function (CDF; Fig. A4). We then identified the cumulative probability associated with the histology-based  $L50=120.8$ , which was 0.17 (Fig. A4). This result suggests that only a small fraction of individuals have emerged from crevices onto exposed substrates at lengths associated with 50% of the population being mature. By understanding that  $L50$  corresponds to the approximate “base” of the length-frequency distribution, or more precisely, to a low cumulative probability of emergence (i.e., 0.17), we utilized this information at other sites. By having this information, we were able to use length-

frequency distributions collected at other sites (where histology information has not been obtained) to provide site-specific estimates of  $L50$ . This quantity is obtained by constructing a CDF from the left-hand side of the length frequency distribution and identifying the length associated with a cumulative probability of 0.17. Having  $L50$  enables calculation of site-specific growth and reproductive characteristics via Beverton-Holt life history invariants (i.e., Jensen, 1996; Prince et al., 2015).

### *Section iii: Per-recruit analysis*

Per-recruit analyses have been widely applied to abalone species and used to derive eggs-per-recruit or biomass-per-recruit based reference points (Leaf et al., 2008; McShane and Naylor, 1995; Nash, 1992; Rogers-Bennett and Leaf, 2006; Shepherd and Baker, 1998). The per-recruit analysis of the simulated red abalone population was conducted as a means to provide context about stock dynamics. The per-recruit analysis (Tables A1, A2 and Fig. A7) was conducted using the average quantities for life history parameters, and the following:

- Maturity-at-length and emergence at length using  $\bar{L}_{\infty} = 254$ , such that  $L50 = \bar{L}_{\infty} \times 0.48 = 122$  and  $L95 = L50 \times 1.15 = 140$  (Fig. A5).
- Availability to the fishery reflected knife-edge selectivity at 178 mm (current regulatory requirement) and emergence pattern (Fig. A6).

Table A1. Reference points obtained relative to **MSY in numbers** of red abalone. SSB is spawning output in eggs, SSB0 is unfished egg production, SPR is spawning potential ratio.

Refererence point	Fishing mortality rate	SPR	SSB/SSB0	Catch in numbers / MSY
F <sub>SPR60%</sub>	0.023	0.60	0.51	0.87
F <sub>SPR50%</sub>	0.034	0.50	0.40	0.88
F <sub>SPR40%</sub>	0.051	0.40	0.28	0.98
F <sub>MSY</sub>	0.063	0.35	0.22	1.00
F <sub>SPR30%</sub>	0.083	0.30	0.16	0.96

Table A2. Reference points obtained relative to **MSY in biomass** of red abalone. SSB is spawning output in eggs, SSB0 is unfished egg production, SPR is spawning potential ratio.

Refererence point	Fishing mortality rate	SPR	SSB/SSB0	Catch in biomass / MSY
F <sub>SPR60%</sub>	0.023	0.60	0.51	0.85
F <sub>SPR50%</sub>	0.034	0.50	0.40	0.96
F <sub>MSY</sub>	0.048	0.41	0.30	1.00
F <sub>SPR40%</sub>	0.051	0.40	0.28	0.99
F <sub>SPR30%</sub>	0.083	0.30	0.16	0.88

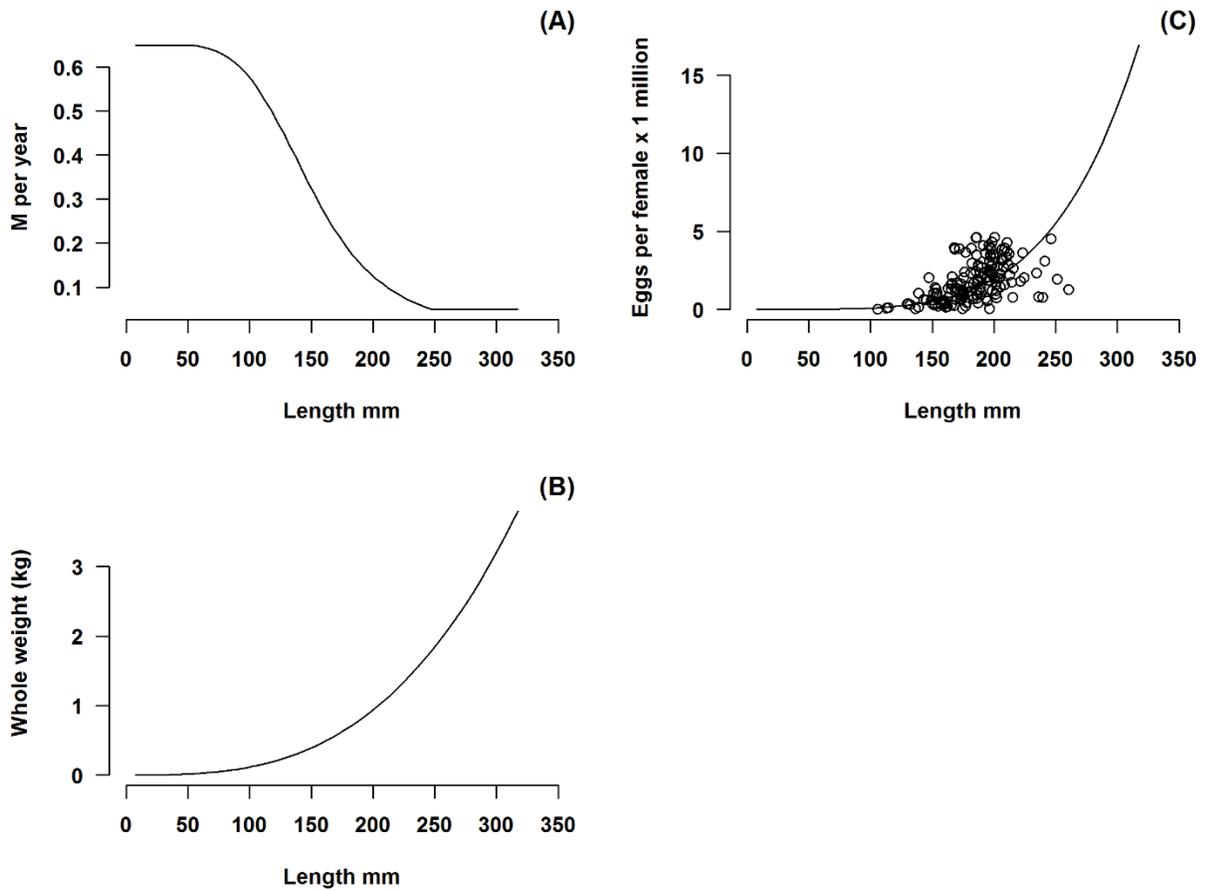


Fig. A1. Simulated relationships between length and (A) natural mortality, (B) weight, and (C) eggs-per-female with actual data points used in fitting fecundity function (see their Figure 2 in Rogers-Bennett et al. (2004)).

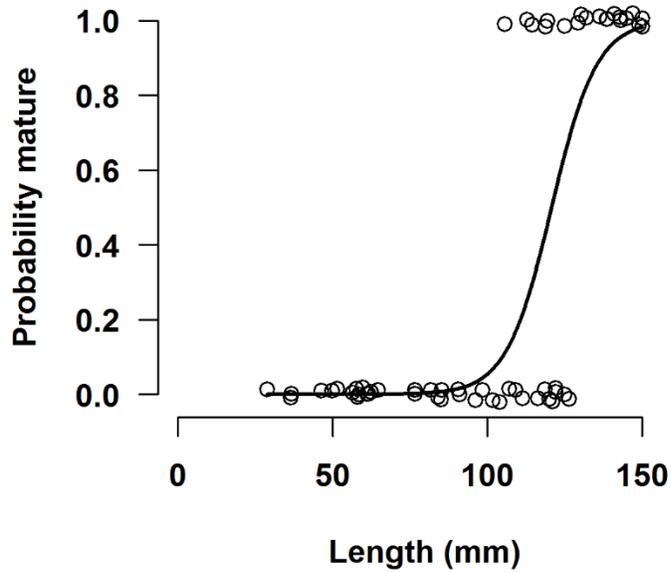


Fig. A2. Simulated maturity function with actual data points used in fitting a logistic curve to histologic examinations from Rogers-Bennett et al. (2004).

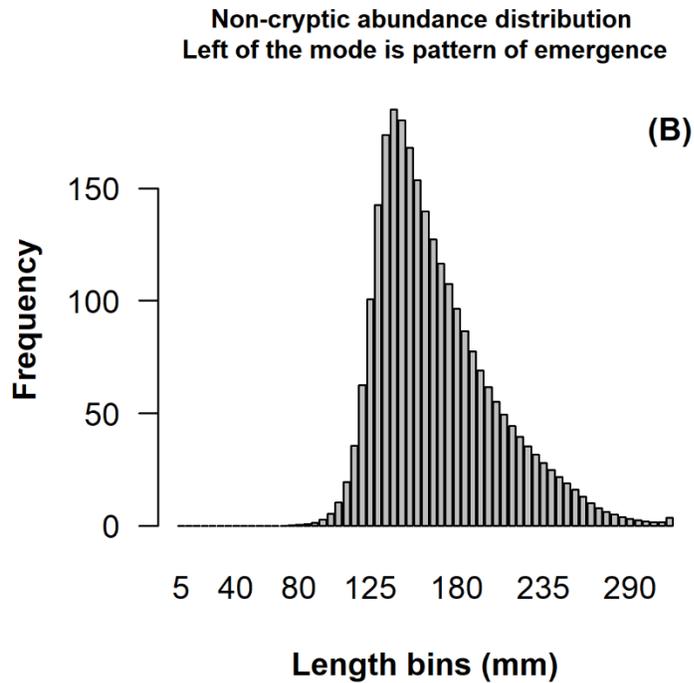
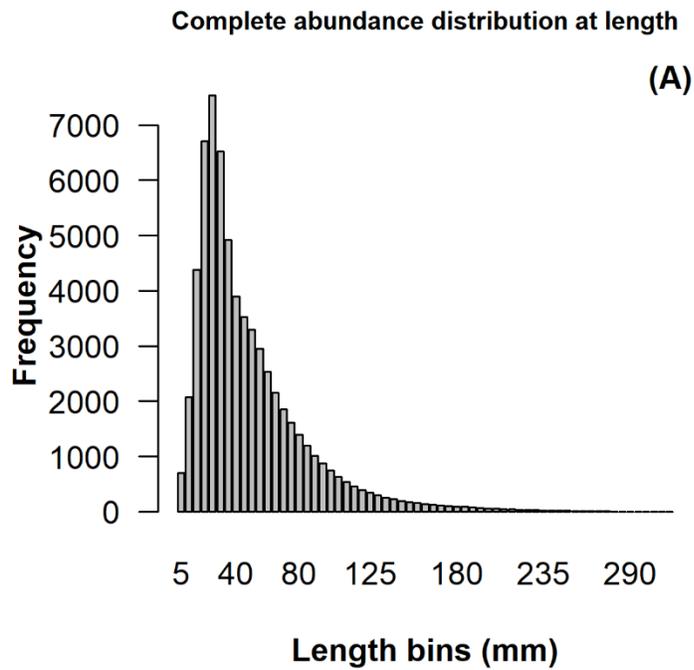


Figure A3. Simulated example of patterns of red abalone (A) abundance-at-length and (B) emergence-at-length pattern in the ascending limb of the length frequency distribution (i.e. the left-hand side of the distribution).

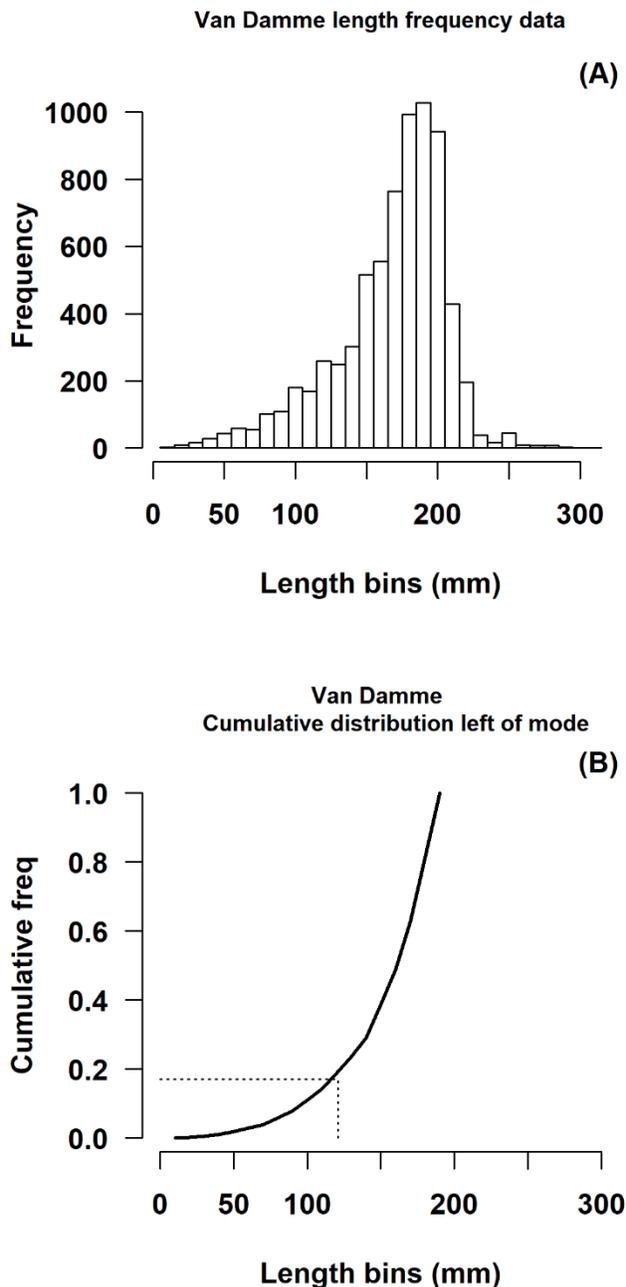


Figure A4. Empirical analysis of actual length-frequency distribution at Van Damme for purpose of calculating length at which 50% of population is mature ( $L_{50}$ ). (A) Pooled CDFW and Reef Check data across all sampled years. Pooling provides most comprehensive dataset reflecting emergence patterns (i.e., the left-hand side of the length frequency distribution). (B) Cumulative density function constructed using length bins to the left of the distribution mode. Dashed lines show cumulative probability associated with  $L_{50}=120.8$ , which was obtained from histological examination presented in Rogers-Bennett et al. (2004).

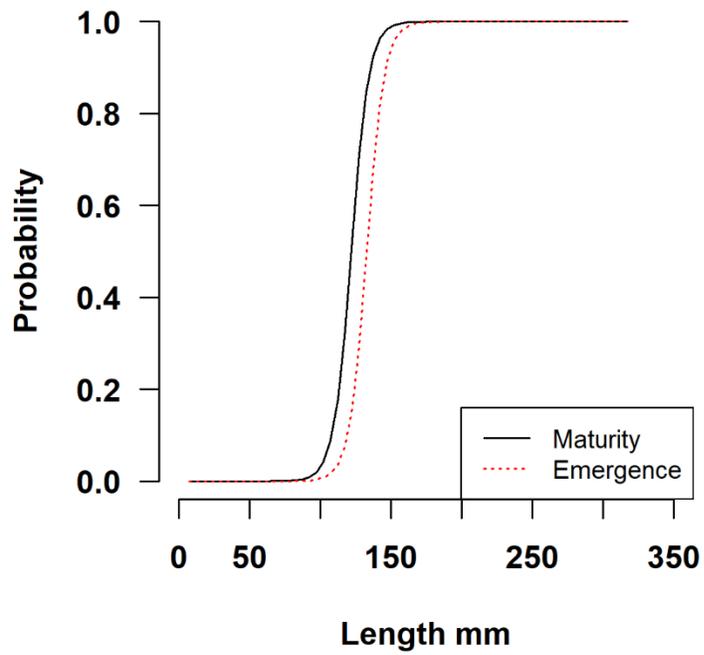


Figure A5. Example simulation of maturity and emergence curves as functions of length, according to the emergence pattern parameterized using actual data from Van Damme.

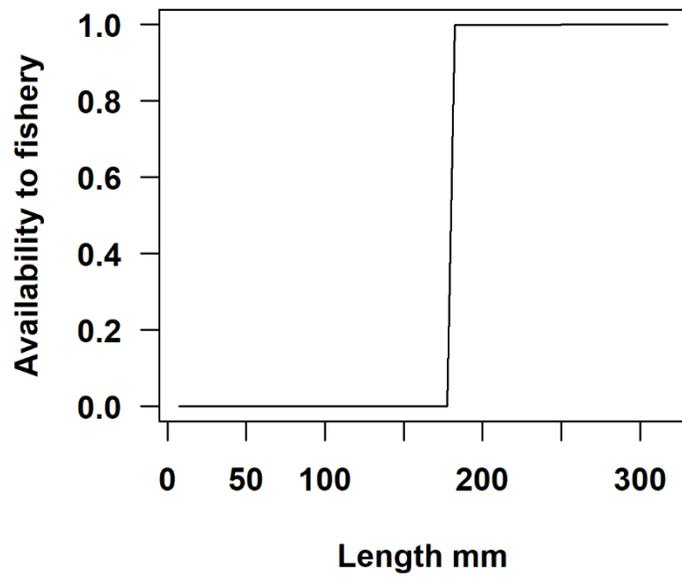


Figure A6. Simulated availability of red abalone to the fishery.

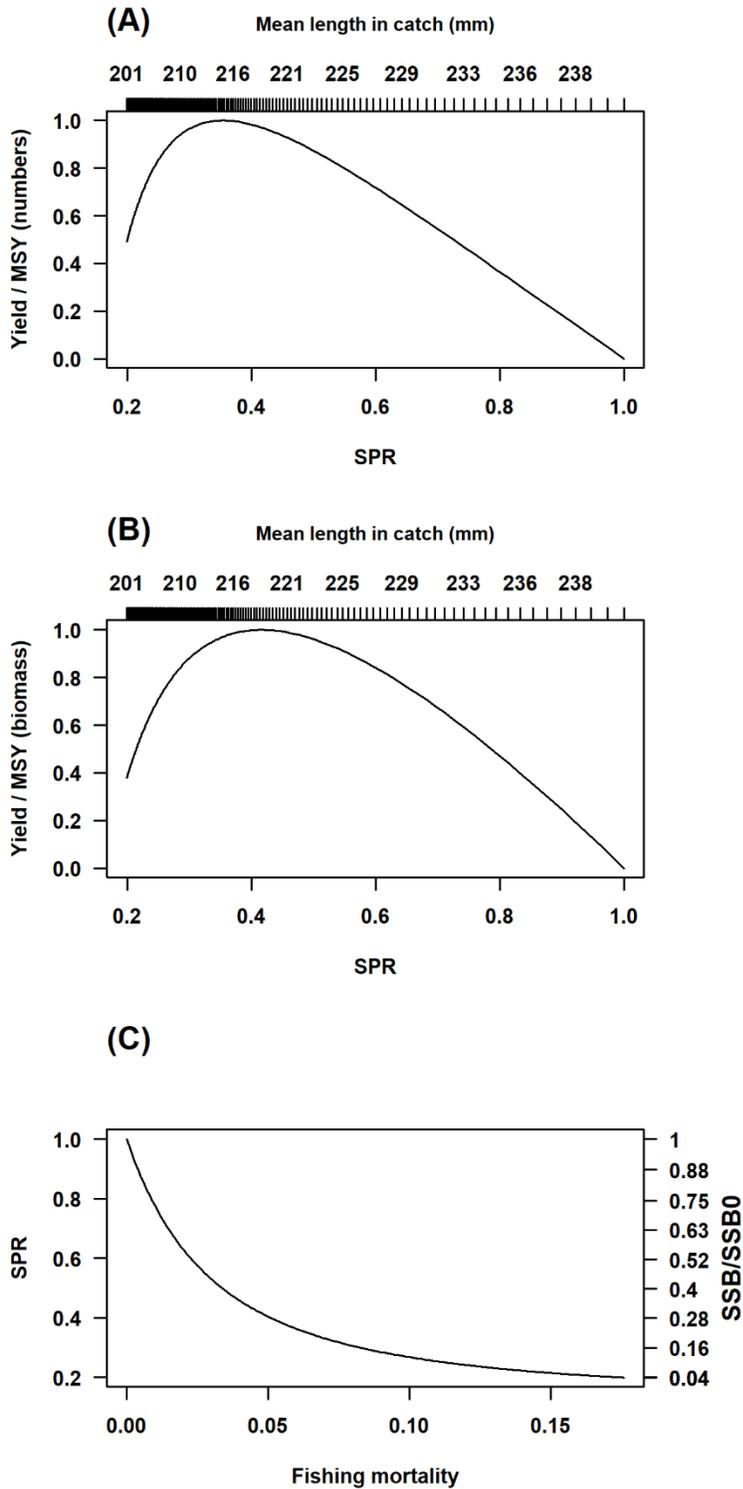


Fig. A7. Simulated equilibrium yield curve (A) number and (B) biomass versus SPR and mean length. (C) reproductive output (SPR and  $SSB/SSB_0$ ) versus fishing mortality rate.  $SSB$  is spawning output in eggs,  $SSB_0$  is unfished egg production, SPR is spawning potential ratio.

## Appendix B. Simulating historical abundance scenarios

Like other data-limited fisheries, historical trends in abundance are not well established for red abalone. Simulated historical trends are required for use in the MSE because these trends are used simulating observational data for use in harvest control rules. Accordingly, our simulations were used to re-constructed scenarios about red abalone stock dynamics from 2002 to 2016. In the simulating stock dynamics, we tuned site-specific unfished recruitment ( $R0$ ), initial depletion, and the time-varying component of natural mortality. Tuning was conducted such that simulated red abalone stock was reasonably consistent with the following metrics derived from actual data collected by CDFW and Reef Check (see Appendix E):

- Observed catches between 2002 – 2016;
- Estimates of SPR calculated from CDFW and Reef Check length frequency data using LB-SPR method (Hordyk et al., 2015c). These estimates were calculated via a batch processing algorithm.
- Estimates of MSY calculated using catch-MSY method (Froese et al., 2017);
- Estimates of relative harvest rate (i.e., harvest rate relative to harvest rate producing MSY in numbers of red abalone) calculated from catch histories using catch-MSY method (Froese et al., 2017).

At each site, visual tuning was an iterative process of adjusting  $R0$  so that absolute stock size was scaled to reflect estimated MSY and adjusting initial depletion so that SPR was consistent with the estimate from LB-SPR and that relative harvest rate was consistent with the estimate from catch-MSY. Note that in our application to actual data, the catch-MSY approach is tracking abundance, MSY in numbers, and harvest rate is in numbers in relation to vulnerable abundance.

When these quantities were compared to the simulation historical trends, comparisons were made in the same currencies. In calculating initial depletion for sites that did not have SPR estimates from actual data, we used the median initial depletion from Mendocino and northward where this information and applied it to other sites in this region. Likewise, the same calculation was made for Sonoma and southward. In model tuning, stochastic elements were excluded. We did however, include the effects of the ENSO index on life history parameters according to the specified (deterministic) relationships between these variables. No red tide events were simulated with the exception of the 2011 severe red tide event, with an approximate mortality increase of  $0.29 \text{ year}^{-1}$ , which was forced to occur at sites in Sonoma and Marin counties (Rogers-Bennett et al., 2012)). The resulting simulated stock was reasonably consistent with actual SPR ratios, MSY, and relative harvest rate (Figs. B1, B2). Simulated MSY was typically within a positive 0.1% bias (range -0.2% to 0.2%, all sites) of estimates obtained from catch-MSY using the actual catch time series. Actual catches were reasonably reproduced in the simulated historical trends (Fig. B3).

A secondary step in model tuning was to recognize concerns about recent non-anthropogenic mortality sources on red abalone (i.e., Rogers-Bennett et al., 2012). We addressed this issue by forcing non-zero values for the time-varying component of natural mortality during the final 5 years of the historical time period (years 2102-2106). Because SPR trends tend to lag behind mortality changes, it is plausible that the most recent SPR estimates may not yet have picked up these mortality increases (although some sites did display declining SPR; Figs. B2 and B3). To produce a declining survival trend, we introduced a 20% increase in natural mortality during the final 5 years of the historical time period (Figs. B3, B4, B5). This 20% natural mortality increase was arbitrary, but was intended to reflect the possibility of recent losses of red abalone and the

possibility of requiring stock rebuilding. Collectively, model tuning produced two scenarios about historical stock trends between 2002 and 2016:

Scenario 1: *Historical natural mortality baseline.*

Scenario 2: *Historical natural mortality anomaly.*

As a final note, while tuning was based on deterministic trends, implementing these reconstructions during MSE simulation runs *did* include stochastic processes, and thus, each simulation run produced a somewhat unique reconstruction.

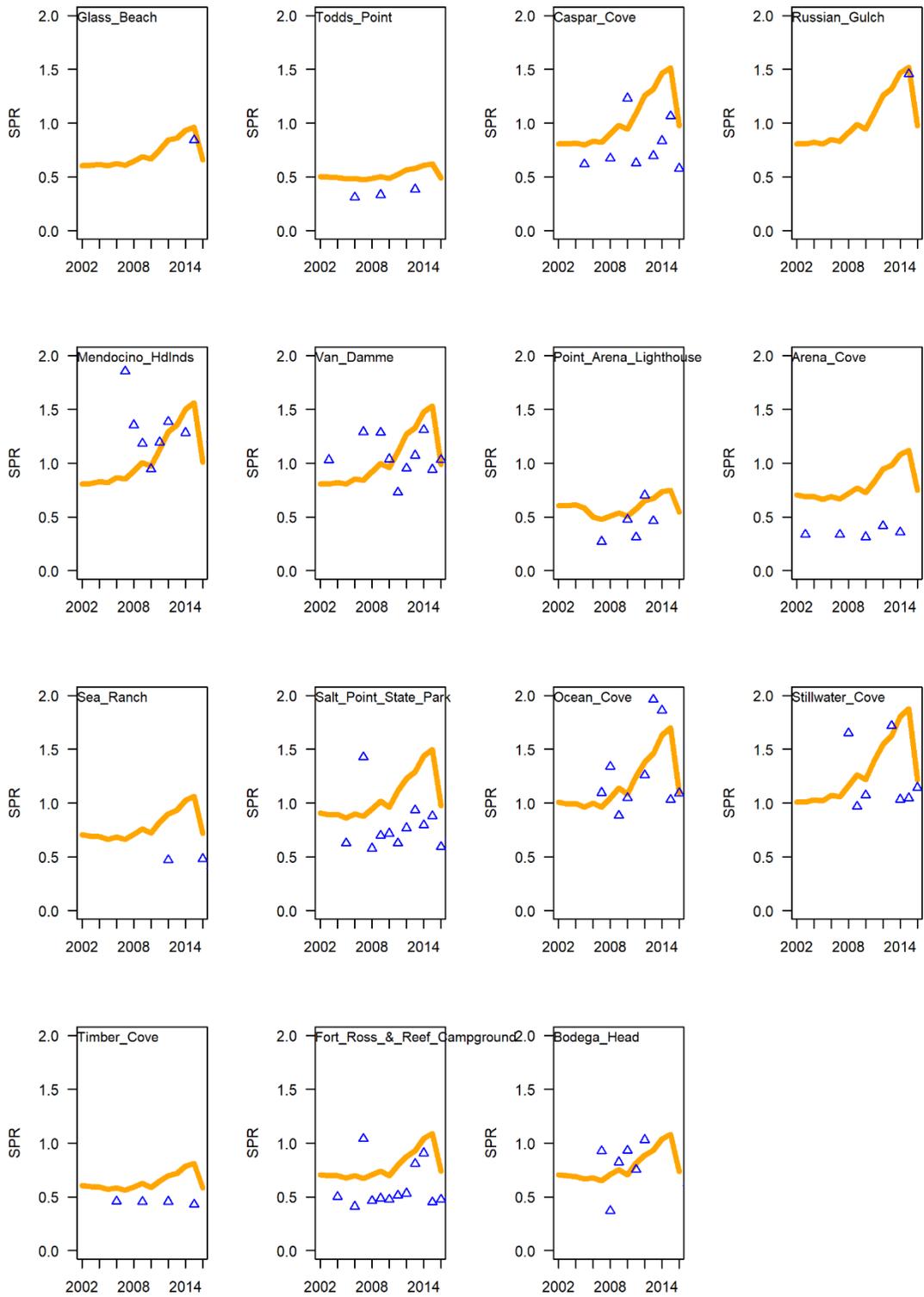


Figure B1. Scenario 1: *Historical natural mortality baseline*. Simulated SPR ratio (lines) and estimates of SPR ratios from length frequency distributions (triangles).

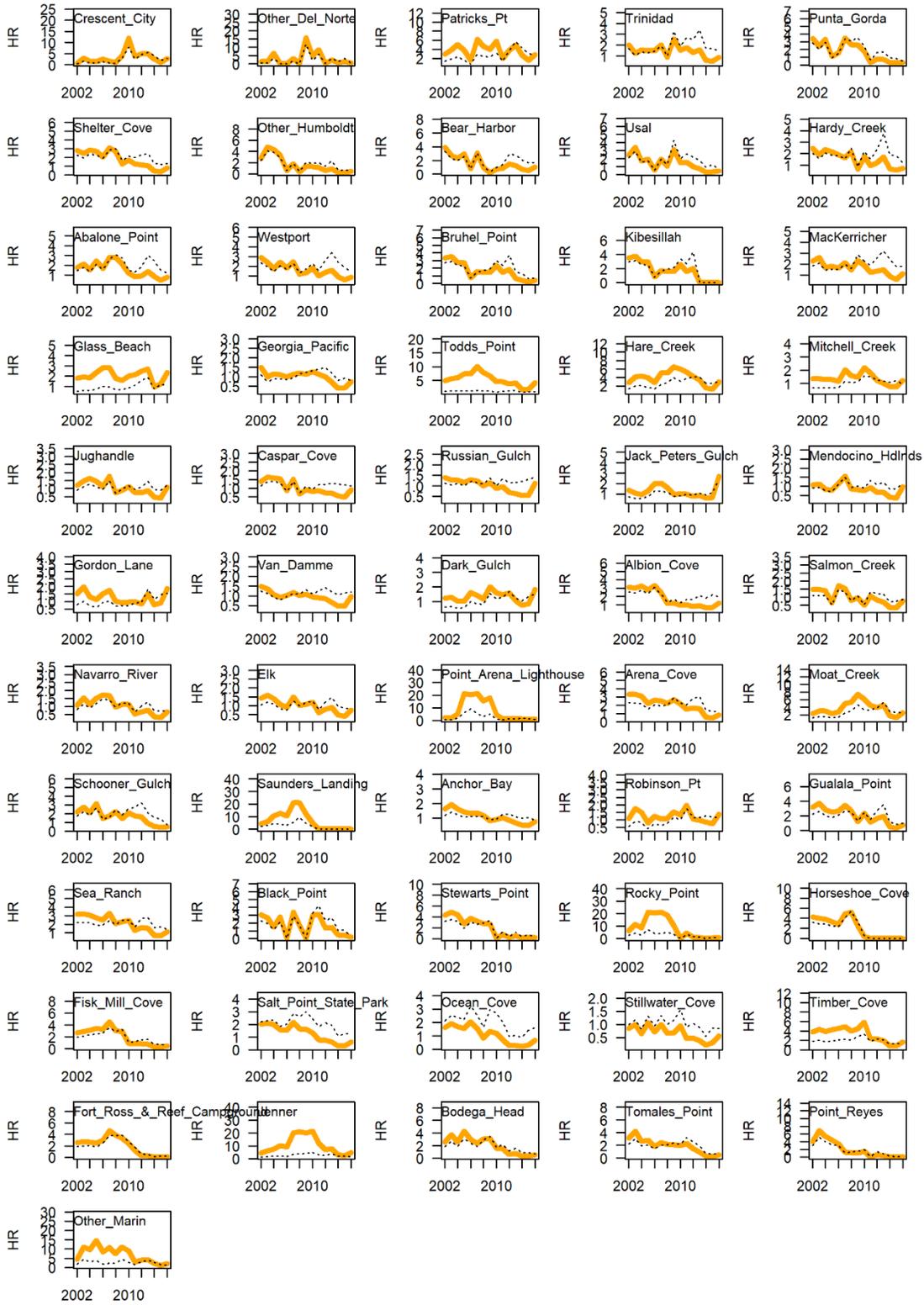


Figure B2. Scenario 1: *Historical natural mortality baseline*. Simulated harvest ratio (solid lines) and estimates of harvest ratio from catch-MSY method (dotted lines).

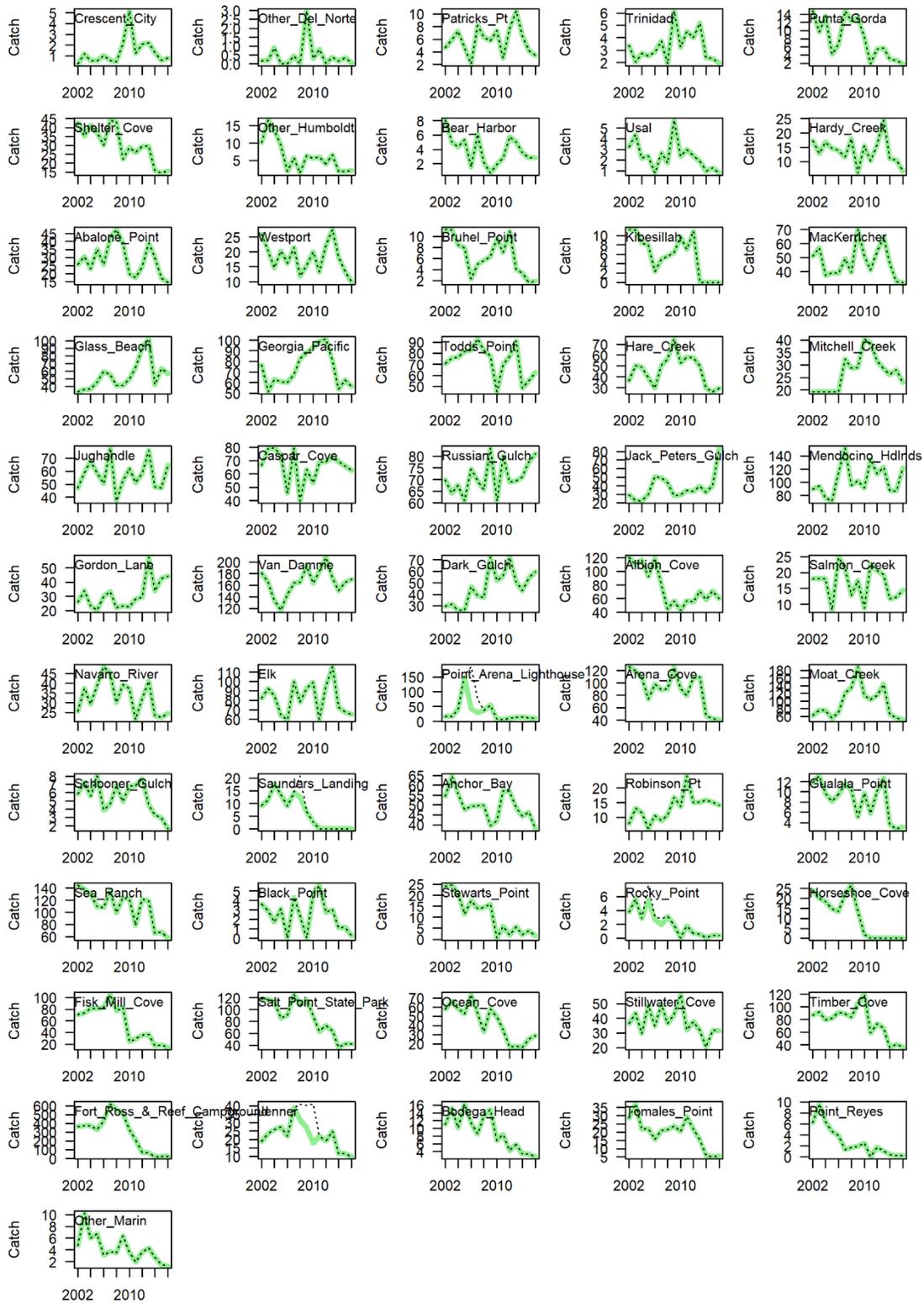


Figure B3. Scenario 1: *Historical natural mortality baseline*. Simulated catches in numbers x 100 (solid lines) and estimated catches (dotted lines).

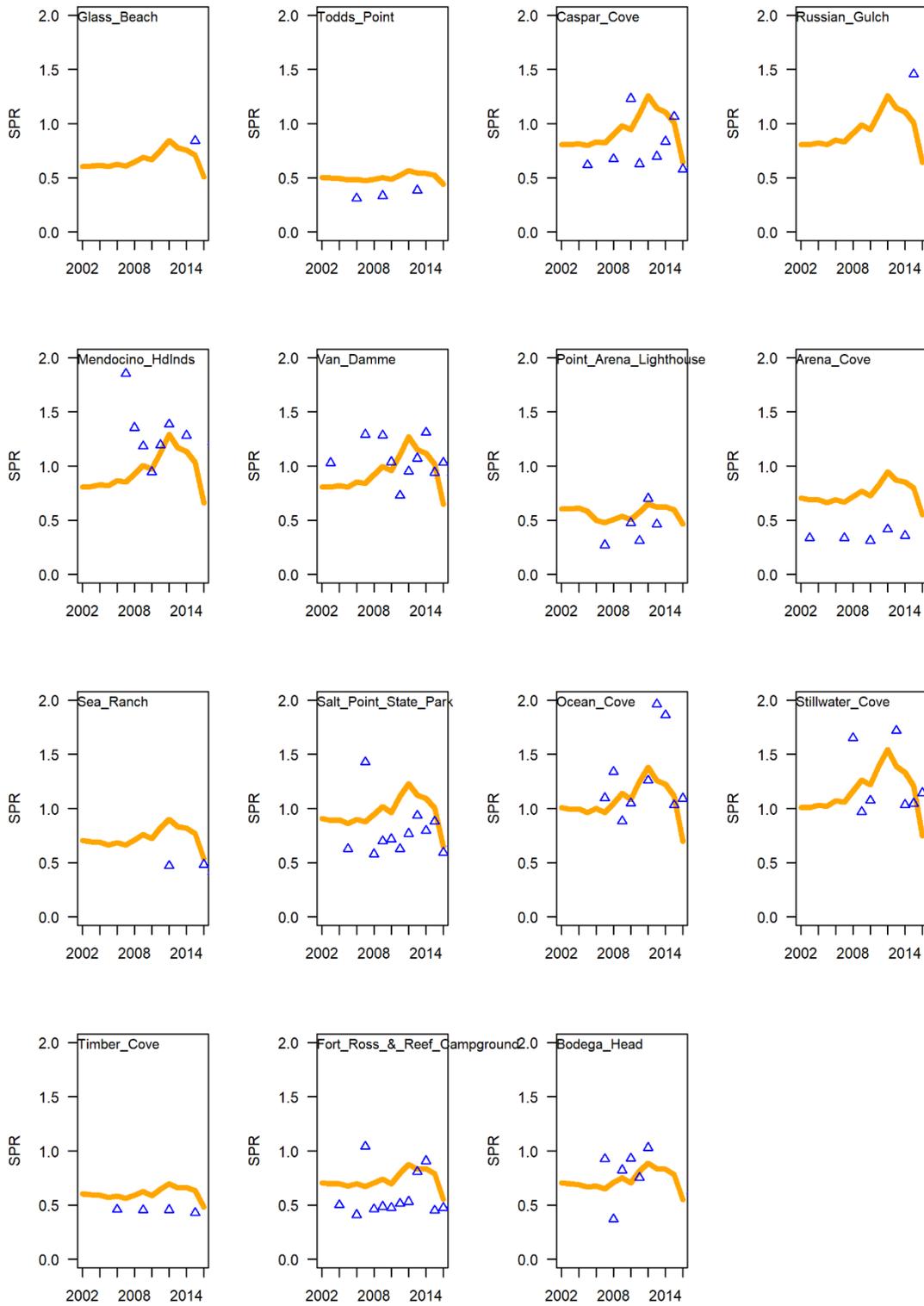


Figure B4. Scenario 2: *Historical natural mortality anomaly* Simulated SPR ratio (lines) and estimates of SPR ratios from length frequency distributions (triangles).

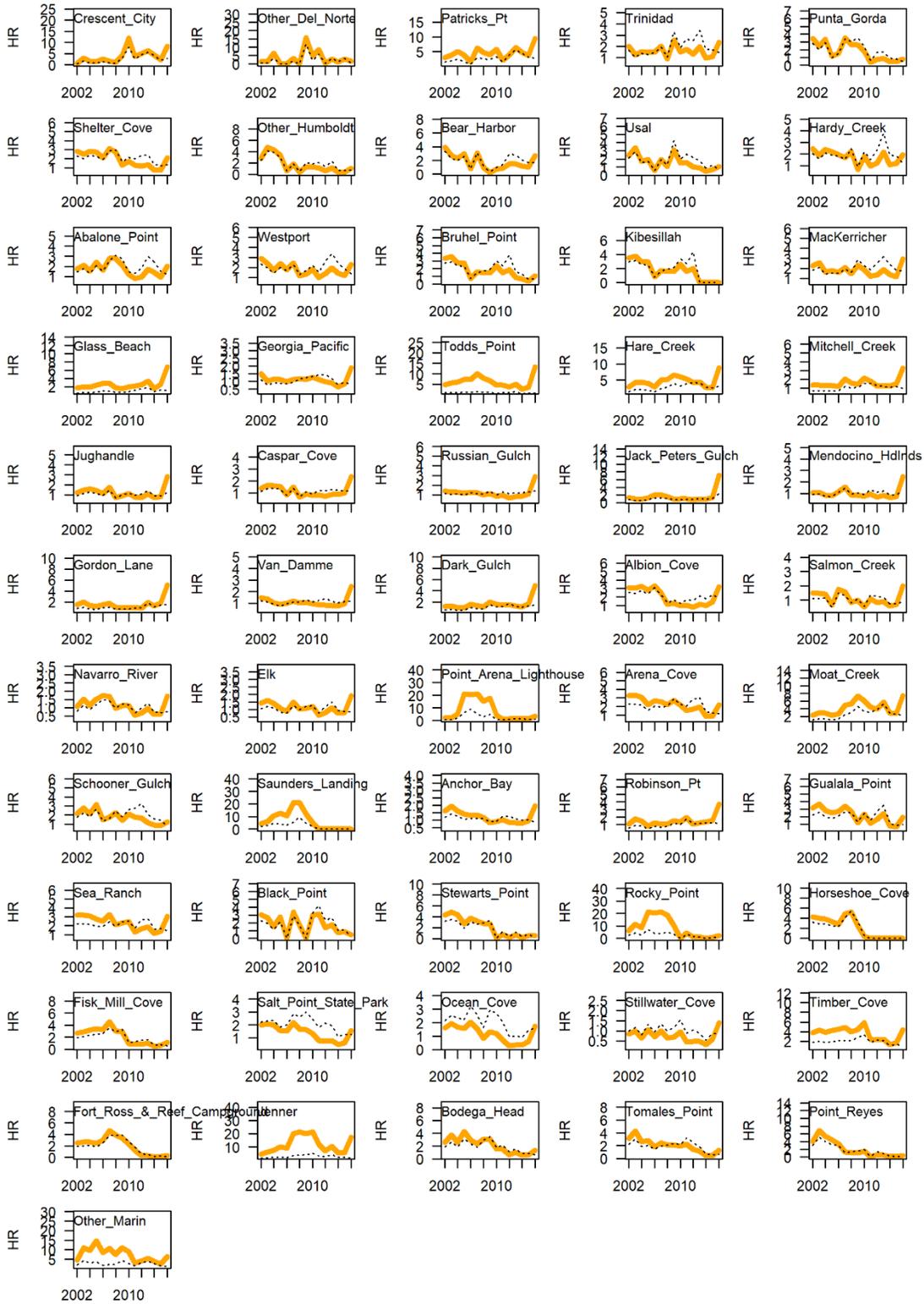


Figure B5. Scenario 2: *Historical natural mortality anomaly*. Simulated harvest ratio (solid lines) and estimates of harvest ratio from catch-MSY method (dotted lines).

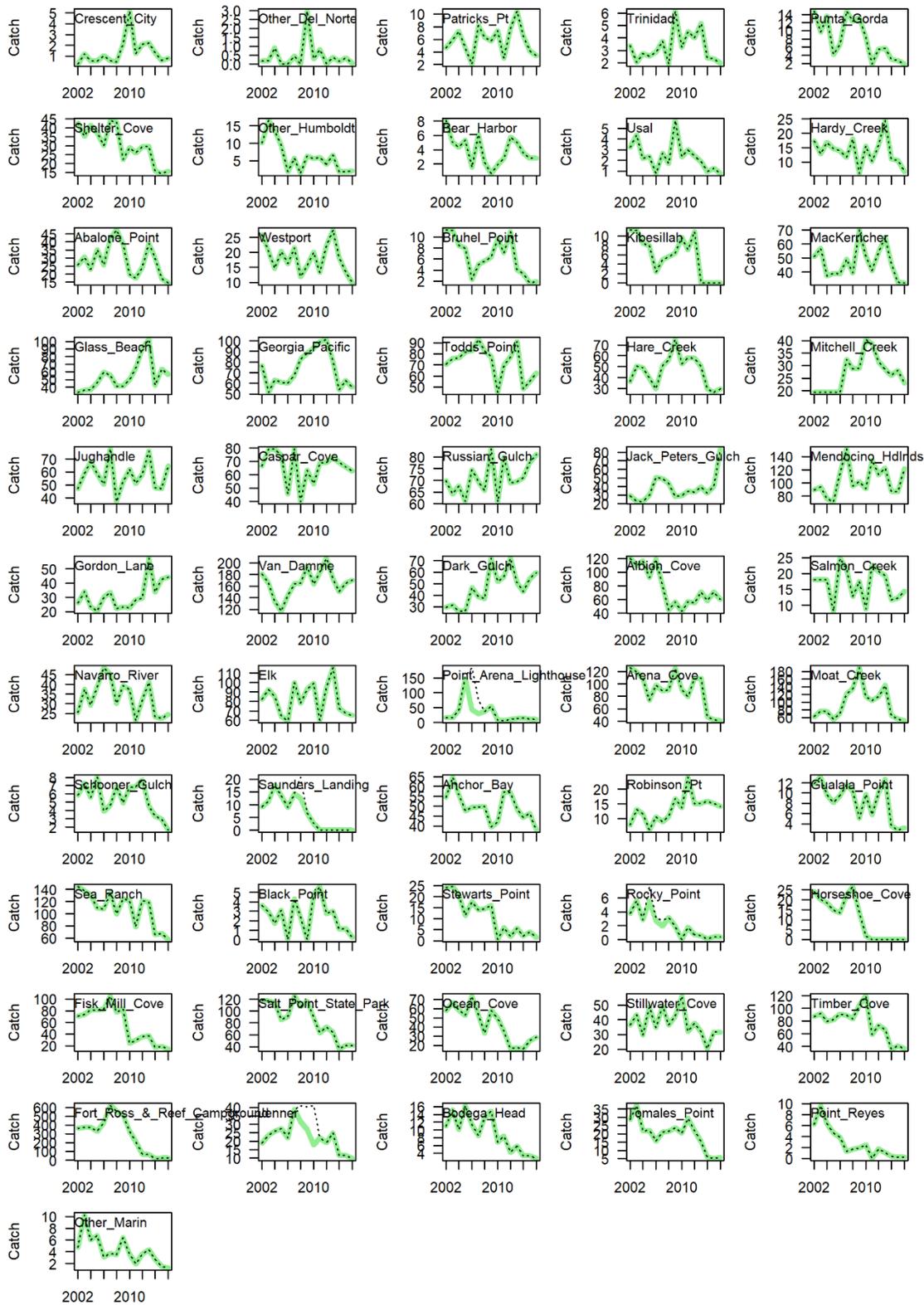


Figure B3. Scenario 2: *Historical natural mortality anomaly*. Simulated catches in numbers x 100 (solid lines) and estimated catches (dotted lines).

## Appendix C. Decision tree demonstration using actual data

This appendix is provided for the purpose of demonstrating practical application of the management strategies presented in this study. The authors reserve the right to modify this section and its results as necessary and as revisions and refinements to various methodologies occur. This appendix should not be interpreted as a comprehensive assessment of current stock status; instead, this appendix simply provides practical guidance for use of a harvest control rule.

### Section headings:

*Section i: Steps involved in harvest control rule calculations.*

*Section ii: Technical notes related to data usage.*

*Section iii. Summary of calculated indicator quantities 2002 - 2017*

*Section iv. Demonstration of TAC calculations for 2017*

*Section i: Steps involved in harvest control rule calculations.*

Full description of technical details in **METHODS: Indicator-based decision tree**

### Step 1. Gather datasets

- Gather catch histories (in numbers of red abalone) for 56 fishing sites.
- Gather length frequency data, all available sites and all years.

### Step 2. Calculate current harvest rate ratio at each site using catch-MSY

- Catch histories available from each of 56 fishing sites
- Sites that have become full or partial MPAs since 2002 are not included in catch-MSY calculations: Kibesillah, Saunders Landing, Horseshoe Cove, Point Arena Lighthouse, Stewarts Point, Rocky Point, Fisk Mill Cove, Fort Ross, Jenner, Bodega Head, Point Reyes
- Parameter inputs: See Table C1.
- Priors for  $B_0$ , start and end relative biomass, as per Martell and Froese (2012). This quantity varies by site.

### Step 3. Calculate SPR using LB-SPR method

- Uses length frequency data for all available sites and years in which sampling was conducted.
- Analysis for each site is carried out separately (no pooling data across sites)
- Parameter inputs: See Table C1.
- Left-hand side of length-frequency histogram used to calculate  $L_{50}$ ,  $L_{95}$ , and  $L_{\infty}$

- CVLinf set to 0.1, but modified up to 0.3 if statistical convergence could not be obtained using the default value;
- SPR and  $SPR_{MSY}$  calculated from length frequency truncated at 178 mm using LB-SPR R library, as per Hordyk et al. (2015c)
- For sites where sampling did not occur, median SPR ratio is used. Median SPR ratios were calculated separately for specified regions. For example, in this study we specified regions as Sonoma county and southward or Mendocino county and northward. Managers may be inclined to consider more or smaller regions or regions with boundaries that differed from the assumptions we made in constructing this analysis.
- Batch processing was used initially to calculate SPR ratios. Some sites required special consideration and were re-analysed as follows:
  - Todd's point,  $L_{\infty}$  seems quite high, results retained, but more sampling may necessary to confirm at this site.
  - Russian Gulch, distinct emergence differences were evident across sampling years, most recent sampling (2017) was used to estimate of  $L_{50}$ ,  $L_{95}$ , and  $L_{\infty}$ .
  - Point Arena Lighthouse, the left-hand side of the length frequency distribution has not been well sampled. This precluded estimation of  $L_{50}$ ,  $L_{95}$ , and  $L_{\infty}$ . This site was removed from analysis
  - Arena Cove, irregularities in 2013 length data. There appears to be a considerably different selectivity pattern This year was excluded
  - Sea Ranch, sampling occurring in 2012, 2016 and 2017 may have been conducted at different areas of this large site. It is unclear whether Sea Ranch should be considered a single site or multiple sites. The 2012 sampling produced very high (perhaps biologically implausible)  $L_{50}$ ,  $L_{95}$ , and  $L_{\infty}$  and was excluded.
  - Salt Point, low sample sizes in 2009, 2013 and 2017 were excluded from analysis
  - Timber Cove, low sample size in 2015 was excluded from analysis
  - Fort Ross, low sample sizes in 2007, 2008, 2010 were excluded from analysis
  - Bodega Head, low sample sizes in 2007, 2008, 2010, 2011, 2014 were excluded from analysis
  - Stillwater cove, small sample sizes produced highly inconsistent emergence trends. Site removed, suggest additional sampling at this site.

Step 4. Apply the selected decision tree separately to each site. Calculate the site-specific TACs in accordance with decision tree outcomes.

Step 5. Calculate regional TACs by summing across site-specific TACs

*Section ii: Technical notes related to data usage.*

Datasets used in this appendix:

- Catches (in numbers) at each of 56 sites, 2002-2016. Data curated by CDFW.
- Length composition 1999 to 2015 from CDFW diver surveys, various sites.
- Length composition 2007 to 2017 from Reef Check diver surveys, various sites.

LB-SPR method:

- CDFW and Reef Check length frequency data pooled (according to site and year in instances where both parties happened to sample the same site in a given year).
- Length composition data binned in 10 mm increments.
- Left-hand side of length frequency distributions were generated by pooling across all years at a given site (with year-specific exclusions noted above). This enabled the most comprehensive dataset possible that reflected site-specific emergence patterns of red abalone. Emergence informs maturity parameters, which are required inputs to LB-SPR fitting routine.
- SPR ratio calculations were made separately for 1999 to 2017. That is, no pooling across years of length frequency data provided to L-SPR fitting routine.
- SPR ratio calculation for years 2014-2016 were used as inputs to decision tree for 2017 TAC calculation (*Section iv*)

Catch-MSY method:

- Ratio of current harvest rate to harvest rate producing MSY is calculated for each site, in each year between 2002 and 2016 (*Section iii*).
- Ratio calculated for 2016 used as input for 2017 TAC calculation (*Section iv*)

Decision tree:

- Since all of the management strategies required adjusting current regulations in relation to the previous year's regulations, a necessary requirement for implementing such an approach is the specification of TACs (or other regulations) in the first year of use. Thus, to demonstrate a hypothetical TAC for 2017, we utilized 2016 site-specific catches as a frame of reference for specifying 2017 TAC.

### *Section iii. Summary of calculated indicator quantities 2002 - 2017*

Summary statistics for the estimates of SPR ratio and the harvest rate ratio are provided in Tables C2 and C3. At many sites, SPR decreases through time, although not all sites demonstrate this trend. At some sites, particularly in Sonoma county, the SPR is also below the level expected to produce MSY in the long-term. Harvest rates ratios vary considerably between sites, with many sites having higher rates during the early 2000's and lower rates in recent years.

### *Section iv. Demonstration of TAC calculations for 2017*

At the majority of sites, the decision trees that we considered recommended TAC reductions for the hypothetical application to 2017. The slower and faster rebuild decision trees differ under

the circumstance of “Extremely low” SPR ratio coupled with “Stable” or “High” harvest rate ratio. These conditions were not met at any site, thus the faster and slower rebuild strategies produced identical TACs (Tables C4 & C5). For our hypothetical application to 2017, length sampling occurred at 14 sites. In calculating regional TACs, the decision trees recommended overall reductions of 15.1% and 8.4% in the regions defined as Sonoma county-southward and Mendocino county-northward, respectively.

Table C1. Parameter used in catch-MSY and LB-SPR for actual TAC calculations.

Parameter	Value	Reference / Notes
$M/K$	0.9	Prince et al. (2015)
$L50/L\infty$	0.48	Obtained as the ratio of $L50$ from Rogers et al. (2004) to $L\infty$ from Rogers-Bennett et al. (2007).
$L95/L50$	1.15	Consistent with histology-based logistic curve fit to binary maturity data from Rogers et al. (2004)
$L$ - $W$ conversion $L=aW^b$	a=0.0001 b=3.03	Rogers-Bennett et al. (2007).
$L50$	Derived	Derived parameter using left-hand side of length frequency distribution. Site-specific; See Appendix A.
$L95$	Derived	Derived parameter using $L50$ and ratio $L95/L50$ . Site-specific.
$L\infty$	Derived	Derived parameter using $L50$ and ratio of $L50/L\infty$ . Site-specific.
CV $L\infty$	Derived	Values beginning with 0.1 and increasing to 0.3 are used to ensure fitting solutions are found using LB-SPR method.
SL50 and SL95	Derived	Estimated by LB-SPR, these two parameters define the selectivity of divers harvesting abalone.
Steepness used in calc. $SPR_{MSY}$	0.6	LB-SPR calculates SPR associated with MSY. Steepness reflects its use other stock assessments: Fu (2014); Gorfine et al. (2005); Rossetto et al. (2013); Zhang et al. (2007)
Intrinsic rate of increase ( $r$ )	U[0.05, 0.15]	Uniform prior, based on resilience criteria. See FishBase. Froese and Pauly (2011)

Table C2. Summary statistics for quantities estimated using length frequency data (i.e., outputs from the LB-SPR R library).

Site	Year	M/K	L50/L $\infty$	CV L $\infty$	L50	L95	L $\infty$	SL50	SL95	SPR	LC_SPR	UC_SPR	F/M	LC_F/M	UC_F/M	SampSize	Mean Length	SPRmsy	SPRratio	Note
Glass Beach	2015	0.9	0.48	0.1	130.0	149.5	270.8	179.8	180.8	0.35	0.30	0.40	2.24	1.60	2.88	157	203.3	0.41	0.84	
Todd's Point	2013															475	201.9	0.44	0.39	1,2
Todd's Point	2009															521	200.0	0.45	0.33	1,2
Todd's Point	2006															436	196.5	0.45	0.31	1,2
Caspar_Cove	2017	0.9	0.48	0.1	130.0	149.5	270.8	181.0	187.4	0.25	0.23	0.27	6.15	4.77	7.53	327	194.2	0.41	0.60	
Caspar_Cove	2016	0.9	0.48	0.1	130.0	149.5	270.8	176.7	177.5	0.24	0.23	0.25	4.52	3.85	5.19	345	194.2	0.41	0.58	
Caspar_Cove	2015	0.9	0.48	0.3	130.0	149.5	270.8	178.4	182.2	0.46	0.38	0.54	1.50	0.94	2.06	229	213.0	0.43	1.07	
Caspar_Cove	2014	0.9	0.48	0.2	130.0	149.5	270.8	194.1	220.3	0.39	0.30	0.48	2.81	1.19	4.43	170	212.2	0.47	0.84	
Caspar_Cove	2013	0.9	0.48	0.1	130.0	149.5	270.8	180.2	184.9	0.29	0.27	0.31	3.59	2.93	4.25	318	199.1	0.42	0.70	
Caspar_Cove	2011	0.9	0.48	0.1	130.0	149.5	270.8	180.6	185.7	0.26	0.25	0.27	5.21	4.48	5.94	547	196.2	0.41	0.63	
Caspar_Cove	2010	0.9	0.48	0.2	130.0	149.5	270.8	179.1	182.1	0.53	0.38	0.68	1.00	0.34	1.66	85	215.3	0.43	1.23	
Caspar_Cove	2008	0.9	0.48	0.1	130.0	149.5	270.8	181.0	187.1	0.28	0.26	0.30	4.06	3.48	4.64	662	199.3	0.41	0.68	
Caspar_Cove	2005	0.9	0.48	0.1	130.0	149.5	270.8	186.3	199.9	0.26	0.24	0.28	6.77	4.88	8.66	427	198.1	0.42	0.62	
Russian Gulch	2017	0.9	0.48	0.1	110	126.5	229.1667	189.43	205.51	0.53	0.44	0.62	4.47	1.18	7.76	119	195.5	0.54	0.99	2
Russian Gulch	2016	0.9	0.48	0.1	110	126.5	229.1667	180.6	185.08	0.53	0.49	0.57	2.74	1.78	3.70	134	193.7	0.49	1.09	2
Russian Gulch	2015															107				1,2
Russian Gulch	2014	0.9	0.48	0.1	110	126.5	229.1667	180.19	183.64	0.64	0.60	0.68	1.15	0.88	1.42	473	199.7	0.48	1.33	2
Jack_Peters_Gulch	2017															53				1
Mendocino_Hdlns	2017	0.9	0.48	0.1	110.0	126.5	229.2	180.2	184.5	0.59	0.53	0.65	1.59	0.96	2.22	136	196.9	0.48	1.22	
Mendocino_Hdlns	2016															57				1
Mendocino_Hdlns	2015															138				1
Mendocino_Hdlns	2014	0.9	0.48	0.1	110.0	126.5	229.2	179.7	183.4	0.61	0.54	0.68	1.40	0.79	2.01	129	197.0	0.48	1.28	
Mendocino_Hdlns	2012	0.9	0.48	0.2	110.0	126.5	229.2	178.2	181.5	0.57	0.51	0.63	1.25	0.81	1.69	255	201.1	0.41	1.39	
Mendocino_Hdlns	2011	0.9	0.48	0.1	110.0	126.5	229.2	179.5	182.1	0.56	0.50	0.62	1.80	1.18	2.42	171	194.5	0.47	1.20	
Mendocino_Hdlns	2010	0.9	0.48	0.3	110.0	126.5	229.2	179.5	182.8	0.36	0.33	0.39	4.81	3.14	6.48	128	192.7	0.38	0.95	
Mendocino_Hdlns	2009	0.9	0.48	0.2	110.0	126.5	229.2	179.7	181.5	0.50	0.45	0.55	2.06	1.30	2.82	151	197.5	0.42	1.18	
Mendocino_Hdlns	2008	0.9	0.48	0.3	110.0	126.5	229.2	178.1	181.3	0.50	0.43	0.57	1.56	0.92	2.20	180	203.2	0.37	1.35	
Mendocino_Hdlns	2007	0.9	0.48	0.1	110.0	126.5	229.2	178.9	182.4	0.86	0.76	0.96	0.23	0.01	0.45	267	205.5	0.46	1.86	

LC is the lower bound of the 95% CI; UC is the upper bound of the 95% CI; SampSize is the total sample size of the length-frequency observations > 178 mm; Lmean is mean length of observations > 178 mm. Notes designated as “1” indicate that no SPR estimate is provided because of lack of LB-SPR fitting routine convergence issues or parameter estimates being at parameter boundaries, or small sample size concerns; “2” indicates sites that were subject to more in-depth analysis beyond that provided by batch processing.

Table C2. Continued.

Site	Year	M/K	L50/L $\infty$	CV L $\infty$	L50	L95	L $\infty$	SL50	SL95	SPR	LC_SPR	UC_SPR	F/M	LC_F/M	UC_F/M	SampSize	Mean Length	SPRmsy	SPRratio	Note
Van_Damme	2017															55				1
Van_Damme	2016	0.9	0.48	0.1	120.0	138.0	250.0	179.9	183.5	0.37	0.34	0.40	3.55	2.69	4.41	204	194.5	0.36	1.03	
Van_Damme	2015															58				1
Van_Damme	2014															70				1
Van_Damme	2013	0.9	0.48	0.1	120.0	138.0	250.0	180.4	185.5	0.39	0.37	0.41	3.06	2.54	3.58	470	197.1	0.36	1.07	
Van_Damme	2012	0.9	0.48	0.2	120.0	138.0	250.0	179.3	182.4	0.37	0.32	0.42	2.99	1.83	4.15	103	197.5	0.39	0.95	
Van_Damme	2011	0.9	0.48	0.3	120.0	138.0	250.0	179.3	182.5	0.31	0.28	0.34	4.66	3.24	6.08	147	196.1	0.42	0.73	
Van_Damme	2010	0.9	0.48	0.1	120.0	138.0	250.0	179.7	183.2	0.37	0.35	0.39	3.54	3.02	4.06	560	195.1	0.36	1.04	
Van_Damme	2009	0.9	0.48	0.1	120.0	138.0	250.0	178.8	181.3	0.44	0.39	0.49	1.89	1.32	2.46	180	198.6	0.34	1.28	
Van_Damme	2008															72				1
Van_Damme	2007	0.9	0.48	0.1	120.0	138.0	250.0	179.1	182.3	0.45	0.42	0.48	1.89	1.60	2.18	670	200.0	0.35	1.29	
Van_Damme	2003	0.9	0.48	0.1	120.0	138.0	250.0	180.0	183.7	0.37	0.35	0.39	3.69	3.15	4.23	544	196.3	0.36	1.03	
Van_Damme	1999	0.9	0.48	0.1	120.0	138.0	250.0	180.6	184.9	0.36	0.34	0.38	4.23	3.59	4.87	505	195.0	0.37	0.98	
Point_Arena_Lighthouse	2013															37				1,2
Point_Arena_Lighthouse	2012															41				1,2
Point_Arena_Lighthouse	2011															3				1,2
Point_Arena_Lighthouse	2010															61				1,2
Point_Arena_Lighthouse	2007															114				1,2
Arena_Cove	2014															443				1,2
Arena_Cove	2013															16				1,2
Arena_Cove	2012															120				1,2
Arena_Cove	2011															11				1,2
Arena_Cove	2010															766				1,2
Arena_Cove	2007															595				1,2
Arena_Cove	2003															652				1,2
Sea_Ranch	2017	0.9	0.48	0.1	130.0	149.5	270.8	184.6	195.7	0.25	0.21	0.29	6.68	3.54	9.82	183	196.6	0.42	0.60	2
Sea_Ranch	2016	0.9	0.48	0.1	130.0	149.5	270.8	180.3	185.1	0.28	0.26	0.30	3.88	3.16	4.60	306	198.7	0.42	0.67	2
Sea_Ranch	2012	0.9	0.48	0.1	130.0	149.5	270.8	189.5	205.8	0.29	0.27	0.31	5.66	4.26	7.06	440	201.1	0.43	0.68	2
Salt_Point_State_Park	2017															64				1,2
Salt_Point_State_Park	2016	0.9	0.48	0.1	130.0	149.5	270.8	188.2	201.9	0.25	0.22	0.28	8.52	5.31	11.73	184	196.5	0.42	0.60	2
Salt_Point_State_Park	2015	0.9	0.48	0.1	130.0	149.5	270.8	181.1	184.4	0.36	0.27	0.45	2.29	1.44	3.14	82	205.9	0.41	0.88	2
Salt_Point_State_Park	2014	0.9	0.48	0.2	130.0	149.5	270.8	179.9	180.7	0.34	0.27	0.41	2.72	1.51	3.93	82	203.3	0.43	0.80	2
Salt_Point_State_Park	2013															28				1,2
Salt_Point_State_Park	2012	0.9	0.48	0.1	130.0	149.5	270.8	180.2	184.2	0.32	0.30	0.34	2.98	2.52	3.44	451	201.9	0.42	0.77	2
Salt_Point_State_Park	2011	0.9	0.48	0.1	130.0	149.5	270.8	179.6	180.3	0.26	0.23	0.29	4.18	2.83	5.53	104	194.6	0.41	0.63	2
Salt_Point_State_Park	2010	0.9	0.48	0.1	130.0	149.5	270.8	179.6	182.9	0.30	0.26	0.34	3.21	2.18	4.24	105	199.2	0.42	0.72	2
Salt_Point_State_Park	2009															64				1,2
Salt_Point_State_Park	2008	0.9	0.48	0.1	130.0	149.5	270.8	179.9	183.0	0.24	0.23	0.25	6.10	5.21	6.99	451	193.7	0.41	0.58	2
Salt_Point_State_Park	2007	0.9	0.48	0.1	130.0	149.5	270.8	178.4	181.2	0.59	0.48	0.70	0.67	0.34	1.00	168	216.1	0.41	1.43	2
Salt_Point_State_Park	2005	0.9	0.48	0.1	130.0	149.5	270.8	172.3	173.0	0.26	0.25	0.27	3.95	3.49	4.41	525	194.7	0.41	0.63	2
Salt_Point_State_Park	2000	0.9	0.48	0.1	130.0	149.5	270.8	181.4	188.0	0.25	0.24	0.26	6.18	5.03	7.33	460	194.9	0.41	0.61	2

Table C2. Continued

Site	Year	M/K	L50/L∞	CV L∞	L50	L95	L∞	SL50	SL95	SPR	LC_SPR	UC_SPR	F/M	LC_F/M	UC_F/M	SampSize	Mean Length	SPRmsy	SPRratio	Note
Ocean_Cove	2016	0.9	0.48	0.1	120.0	138.0	250.0	189.2	203.3	0.45	0.41	0.49	2.97	1.92	4.02	179	202.8	0.41	1.09	
Ocean_Cove	2015															59				1
Ocean_Cove	2014	0.9	0.48	0.2	120.0	138.0	250.0	179.8	182.7	0.72	0.53	0.91	0.45	0.00	0.91	104	217.6	0.39	1.86	
Ocean_Cove	2013	0.9	0.48	0.1	120.0	138.0	250.0	179.5	182.2	0.69	0.57	0.81	0.55	0.21	0.89	144	210.3	0.35	1.96	
Ocean_Cove	2012	0.9	0.48	0.1	120.0	138.0	250.0	179.8	182.7	0.45	0.42	0.48	1.90	1.52	2.28	398	200.9	0.36	1.26	
Ocean_Cove	2011															75				1
Ocean_Cove	2010	0.9	0.48	0.1	120.0	138.0	250.0	180.2	184.4	0.38	0.36	0.40	3.34	2.79	3.89	453	196.7	0.36	1.05	
Ocean_Cove	2009	0.9	0.48	0.3	120.0	138.0	250.0	178.7	181.4	0.37	0.30	0.44	2.82	1.53	4.11	89	201.2	0.42	0.88	
Ocean_Cove	2008	0.9	0.48	0.1	120.0	138.0	250.0	188.4	199.3	0.55	0.46	0.64	1.43	0.62	2.24	81	209.0	0.41	1.34	
Ocean_Cove	2007	0.9	0.48	0.1	120.0	138.0	250.0	179.7	183.2	0.39	0.37	0.41	2.97	2.55	3.39	636	196.8	0.36	1.10	
Stillwater_Cove	2016															166				1,2
Stillwater_Cove	2015															56				1,2
Stillwater_Cove	2014															102				1,2
Stillwater_Cove	2013															58				1,2
Stillwater_Cove	2012															130				1,2
Stillwater_Cove	2011															66				1,2
Stillwater_Cove	2010															178				1,2
Stillwater_Cove	2009															91				1,2
Stillwater_Cove	2008															122				1,2
Stillwater_Cove	2007															62				1,2
Timber_Cove	2015	0.9	0.48	0.1	140.0	161.0	291.7	191.6	208.2	0.19	0.16	0.22	11.06	6.99	15.13	226	199.2	0.44	0.43	
Timber_Cove	2012	0.9	0.48	0.1	140.0	161.0	291.7	180.8	186.2	0.20	0.18	0.22	5.64	4.61	6.67	302	198.6	0.44	0.46	
Timber_Cove	2009	0.9	0.48	0.1	140.0	161.0	291.7	186.5	200.6	0.20	0.18	0.22	7.15	5.59	8.71	586	199.7	0.44	0.46	
Timber_Cove	2006	0.9	0.48	0.1	140.0	161.0	291.7	180.3	185.1	0.20	0.19	0.21	5.73	5.14	6.32	877	197.9	0.44	0.46	
Fort_Ross_&_Reef_Campground	2017	0.9	0.48	0.1	130.0	149.5	270.8	180.2	183.2	0.30	0.27	0.33	3.37	2.66	4.08	233	200.1	0.42	0.72	2
Fort_Ross_&_Reef_Campground	2016	0.9	0.48	0.1	130.0	149.5	270.8	189.0	205.0	0.29	0.26	0.32	5.30	3.57	7.03	268	201.0	0.43	0.68	2
Fort_Ross_&_Reef_Campground	2015	0.9	0.48	0.1	130.0	149.5	270.8	188.8	206.2	0.28	0.25	0.31	5.61	3.95	7.27	420	200.2	0.43	0.65	2
Fort_Ross_&_Reef_Campground	2014	0.9	0.48	0.2	130.0	149.5	270.8	178.6	181.7	0.52	0.39	0.65	1.04	0.46	1.62	115	214.3	0.43	1.22	2
Fort_Ross_&_Reef_Campground	2013	0.9	0.48	0.1	130.0	149.5	270.8	180.1	183.0	0.61	0.47	0.75	0.66	0.25	1.07	101	218.4	0.42	1.47	2
Fort_Ross_&_Reef_Campground	2012	0.9	0.48	0.1	130.0	149.5	270.8	179.6	182.9	0.31	0.29	0.33	2.86	2.38	3.34	392	201.6	0.42	0.74	2
Fort_Ross_&_Reef_Campground	2011	0.9	0.48	0.1	130.0	149.5	270.8	190.3	209.1	0.31	0.24	0.38	4.41	1.88	6.94	99	202.8	0.43	0.72	2
Fort_Ross_&_Reef_Campground	2010															78				1,2
Fort_Ross_&_Reef_Campground	2009	0.9	0.48	0.1	130.0	149.5	270.8	181.2	187.1	0.29	0.27	0.31	3.87	3.30	4.44	553	199.6	0.41	0.71	2
Fort_Ross_&_Reef_Campground	2008															64				1,2
Fort_Ross_&_Reef_Campground	2007															77				1,2
Fort_Ross_&_Reef_Campground	2006	0.9	0.48	0.1	130.0	149.5	270.8	180.8	186.5	0.25	0.24	0.26	5.59	4.72	6.46	493	195.9	0.41	0.60	2
Fort_Ross_&_Reef_Campground	2004	0.9	0.48	0.1	130.0	149.5	270.8	179.4	182.7	0.31	0.29	0.33	2.97	2.46	3.48	371	201.1	0.42	0.75	2
Fort_Ross_&_Reef_Campground	2000	0.9	0.48	0.1	130.0	149.5	270.8	180.2	184.3	0.26	0.23	0.29	4.74	3.26	6.22	101	196.3	0.42	0.62	2
Fort_Ross_&_Reef_Campground	1999	0.9	0.48	0.1	130.0	149.5	270.8	180.5	185.0	0.25	0.23	0.27	5.60	4.57	6.63	294	194.8	0.41	0.61	2
Bodega_Head	2017	0.9	0.48	0.2	130.0	149.5	270.8	179.7	183.6	0.27	0.24	0.30	4.67	3.02	6.32	97	196.9	0.43	0.63	2
Bodega_Head	2014															36				1
Bodega_Head	2012	0.9	0.48	0.1	130.0	149.5	270.8	179.7	182.6	0.43	0.36	0.50	1.47	0.93	2.01	126	208.9	0.42	1.03	2
Bodega_Head	2011															72				1,2
Bodega_Head	2010															85				1,2
Bodega_Head	2009	0.9	0.48	0.1	130.0	149.5	270.8	186.4	201.8	0.35	0.31	0.39	2.77	1.92	3.62	261	205.8	0.43	0.82	2
Bodega_Head	2008															37				1,2
Bodega_Head	2007															39				1

Table C3. Summary statistics for quantities estimated using catch histories (i.e., outputs from catch-MSY; Martell and Froese (2012)). Empty rows indicate that data was available at these sites and years, but was not used in TAC calculations because of (\*) partial or no-take MPA.

Site	year	HRratio	LC	UC		Site	year	HRratio	LC	UC
Crescent_City	2002	0.33	0.20	0.59		Trinidad	2002	1.77	0.90	3.50
Crescent_City	2003	1.56	0.95	2.75		Trinidad	2003	1.09	0.56	2.15
Crescent_City	2004	0.77	0.47	1.36		Trinidad	2004	1.45	0.74	2.87
Crescent_City	2005	0.76	0.47	1.34		Trinidad	2005	1.35	0.69	2.67
Crescent_City	2006	1.40	0.86	2.47		Trinidad	2006	1.52	0.78	3.01
Crescent_City	2007	0.84	0.51	1.48		Trinidad	2007	2.02	1.03	3.99
Crescent_City	2008	0.66	0.40	1.16		Trinidad	2008	1.05	0.54	2.07
Crescent_City	2009	2.95	1.81	5.21		Trinidad	2009	3.38	1.73	6.68
Crescent_City	2010	7.84	4.80	13.84		Trinidad	2010	1.92	0.98	3.80
Crescent_City	2011	2.52	1.54	4.45		Trinidad	2011	2.73	1.39	5.39
Crescent_City	2012	4.51	2.76	7.96		Trinidad	2012	2.55	1.30	5.03
Crescent_City	2013	5.43	3.33	9.58		Trinidad	2013	3.42	1.74	6.75
Crescent_City	2014	3.93	2.41	6.94		Trinidad	2014	1.71	0.87	3.39
Crescent_City	2015	1.96	1.20	3.46		Trinidad	2015	1.67	0.85	3.31
Crescent_City	2016	2.64	1.62	4.66		Trinidad	2016	1.44	0.74	2.85
Other_Del_Norte	2002	0.72	0.43	1.25		Punta_Gorda	2002	2.99	1.54	5.94
Other_Del_Norte	2003	0.75	0.45	1.32		Punta_Gorda	2003	2.08	1.07	4.12
Other_Del_Norte	2004	3.44	2.07	6.02		Punta_Gorda	2004	3.01	1.56	5.99
Other_Del_Norte	2005	0.08	0.05	0.15		Punta_Gorda	2005	1.00	0.52	1.98
Other_Del_Norte	2006	0.08	0.05	0.14		Punta_Gorda	2006	1.57	0.81	3.11
Other_Del_Norte	2007	1.79	1.08	3.14		Punta_Gorda	2007	3.47	1.79	6.89
Other_Del_Norte	2008	0.08	0.05	0.14		Punta_Gorda	2008	3.13	1.61	6.21
Other_Del_Norte	2009	11.97	7.19	20.95		Punta_Gorda	2009	3.57	1.84	7.08
Other_Del_Norte	2010	2.07	1.24	3.62		Punta_Gorda	2010	2.42	1.25	4.82
Other_Del_Norte	2011	5.81	3.49	10.17		Punta_Gorda	2011	0.58	0.30	1.16
Other_Del_Norte	2012	0.26	0.15	0.45		Punta_Gorda	2012	1.62	0.83	3.21
Other_Del_Norte	2013	3.12	1.87	5.46		Punta_Gorda	2013	1.68	0.87	3.33
Other_Del_Norte	2014	1.32	0.79	2.31		Punta_Gorda	2014	0.91	0.47	1.81
Other_Del_Norte	2015	3.16	1.90	5.53		Punta_Gorda	2015	0.81	0.42	1.60
Other_Del_Norte	2016	0.57	0.34	0.99		Punta_Gorda	2016	0.52	0.27	1.03
Patricks_Pt	2002	1.46	0.88	2.57		Shelter_Cove	2002	2.34	1.26	4.48
Patricks_Pt	2003	1.90	1.14	3.35		Shelter_Cove	2003	2.03	1.09	3.89
Patricks_Pt	2004	2.45	1.48	4.32		Shelter_Cove	2004	2.45	1.31	4.69
Patricks_Pt	2005	1.66	1.00	2.94		Shelter_Cove	2005	2.30	1.23	4.40
Patricks_Pt	2006	0.76	0.46	1.35		Shelter_Cove	2006	1.95	1.04	3.72
Patricks_Pt	2007	3.00	1.81	5.30		Shelter_Cove	2007	2.91	1.56	5.56
Patricks_Pt	2008	2.51	1.51	4.43		Shelter_Cove	2008	3.03	1.62	5.80
Patricks_Pt	2009	2.41	1.46	4.27		Shelter_Cove	2009	1.67	0.90	3.20
Patricks_Pt	2010	3.27	1.97	5.77		Shelter_Cove	2010	2.18	1.17	4.16
Patricks_Pt	2011	1.44	0.87	2.55		Shelter_Cove	2011	2.01	1.08	3.84
Patricks_Pt	2012	3.72	2.25	6.58		Shelter_Cove	2012	2.34	1.25	4.47
Patricks_Pt	2013	5.60	3.38	9.90		Shelter_Cove	2013	2.44	1.31	4.66
Patricks_Pt	2014	4.27	2.57	7.54		Shelter_Cove	2014	1.36	0.73	2.61
Patricks_Pt	2015	3.06	1.85	5.41		Shelter_Cove	2015	1.22	0.65	2.33
Patricks_Pt	2016	2.67	1.61	4.71		Shelter_Cove	2016	1.31	0.70	2.51

Table C3. Continued

Site	year	HRratio	LC	UC	Site	year	HRratio	LC	UC
Other_Humboldt	2002	2.55	1.23	5.28	Abalone_Point	2002	1.51	0.79	2.96
Other_Humboldt	2003	4.34	2.09	9.01	Abalone_Point	2003	1.82	0.96	3.56
Other_Humboldt	2004	4.01	1.93	8.32	Abalone_Point	2004	1.40	0.73	2.74
Other_Humboldt	2005	3.03	1.46	6.28	Abalone_Point	2005	2.14	1.13	4.20
Other_Humboldt	2006	0.61	0.30	1.27	Abalone_Point	2006	1.61	0.85	3.15
Other_Humboldt	2007	1.86	0.90	3.86	Abalone_Point	2007	2.67	1.40	5.22
Other_Humboldt	2008	0.42	0.20	0.87	Abalone_Point	2008	3.18	1.67	6.22
Other_Humboldt	2009	2.04	0.98	4.24	Abalone_Point	2009	2.78	1.46	5.45
Other_Humboldt	2010	1.83	0.88	3.80	Abalone_Point	2010	1.50	0.79	2.93
Other_Humboldt	2011	1.99	0.96	4.14	Abalone_Point	2011	1.33	0.70	2.60
Other_Humboldt	2012	1.33	0.64	2.75	Abalone_Point	2012	1.83	0.96	3.59
Other_Humboldt	2013	2.22	1.07	4.61	Abalone_Point	2013	3.01	1.58	5.88
Other_Humboldt	2014	0.61	0.30	1.28	Abalone_Point	2014	2.45	1.29	4.80
Other_Humboldt	2015	0.62	0.30	1.28	Abalone_Point	2015	1.48	0.78	2.89
Other_Humboldt	2016	0.68	0.33	1.42	Abalone_Point	2016	1.23	0.65	2.41
Bear_Harbor	2002	3.48	1.74	7.05	Westport	2002	2.43	1.29	4.71
Bear_Harbor	2003	2.35	1.18	4.77	Westport	2003	2.00	1.06	3.87
Bear_Harbor	2004	2.11	1.06	4.28	Westport	2004	1.48	0.79	2.87
Bear_Harbor	2005	2.68	1.34	5.42	Westport	2005	2.08	1.10	4.02
Bear_Harbor	2006	0.81	0.41	1.64	Westport	2006	1.72	0.91	3.32
Bear_Harbor	2007	3.13	1.57	6.34	Westport	2007	2.30	1.22	4.45
Bear_Harbor	2008	1.20	0.60	2.44	Westport	2008	1.31	0.70	2.55
Bear_Harbor	2009	0.35	0.17	0.70	Westport	2009	1.72	0.92	3.34
Bear_Harbor	2010	1.00	0.50	2.02	Westport	2010	2.26	1.20	4.38
Bear_Harbor	2011	1.40	0.70	2.83	Westport	2011	1.55	0.82	3.00
Bear_Harbor	2012	2.93	1.47	5.93	Westport	2012	2.62	1.39	5.06
Bear_Harbor	2013	2.72	1.36	5.50	Westport	2013	3.38	1.80	6.55
Bear_Harbor	2014	2.00	1.00	4.04	Westport	2014	2.44	1.30	4.72
Bear_Harbor	2015	1.64	0.82	3.31	Westport	2015	1.86	0.99	3.61
Bear_Harbor	2016	1.65	0.83	3.34	Westport	2016	1.37	0.73	2.65
Usal	2002	2.19	1.07	4.46	Bruhel_Point	2002	2.84	1.48	5.59
Usal	2003	3.05	1.49	6.21	Bruhel_Point	2003	2.99	1.56	5.88
Usal	2004	1.62	0.79	3.29	Bruhel_Point	2004	2.43	1.27	4.78
Usal	2005	1.80	0.88	3.66	Bruhel_Point	2005	2.35	1.22	4.62
Usal	2006	0.59	0.29	1.21	Bruhel_Point	2006	0.72	0.37	1.41
Usal	2007	1.99	0.97	4.05	Bruhel_Point	2007	1.51	0.79	2.96
Usal	2008	1.31	0.64	2.66	Bruhel_Point	2008	1.69	0.88	3.33
Usal	2009	4.32	2.11	8.80	Bruhel_Point	2009	1.95	1.02	3.84
Usal	2010	1.97	0.97	4.02	Bruhel_Point	2010	3.02	1.58	5.94
Usal	2011	2.56	1.25	5.22	Bruhel_Point	2011	2.36	1.23	4.64
Usal	2012	2.14	1.05	4.36	Bruhel_Point	2012	3.75	1.95	7.37
Usal	2013	1.73	0.85	3.53	Bruhel_Point	2013	1.53	0.80	3.02
Usal	2014	0.89	0.44	1.82	Bruhel_Point	2014	1.26	0.66	2.47
Usal	2015	1.17	0.57	2.38	Bruhel_Point	2015	0.61	0.32	1.20
Usal	2016	0.69	0.34	1.41	Bruhel_Point	2016	0.68	0.36	1.34
Hardy_Creek	2002	2.12	1.10	4.18	Kibesillah	2002	*		
Hardy_Creek	2003	1.63	0.85	3.22	Kibesillah	2003	*		
Hardy_Creek	2004	2.15	1.12	4.24	Kibesillah	2004	*		
Hardy_Creek	2005	1.94	1.00	3.82	Kibesillah	2005	*		
Hardy_Creek	2006	1.89	0.98	3.73	Kibesillah	2006	*		
Hardy_Creek	2007	1.61	0.83	3.17	Kibesillah	2007	*		
Hardy_Creek	2008	2.53	1.31	4.99	Kibesillah	2008	*		
Hardy_Creek	2009	0.87	0.45	1.71	Kibesillah	2009	*		
Hardy_Creek	2010	2.24	1.16	4.41	Kibesillah	2010	*		
Hardy_Creek	2011	1.51	0.78	2.98	Kibesillah	2011	*		
Hardy_Creek	2012	2.35	1.22	4.63	Kibesillah	2012	*		
Hardy_Creek	2013	3.75	1.95	7.40	Kibesillah	2013	*		
Hardy_Creek	2014	1.86	0.96	3.66	Kibesillah	2014	*		
Hardy_Creek	2015	1.81	0.94	3.56	Kibesillah	2015	*		
Hardy_Creek	2016	1.16	0.60	2.28	Kibesillah	2016	*		

Table C3. Continued

Site	year	HRratio	LC	UC	Site	year	HRratio	LC	UC
MacKerricher	2002	1.89	1.01	3.62	Hare_Creek	2002	1.44	0.86	2.52
MacKerricher	2003	2.15	1.15	4.13	Hare_Creek	2003	2.05	1.23	3.60
MacKerricher	2004	1.45	0.77	2.78	Hare_Creek	2004	2.10	1.26	3.69
MacKerricher	2005	1.54	0.82	2.96	Hare_Creek	2005	1.76	1.06	3.09
MacKerricher	2006	1.54	0.82	2.96	Hare_Creek	2006	1.38	0.83	2.43
MacKerricher	2007	1.97	1.05	3.78	Hare_Creek	2007	2.41	1.45	4.24
MacKerricher	2008	1.62	0.86	3.10	Hare_Creek	2008	2.82	1.70	4.96
MacKerricher	2009	2.92	1.56	5.60	Hare_Creek	2009	4.03	2.42	7.08
MacKerricher	2010	2.28	1.22	4.37	Hare_Creek	2010	3.15	1.90	5.55
MacKerricher	2011	1.88	1.00	3.60	Hare_Creek	2011	3.76	2.26	6.61
MacKerricher	2012	2.55	1.36	4.89	Hare_Creek	2012	4.12	2.48	7.25
MacKerricher	2013	3.16	1.69	6.07	Hare_Creek	2013	4.04	2.43	7.10
MacKerricher	2014	2.38	1.27	4.57	Hare_Creek	2014	2.68	1.61	4.71
MacKerricher	2015	1.78	0.95	3.41	Hare_Creek	2015	2.49	1.50	4.37
MacKerricher	2016	1.76	0.94	3.38	Hare_Creek	2016	2.92	1.76	5.13
Glass_Beach	2002	0.58	0.27	1.29	Mitchell_Creek	2002	0.71	0.35	1.51
Glass_Beach	2003	0.63	0.29	1.41	Mitchell_Creek	2003	0.71	0.35	1.51
Glass_Beach	2004	0.64	0.29	1.42	Mitchell_Creek	2004	0.71	0.35	1.51
Glass_Beach	2005	0.79	0.36	1.77	Mitchell_Creek	2005	0.71	0.35	1.51
Glass_Beach	2006	1.03	0.48	2.31	Mitchell_Creek	2006	0.71	0.35	1.51
Glass_Beach	2007	0.97	0.45	2.18	Mitchell_Creek	2007	1.20	0.59	2.55
Glass_Beach	2008	0.73	0.34	1.64	Mitchell_Creek	2008	1.08	0.53	2.30
Glass_Beach	2009	0.73	0.34	1.63	Mitchell_Creek	2009	1.11	0.54	2.35
Glass_Beach	2010	0.88	0.40	1.96	Mitchell_Creek	2010	1.57	0.77	3.34
Glass_Beach	2011	1.16	0.53	2.59	Mitchell_Creek	2011	1.55	0.76	3.29
Glass_Beach	2012	1.62	0.75	3.63	Mitchell_Creek	2012	1.30	0.64	2.76
Glass_Beach	2013	1.90	0.88	4.25	Mitchell_Creek	2013	1.19	0.59	2.54
Glass_Beach	2014	0.82	0.38	1.82	Mitchell_Creek	2014	1.11	0.54	2.36
Glass_Beach	2015	1.23	0.56	2.74	Mitchell_Creek	2015	1.20	0.59	2.55
Glass_Beach	2016	1.11	0.51	2.48	Mitchell_Creek	2016	0.99	0.49	2.11
Georgia_Pacific	2002	1.04	0.58	1.95	Jughandle	2002	0.84	0.47	1.57
Georgia_Pacific	2003	0.70	0.39	1.31	Jughandle	2003	1.04	0.58	1.96
Georgia_Pacific	2004	0.85	0.48	1.60	Jughandle	2004	1.20	0.67	2.26
Georgia_Pacific	2005	0.82	0.46	1.53	Jughandle	2005	1.05	0.59	1.98
Georgia_Pacific	2006	0.81	0.45	1.50	Jughandle	2006	0.91	0.51	1.71
Georgia_Pacific	2007	0.90	0.50	1.68	Jughandle	2007	1.40	0.78	2.63
Georgia_Pacific	2008	1.10	0.62	2.06	Jughandle	2008	0.68	0.38	1.28
Georgia_Pacific	2009	1.18	0.66	2.21	Jughandle	2009	0.96	0.53	1.80
Georgia_Pacific	2010	1.27	0.71	2.37	Jughandle	2010	1.13	0.63	2.12
Georgia_Pacific	2011	1.36	0.76	2.55	Jughandle	2011	0.94	0.52	1.76
Georgia_Pacific	2012	1.43	0.80	2.67	Jughandle	2012	1.07	0.60	2.01
Georgia_Pacific	2013	1.16	0.65	2.17	Jughandle	2013	1.40	0.78	2.64
Georgia_Pacific	2014	0.78	0.44	1.45	Jughandle	2014	0.90	0.50	1.69
Georgia_Pacific	2015	0.89	0.50	1.67	Jughandle	2015	0.88	0.49	1.66
Georgia_Pacific	2016	0.80	0.45	1.49	Jughandle	2016	1.20	0.67	2.25
Todds_Point	2002	1.00	0.57	1.83	Caspar_Cove	2002	1.08	0.62	1.94
Todds_Point	2003	1.08	0.61	1.97	Caspar_Cove	2003	1.29	0.73	2.31
Todds_Point	2004	1.10	0.62	2.00	Caspar_Cove	2004	1.31	0.75	2.35
Todds_Point	2005	1.18	0.67	2.15	Caspar_Cove	2005	1.25	0.71	2.24
Todds_Point	2006	1.21	0.68	2.20	Caspar_Cove	2006	0.78	0.44	1.40
Todds_Point	2007	1.37	0.78	2.50	Caspar_Cove	2007	1.34	0.76	2.40
Todds_Point	2008	1.24	0.71	2.27	Caspar_Cove	2008	0.68	0.39	1.22
Todds_Point	2009	1.19	0.68	2.17	Caspar_Cove	2009	1.08	0.61	1.93
Todds_Point	2010	0.70	0.40	1.27	Caspar_Cove	2010	0.91	0.52	1.64
Todds_Point	2011	1.06	0.60	1.93	Caspar_Cove	2011	1.17	0.67	2.11
Todds_Point	2012	1.18	0.67	2.15	Caspar_Cove	2012	1.19	0.68	2.14
Todds_Point	2013	1.40	0.80	2.56	Caspar_Cove	2013	1.26	0.72	2.27
Todds_Point	2014	0.76	0.43	1.38	Caspar_Cove	2014	1.21	0.69	2.18
Todds_Point	2015	0.85	0.48	1.55	Caspar_Cove	2015	1.16	0.66	2.09
Todds_Point	2016	0.96	0.55	1.75	Caspar_Cove	2016	1.12	0.64	2.00

Table C3. Continued

Site	year	HRratio	LC	UC	Site	year	HRratio	LC	UC
Russian_Gulch	2002	1.07	0.61	1.91	Van_Damme	2002	1.15	0.65	2.10
Russian_Gulch	2003	0.99	0.57	1.77	Van_Damme	2003	1.06	0.60	1.93
Russian_Gulch	2004	1.04	0.60	1.86	Van_Damme	2004	0.87	0.49	1.58
Russian_Gulch	2005	0.94	0.54	1.68	Van_Damme	2005	0.75	0.42	1.36
Russian_Gulch	2006	1.15	0.66	2.05	Van_Damme	2006	0.92	0.52	1.67
Russian_Gulch	2007	1.09	0.62	1.94	Van_Damme	2007	1.04	0.59	1.90
Russian_Gulch	2008	1.04	0.59	1.85	Van_Damme	2008	1.05	0.59	1.92
Russian_Gulch	2009	1.32	0.76	2.36	Van_Damme	2009	1.25	0.70	2.27
Russian_Gulch	2010	0.98	0.56	1.75	Van_Damme	2010	1.07	0.60	1.94
Russian_Gulch	2011	1.29	0.74	2.31	Van_Damme	2011	1.21	0.68	2.20
Russian_Gulch	2012	1.13	0.65	2.01	Van_Damme	2012	1.38	0.78	2.52
Russian_Gulch	2013	1.14	0.65	2.04	Van_Damme	2013	1.17	0.66	2.13
Russian_Gulch	2014	1.18	0.67	2.10	Van_Damme	2014	1.03	0.58	1.87
Russian_Gulch	2015	1.30	0.74	2.31	Van_Damme	2015	1.12	0.64	2.05
Russian_Gulch	2016	1.37	0.78	2.44	Van_Damme	2016	1.16	0.66	2.12
Jack_Peters_Gulch	2002	0.78	0.33	1.85	Dark_Gulch	2002	0.64	0.31	1.37
Jack_Peters_Gulch	2003	0.60	0.25	1.43	Dark_Gulch	2003	0.68	0.33	1.47
Jack_Peters_Gulch	2004	0.58	0.24	1.37	Dark_Gulch	2004	0.56	0.27	1.21
Jack_Peters_Gulch	2005	0.78	0.33	1.85	Dark_Gulch	2005	0.57	0.27	1.22
Jack_Peters_Gulch	2006	1.33	0.56	3.15	Dark_Gulch	2006	0.99	0.48	2.14
Jack_Peters_Gulch	2007	1.32	0.56	3.12	Dark_Gulch	2007	0.83	0.40	1.80
Jack_Peters_Gulch	2008	1.20	0.50	2.83	Dark_Gulch	2008	0.81	0.39	1.74
Jack_Peters_Gulch	2009	0.79	0.33	1.86	Dark_Gulch	2009	1.57	0.76	3.37
Jack_Peters_Gulch	2010	0.81	0.34	1.92	Dark_Gulch	2010	1.17	0.57	2.51
Jack_Peters_Gulch	2011	0.94	0.40	2.23	Dark_Gulch	2011	1.29	0.62	2.76
Jack_Peters_Gulch	2012	0.92	0.39	2.17	Dark_Gulch	2012	1.68	0.81	3.61
Jack_Peters_Gulch	2013	1.08	0.46	2.57	Dark_Gulch	2013	1.22	0.59	2.63
Jack_Peters_Gulch	2014	0.91	0.38	2.15	Dark_Gulch	2014	1.06	0.51	2.28
Jack_Peters_Gulch	2015	1.11	0.47	2.63	Dark_Gulch	2015	1.34	0.65	2.88
Jack_Peters_Gulch	2016	2.36	1.00	5.60	Dark_Gulch	2016	1.49	0.72	3.20
Mendocino_HdInds	2002	0.86	0.47	1.65	Albion_Cove	2002	2.64	1.40	5.14
Mendocino_HdInds	2003	0.90	0.49	1.73	Albion_Cove	2003	2.52	1.34	4.92
Mendocino_HdInds	2004	0.72	0.39	1.38	Albion_Cove	2004	2.83	1.50	5.52
Mendocino_HdInds	2005	0.67	0.37	1.28	Albion_Cove	2005	2.37	1.25	4.62
Mendocino_HdInds	2006	1.06	0.58	2.05	Albion_Cove	2006	3.17	1.68	6.19
Mendocino_HdInds	2007	1.41	0.78	2.71	Albion_Cove	2007	2.31	1.22	4.51
Mendocino_HdInds	2008	0.91	0.50	1.75	Albion_Cove	2008	1.33	0.70	2.59
Mendocino_HdInds	2009	0.97	0.53	1.87	Albion_Cove	2009	1.66	0.88	3.25
Mendocino_HdInds	2010	0.87	0.48	1.68	Albion_Cove	2010	1.27	0.67	2.47
Mendocino_HdInds	2011	1.28	0.70	2.47	Albion_Cove	2011	1.67	0.89	3.26
Mendocino_HdInds	2012	1.09	0.60	2.09	Albion_Cove	2012	1.63	0.86	3.18
Mendocino_HdInds	2013	1.19	0.65	2.29	Albion_Cove	2013	2.12	1.12	4.15
Mendocino_HdInds	2014	0.85	0.47	1.65	Albion_Cove	2014	1.79	0.95	3.50
Mendocino_HdInds	2015	0.85	0.47	1.63	Albion_Cove	2015	2.22	1.17	4.32
Mendocino_HdInds	2016	1.18	0.65	2.26	Albion_Cove	2016	1.94	1.03	3.78
Gordon_Lane	2002	0.82	0.38	1.86	Salmon_Creek	2002	1.05	0.58	2.03
Gordon_Lane	2003	1.07	0.49	2.41	Salmon_Creek	2003	1.05	0.58	2.03
Gordon_Lane	2004	0.75	0.34	1.68	Salmon_Creek	2004	1.06	0.58	2.04
Gordon_Lane	2005	0.64	0.30	1.45	Salmon_Creek	2005	0.47	0.26	0.91
Gordon_Lane	2006	0.95	0.44	2.15	Salmon_Creek	2006	1.43	0.79	2.75
Gordon_Lane	2007	1.07	0.49	2.41	Salmon_Creek	2007	1.25	0.69	2.42
Gordon_Lane	2008	0.71	0.33	1.61	Salmon_Creek	2008	0.76	0.42	1.47
Gordon_Lane	2009	0.75	0.35	1.70	Salmon_Creek	2009	1.03	0.57	1.99
Gordon_Lane	2010	0.73	0.33	1.64	Salmon_Creek	2010	0.52	0.29	1.00
Gordon_Lane	2011	0.91	0.42	2.05	Salmon_Creek	2011	1.32	0.73	2.54
Gordon_Lane	2012	0.95	0.44	2.14	Salmon_Creek	2012	1.22	0.67	2.36
Gordon_Lane	2013	1.87	0.86	4.21	Salmon_Creek	2013	1.15	0.64	2.23
Gordon_Lane	2014	1.14	0.52	2.57	Salmon_Creek	2014	0.71	0.39	1.36
Gordon_Lane	2015	1.46	0.67	3.29	Salmon_Creek	2015	0.73	0.40	1.40
Gordon_Lane	2016	1.55	0.71	3.49	Salmon_Creek	2016	0.85	0.47	1.64

Table C3. Continued

Site	year	HRratio	LC	UC	Site	year	HRratio	LC	UC
Navarro_River	2002	0.75	0.42	1.42	Moat_Creek	2002	1.16	0.70	2.03
Navarro_River	2003	1.09	0.60	2.07	Moat_Creek	2003	1.48	0.89	2.59
Navarro_River	2004	0.85	0.47	1.61	Moat_Creek	2004	1.48	0.89	2.60
Navarro_River	2005	1.12	0.62	2.14	Moat_Creek	2005	1.12	0.67	1.96
Navarro_River	2006	1.42	0.79	2.69	Moat_Creek	2006	1.45	0.88	2.54
Navarro_River	2007	1.34	0.74	2.55	Moat_Creek	2007	2.55	1.54	4.48
Navarro_River	2008	0.92	0.51	1.74	Moat_Creek	2008	3.03	1.83	5.33
Navarro_River	2009	1.19	0.66	2.27	Moat_Creek	2009	4.61	2.78	8.09
Navarro_River	2010	1.15	0.64	2.19	Moat_Creek	2010	3.20	1.93	5.62
Navarro_River	2011	0.66	0.36	1.25	Moat_Creek	2011	3.15	1.90	5.53
Navarro_River	2012	0.98	0.54	1.86	Moat_Creek	2012	3.73	2.25	6.54
Navarro_River	2013	1.25	0.69	2.37	Moat_Creek	2013	5.20	3.14	9.12
Navarro_River	2014	0.72	0.40	1.36	Moat_Creek	2014	2.80	1.69	4.91
Navarro_River	2015	0.68	0.38	1.29	Moat_Creek	2015	2.54	1.53	4.45
Navarro_River	2016	0.73	0.40	1.39	Moat_Creek	2016	2.41	1.45	4.23
Elk	2002	0.99	0.55	1.88	Schooner_Gulch	2002	1.82	0.97	3.47
Elk	2003	1.12	0.62	2.11	Schooner_Gulch	2003	2.31	1.24	4.42
Elk	2004	1.04	0.58	1.95	Schooner_Gulch	2004	1.84	0.99	3.52
Elk	2005	0.78	0.43	1.47	Schooner_Gulch	2005	2.71	1.45	5.18
Elk	2006	0.72	0.40	1.35	Schooner_Gulch	2006	1.38	0.74	2.64
Elk	2007	1.19	0.67	2.25	Schooner_Gulch	2007	1.65	0.88	3.15
Elk	2008	0.95	0.53	1.79	Schooner_Gulch	2008	2.51	1.34	4.80
Elk	2009	1.13	0.63	2.14	Schooner_Gulch	2009	1.85	0.99	3.54
Elk	2010	1.22	0.68	2.29	Schooner_Gulch	2010	2.62	1.40	5.01
Elk	2011	0.74	0.41	1.41	Schooner_Gulch	2011	2.76	1.48	5.28
Elk	2012	1.16	0.65	2.20	Schooner_Gulch	2012	3.26	1.74	6.22
Elk	2013	1.43	0.80	2.70	Schooner_Gulch	2013	2.07	1.11	3.96
Elk	2014	0.92	0.51	1.73	Schooner_Gulch	2014	1.57	0.84	3.00
Elk	2015	0.85	0.47	1.60	Schooner_Gulch	2015	1.39	0.74	2.66
Elk	2016	0.81	0.45	1.53	Schooner_Gulch	2016	0.76	0.40	1.45
Point_Arena_Lighthouse	2002	*			Saunders_Landing	2002	*		
Point_Arena_Lighthouse	2003	*			Saunders_Landing	2003	*		
Point_Arena_Lighthouse	2004	*			Saunders_Landing	2004	*		
Point_Arena_Lighthouse	2005	*			Saunders_Landing	2005	*		
Point_Arena_Lighthouse	2006	*			Saunders_Landing	2006	*		
Point_Arena_Lighthouse	2007	*			Saunders_Landing	2007	*		
Point_Arena_Lighthouse	2008	*			Saunders_Landing	2008	*		
Point_Arena_Lighthouse	2009	*			Saunders_Landing	2009	*		
Point_Arena_Lighthouse	2010	*			Saunders_Landing	2010	*		
Point_Arena_Lighthouse	2011	*			Saunders_Landing	2011	*		
Point_Arena_Lighthouse	2012	*			Saunders_Landing	2012	*		
Point_Arena_Lighthouse	2013	*			Saunders_Landing	2013	*		
Point_Arena_Lighthouse	2014	*			Saunders_Landing	2014	*		
Point_Arena_Lighthouse	2015	*			Saunders_Landing	2015	*		
Point_Arena_Lighthouse	2016	*			Saunders_Landing	2016	*		
Arena_Cove	2002	2.36	1.27	4.51	Anchor_Bay	2002	1.12	0.64	2.06
Arena_Cove	2003	2.33	1.25	4.45	Anchor_Bay	2003	1.34	0.76	2.46
Arena_Cove	2004	2.23	1.20	4.26	Anchor_Bay	2004	1.18	0.67	2.16
Arena_Cove	2005	1.56	0.84	2.97	Anchor_Bay	2005	1.01	0.57	1.86
Arena_Cove	2006	2.09	1.12	3.99	Anchor_Bay	2006	1.05	0.60	1.93
Arena_Cove	2007	1.96	1.05	3.74	Anchor_Bay	2007	1.06	0.60	1.95
Arena_Cove	2008	2.04	1.10	3.88	Anchor_Bay	2008	1.08	0.61	1.98
Arena_Cove	2009	2.93	1.57	5.58	Anchor_Bay	2009	0.86	0.49	1.57
Arena_Cove	2010	2.25	1.21	4.29	Anchor_Bay	2010	0.91	0.52	1.67
Arena_Cove	2011	2.03	1.09	3.87	Anchor_Bay	2011	1.23	0.70	2.26
Arena_Cove	2012	2.85	1.53	5.43	Anchor_Bay	2012	1.24	0.70	2.27
Arena_Cove	2013	3.02	1.62	5.75	Anchor_Bay	2013	1.07	0.61	1.96
Arena_Cove	2014	1.38	0.74	2.63	Anchor_Bay	2014	0.98	0.55	1.79
Arena_Cove	2015	1.24	0.67	2.37	Anchor_Bay	2015	1.02	0.58	1.87
Arena_Cove	2016	1.17	0.63	2.24	Anchor_Bay	2016	0.83	0.47	1.52

Table C3. Continued

Site	year	HRatio	LC	UC	Site	year	HRatio	LC	UC
Robinson_Pt	2002	0.57	0.27	1.28	Stewarts_Point	2002	*		
Robinson_Pt	2003	0.95	0.44	2.11	Stewarts_Point	2003	*		
Robinson_Pt	2004	0.85	0.39	1.89	Stewarts_Point	2004	*		
Robinson_Pt	2005	0.44	0.21	0.98	Stewarts_Point	2005	*		
Robinson_Pt	2006	0.77	0.36	1.71	Stewarts_Point	2006	*		
Robinson_Pt	2007	0.65	0.30	1.44	Stewarts_Point	2007	*		
Robinson_Pt	2008	0.77	0.36	1.72	Stewarts_Point	2008	*		
Robinson_Pt	2009	1.21	0.56	2.70	Stewarts_Point	2009	*		
Robinson_Pt	2010	1.01	0.47	2.25	Stewarts_Point	2010	*		
Robinson_Pt	2011	1.79	0.83	3.98	Stewarts_Point	2011	*		
Robinson_Pt	2012	1.16	0.54	2.59	Stewarts_Point	2012	*		
Robinson_Pt	2013	1.19	0.55	2.64	Stewarts_Point	2013	*		
Robinson_Pt	2014	1.26	0.59	2.81	Stewarts_Point	2014	*		
Robinson_Pt	2015	1.22	0.56	2.71	Stewarts_Point	2015	*		
Robinson_Pt	2016	1.15	0.53	2.56	Stewarts_Point	2016	*		
Gualala_Point	2002	2.30	1.23	4.44	Rocky_Point	2002	*		
Gualala_Point	2003	2.65	1.42	5.14	Rocky_Point	2003	*		
Gualala_Point	2004	2.07	1.11	4.01	Rocky_Point	2004	*		
Gualala_Point	2005	1.80	0.96	3.48	Rocky_Point	2005	*		
Gualala_Point	2006	2.14	1.14	4.14	Rocky_Point	2006	*		
Gualala_Point	2007	2.69	1.44	5.20	Rocky_Point	2007	*		
Gualala_Point	2008	2.54	1.35	4.91	Rocky_Point	2008	*		
Gualala_Point	2009	1.30	0.69	2.51	Rocky_Point	2009	*		
Gualala_Point	2010	2.47	1.32	4.78	Rocky_Point	2010	*		
Gualala_Point	2011	1.52	0.81	2.94	Rocky_Point	2011	*		
Gualala_Point	2012	2.69	1.44	5.20	Rocky_Point	2012	*		
Gualala_Point	2013	3.53	1.88	6.82	Rocky_Point	2013	*		
Gualala_Point	2014	1.11	0.59	2.16	Rocky_Point	2014	*		
Gualala_Point	2015	0.83	0.44	1.60	Rocky_Point	2015	*		
Gualala_Point	2016	0.93	0.50	1.81	Rocky_Point	2016	*		
Sea_Ranch	2002	2.28	1.23	4.30	Horseshoe_Cove	2002	*		
Sea_Ranch	2003	2.25	1.22	4.25	Horseshoe_Cove	2003	*		
Sea_Ranch	2004	2.24	1.21	4.23	Horseshoe_Cove	2004	*		
Sea_Ranch	2005	1.94	1.05	3.67	Horseshoe_Cove	2005	*		
Sea_Ranch	2006	1.96	1.06	3.71	Horseshoe_Cove	2006	*		
Sea_Ranch	2007	2.51	1.36	4.75	Horseshoe_Cove	2007	*		
Sea_Ranch	2008	1.95	1.05	3.68	Horseshoe_Cove	2008	*		
Sea_Ranch	2009	2.50	1.35	4.72	Horseshoe_Cove	2009	*		
Sea_Ranch	2010	2.54	1.37	4.81	Horseshoe_Cove	2010	*		
Sea_Ranch	2011	1.72	0.93	3.25	Horseshoe_Cove	2011	*		
Sea_Ranch	2012	2.71	1.46	5.11	Horseshoe_Cove	2012	*		
Sea_Ranch	2013	2.78	1.50	5.25	Horseshoe_Cove	2013	*		
Sea_Ranch	2014	1.62	0.88	3.06	Horseshoe_Cove	2014	*		
Sea_Ranch	2015	1.68	0.91	3.18	Horseshoe_Cove	2015	*		
Sea_Ranch	2016	1.43	0.77	2.70	Horseshoe_Cove	2016	*		
Black_Point	2002	2.38	1.17	4.80	Fisk_Mill_Cove	2002	*		
Black_Point	2003	2.02	0.99	4.07	Fisk_Mill_Cove	2003	*		
Black_Point	2004	1.21	0.59	2.44	Fisk_Mill_Cove	2004	*		
Black_Point	2005	2.18	1.07	4.41	Fisk_Mill_Cove	2005	*		
Black_Point	2006	0.01	0.01	0.03	Fisk_Mill_Cove	2006	*		
Black_Point	2007	3.00	1.47	6.05	Fisk_Mill_Cove	2007	*		
Black_Point	2008	1.73	0.85	3.50	Fisk_Mill_Cove	2008	*		
Black_Point	2009	0.01	0.01	0.03	Fisk_Mill_Cove	2009	*		
Black_Point	2010	3.35	1.65	6.77	Fisk_Mill_Cove	2010	*		
Black_Point	2011	4.26	2.10	8.61	Fisk_Mill_Cove	2011	*		
Black_Point	2012	2.32	1.14	4.68	Fisk_Mill_Cove	2012	*		
Black_Point	2013	2.62	1.29	5.30	Fisk_Mill_Cove	2013	*		
Black_Point	2014	1.03	0.51	2.09	Fisk_Mill_Cove	2014	*		
Black_Point	2015	1.05	0.52	2.11	Fisk_Mill_Cove	2015	*		
Black_Point	2016	0.23	0.11	0.47	Fisk_Mill_Cove	2016	*		

Table C3. Continued

Site	year	HRratio	LC	UC	Site	year	HRratio	LC	UC
Salt_Point_State_Park	2002	2.29	1.23	4.39	Fort_Ross_&_Reef_Campground	2002	*		
Salt_Point_State_Park	2003	2.38	1.27	4.56	Fort_Ross_&_Reef_Campground	2003	*		
Salt_Point_State_Park	2004	2.41	1.29	4.63	Fort_Ross_&_Reef_Campground	2004	*		
Salt_Point_State_Park	2005	1.88	1.01	3.61	Fort_Ross_&_Reef_Campground	2005	*		
Salt_Point_State_Park	2006	2.05	1.10	3.94	Fort_Ross_&_Reef_Campground	2006	*		
Salt_Point_State_Park	2007	2.91	1.56	5.58	Fort_Ross_&_Reef_Campground	2007	*		
Salt_Point_State_Park	2008	2.66	1.42	5.10	Fort_Ross_&_Reef_Campground	2008	*		
Salt_Point_State_Park	2009	3.03	1.62	5.81	Fort_Ross_&_Reef_Campground	2009	*		
Salt_Point_State_Park	2010	2.46	1.32	4.73	Fort_Ross_&_Reef_Campground	2010	*		
Salt_Point_State_Park	2011	1.81	0.97	3.48	Fort_Ross_&_Reef_Campground	2011	*		
Salt_Point_State_Park	2012	2.12	1.14	4.07	Fort_Ross_&_Reef_Campground	2012	*		
Salt_Point_State_Park	2013	1.91	1.02	3.67	Fort_Ross_&_Reef_Campground	2013	*		
Salt_Point_State_Park	2014	1.08	0.58	2.08	Fort_Ross_&_Reef_Campground	2014	*		
Salt_Point_State_Park	2015	1.24	0.67	2.39	Fort_Ross_&_Reef_Campground	2015	*		
Salt_Point_State_Park	2016	1.25	0.67	2.39	Fort_Ross_&_Reef_Campground	2016	*		
Ocean_Cove	2002	2.21	1.16	4.36	Jenner	2002	*		
Ocean_Cove	2003	2.65	1.39	5.22	Jenner	2003	*		
Ocean_Cove	2004	2.45	1.28	4.83	Jenner	2004	*		
Ocean_Cove	2005	2.31	1.21	4.54	Jenner	2005	*		
Ocean_Cove	2006	3.33	1.74	6.57	Jenner	2006	*		
Ocean_Cove	2007	2.60	1.36	5.13	Jenner	2007	*		
Ocean_Cove	2008	1.69	0.88	3.33	Jenner	2008	*		
Ocean_Cove	2009	2.99	1.56	5.88	Jenner	2009	*		
Ocean_Cove	2010	2.71	1.42	5.33	Jenner	2010	*		
Ocean_Cove	2011	2.04	1.07	4.02	Jenner	2011	*		
Ocean_Cove	2012	0.97	0.51	1.91	Jenner	2012	*		
Ocean_Cove	2013	1.02	0.53	2.01	Jenner	2013	*		
Ocean_Cove	2014	0.90	0.47	1.77	Jenner	2014	*		
Ocean_Cove	2015	1.40	0.73	2.75	Jenner	2015	*		
Ocean_Cove	2016	1.60	0.84	3.14	Jenner	2016	*		
Stillwater_Cove	2002	0.94	0.52	1.80	Bodega_Head	2002	*		
Stillwater_Cove	2003	1.11	0.61	2.13	Bodega_Head	2003	*		
Stillwater_Cove	2004	0.76	0.42	1.46	Bodega_Head	2004	*		
Stillwater_Cove	2005	1.24	0.68	2.39	Bodega_Head	2005	*		
Stillwater_Cove	2006	0.90	0.50	1.73	Bodega_Head	2006	*		
Stillwater_Cove	2007	1.27	0.70	2.44	Bodega_Head	2007	*		
Stillwater_Cove	2008	0.95	0.53	1.84	Bodega_Head	2008	*		
Stillwater_Cove	2009	1.12	0.62	2.14	Bodega_Head	2009	*		
Stillwater_Cove	2010	1.48	0.82	2.85	Bodega_Head	2010	*		
Stillwater_Cove	2011	0.87	0.48	1.66	Bodega_Head	2011	*		
Stillwater_Cove	2012	1.01	0.56	1.94	Bodega_Head	2012	*		
Stillwater_Cove	2013	0.85	0.47	1.64	Bodega_Head	2013	*		
Stillwater_Cove	2014	0.53	0.29	1.03	Bodega_Head	2014	*		
Stillwater_Cove	2015	0.83	0.46	1.60	Bodega_Head	2015	*		
Stillwater_Cove	2016	0.81	0.45	1.57	Bodega_Head	2016	*		
Timber_Cove	2002	1.91	1.01	3.69	Tomales_Point	2002	2.35	1.21	4.68
Timber_Cove	2003	2.08	1.10	4.01	Tomales_Point	2003	3.17	1.64	6.32
Timber_Cove	2004	1.85	0.98	3.58	Tomales_Point	2004	2.00	1.03	3.99
Timber_Cove	2005	1.95	1.03	3.76	Tomales_Point	2005	2.07	1.07	4.13
Timber_Cove	2006	2.22	1.18	4.30	Tomales_Point	2006	1.51	0.78	3.02
Timber_Cove	2007	2.25	1.19	4.34	Tomales_Point	2007	2.03	1.05	4.05
Timber_Cove	2008	2.19	1.16	4.22	Tomales_Point	2008	2.12	1.10	4.23
Timber_Cove	2009	2.83	1.50	5.48	Tomales_Point	2009	2.39	1.24	4.77
Timber_Cove	2010	3.40	1.80	6.57	Tomales_Point	2010	2.16	1.11	4.30
Timber_Cove	2011	1.80	0.95	3.48	Tomales_Point	2011	3.17	1.64	6.32
Timber_Cove	2012	2.29	1.21	4.42	Tomales_Point	2012	2.52	1.30	5.02
Timber_Cove	2013	2.13	1.13	4.11	Tomales_Point	2013	1.82	0.94	3.63
Timber_Cove	2014	1.18	0.62	2.27	Tomales_Point	2014	0.70	0.36	1.41
Timber_Cove	2015	1.34	0.71	2.58	Tomales_Point	2015	0.59	0.30	1.17
Timber_Cove	2016	1.19	0.63	2.30	Tomales_Point	2016	0.64	0.33	1.28

Table C3. Continued

Site	year	HRratio	LC	UC
Point_Reyes	2002	*		
Point_Reyes	2003	*		
Point_Reyes	2004	*		
Point_Reyes	2005	*		
Point_Reyes	2006	*		
Point_Reyes	2007	*		
Point_Reyes	2008	*		
Point_Reyes	2009	*		
Point_Reyes	2010	*		
Point_Reyes	2011	*		
Point_Reyes	2012	*		
Point_Reyes	2013	*		
Point_Reyes	2014	*		
Point_Reyes	2015	*		
Point_Reyes	2016	*		
Other_Marin	2002	2.00	1.19	3.54
Other_Marin	2003	4.52	2.69	8.01
Other_Marin	2004	3.09	1.84	5.47
Other_Marin	2005	3.74	2.23	6.64
Other_Marin	2006	1.91	1.13	3.38
Other_Marin	2007	2.36	1.40	4.17
Other_Marin	2008	2.29	1.36	4.05
Other_Marin	2009	4.37	2.61	7.75
Other_Marin	2010	2.75	1.64	4.88
Other_Marin	2011	1.58	0.94	2.79
Other_Marin	2012	3.06	1.82	5.42
Other_Marin	2013	3.85	2.29	6.82
Other_Marin	2014	2.75	1.64	4.86
Other_Marin	2015	1.54	0.92	2.73
Other_Marin	2016	1.30	0.77	2.30

Table C4. Demonstration of TAC adjustments for Mendocino, Humboldt, and Del Norte counties as informed by two management strategies described in this study. Catch and TAC in numbers. Site names with \* indicates site-specific SPR ratios were available.

Site	Decision table inputs		SPR status	HR ratio	HR status	Slower rebuild		Faster rebuild	
	2016 Catch	SPR ratio				Adjust	2017 TAC	Adjust	2017 TAC
Crescent_City	79	1.03	Stable	2.64	High	-0.10	71.1	-0.10	71.1
Other_Del_Norte	6	1.03	Stable	0.57	Low	0.10	6.6	0.10	6.6
Patrick's_Pt	343	1.03	Stable	2.67	High	-0.10	308.7	-0.10	308.7
Trinidad	198	1.03	Stable	1.44	High	-0.10	178.2	-0.10	178.2
Punta_Gorda	182	1.03	Stable	0.52	Low	0.10	200.2	0.10	200.2
Shelter_Cove	1557	1.03	Stable	1.31	High	-0.10	1401.3	-0.10	1401.3
Other_Humboldt	209	1.03	Stable	0.68	Low	0.10	229.9	0.10	229.9
Bear_Harbor	282	1.03	Stable	1.65	High	-0.10	253.8	-0.10	253.8
Usal	77	1.03	Stable	0.69	Low	0.10	84.7	0.10	84.7
Hardy_Creek	669	1.03	Stable	1.16	High	-0.10	602.1	-0.10	602.1
Abalone_Point	1445	1.03	Stable	1.23	High	-0.10	1300.5	-0.10	1300.5
Westport	974	1.03	Stable	1.37	High	-0.10	876.6	-0.10	876.6
Bruhel_Point	188	1.03	Stable	0.68	Low	0.10	206.8	0.10	206.8
Kibesillah	0	1.03	Stable	NA		0.00	0	0.00	0
MacKerricher	3204	1.03	Stable	1.76	High	-0.10	2883.6	-0.10	2883.6
Glass_Beach *	5685	<b>0.84</b>	Low	1.11	High	-0.20	4548	-0.20	4548
Georgia_Pacific	5627	1.03	Stable	0.80	Stable	0.00	5627	0.00	5627
Todds_Point	6272	1.03	Stable	0.96	Stable	0.00	6272	0.00	6272
Hare_Creek	2949	1.03	Stable	2.92	High	-0.10	2654.1	-0.10	2654.1
Mitchell_Creek	2290	1.03	Stable	0.99	Stable	0.00	2290	0.00	2290
Jughandle	6464	1.03	Stable	1.20	High	-0.10	5817.6	-0.10	5817.6
Caspar_Cove *	6283	<b>0.58</b>	Low	1.12	High	-0.20	5026.4	-0.20	5026.4
Russian_Gulch *	8110	<b>1.09</b>	Stable	1.37	High	-0.10	7299	-0.10	7299
Jack_Peters_Gulch	8404	1.03	Stable	2.36	High	-0.10	7563.6	-0.10	7563.6
Mendocino_HdInds *	12222	<b>1.28</b>	High	1.18	High	-0.10	10999.8	-0.10	10999.8
Gordon_Lane	4424	1.03	Stable	1.55	High	-0.10	3981.6	-0.10	3981.6
Van_Damme *	17051	<b>1.03</b>	Stable	1.16	High	-0.10	15345.9	-0.10	15345.9
Dark_Gulch	5941	1.03	Stable	1.49	High	-0.10	5346.9	-0.10	5346.9
Albion_Cove	6016	1.03	Stable	1.94	High	-0.10	5414.4	-0.10	5414.4
Salmon_Creek	1449	1.03	Stable	0.85	Stable	0.00	1449	0.00	1449
Navarro_River	2447	1.03	Stable	0.73	Low	0.10	2691.7	0.10	2691.7
Elk	6506	1.03	Stable	0.81	Stable	0.00	6506	0.00	6506
Point_Arena_Lighthouse	1010	1.03	Stable	NA		0.00	1010	0.00	1010
Arena_Cove	4040	1.03	Stable	1.17	High	-0.10	3636	-0.10	3636
Moat_Creek	5132	1.03	Stable	2.41	High	-0.10	4618.8	-0.10	4618.8
Schooner_Gulch	161	1.03	Stable	0.76	Stable	0.00	161	0.00	161
Saunders_Landing	0	1.03	Stable	NA		0.00	0	0.00	0
Anchor_Bay	3785	1.03	Stable	0.83	Stable	0.00	3785	0.00	3785
Robinson_Pt	1414	1.03	Stable	1.15	High	-0.10	1272.6	-0.10	1272.6
Median SPR ratio		1.03							
Totals	133,095						121,921		121,921
Percent change							-8.40%		-8.40%

Table C5. Demonstration of TAC adjustments for Sonoma and Marin counties as informed by two management strategies described in this study. Catch and TAC in numbers.

	Decision table inputs				HR status	Slower rebuild		Faster rebuild	
	2016 Catch	SPR ratio	SPR status	HR ratio		Adjust	2017 TAC	Adjust	2017 TAC
Gualala_Point	321	0.67	Low	0.93	Stable	-0.10	288.9	-0.10	288.9
Sea_Ranch *	5723	<b>0.67</b>	Low	1.43	High	-0.20	4578.4	-0.20	4578.4
Black_Point	26	0.67	Low	0.23	Low	0.00	26	0.00	26
Stewarts_Point	153	0.67	Low	NA		-0.10	137.7	-0.10	137.7
Rocky_Point	39	0.67	Low	NA		-0.10	35.1	-0.10	35.1
Horseshoe_Cove	0	0.67	Low	NA		-0.10	0	-0.10	0
Fisk_Mill_Cove	1415	0.67	Low	NA		-0.10	1273.5	-0.10	1273.5
Salt_Point_State_Park *	4197	<b>0.60</b>	Low	1.25	High	-0.20	3357.6	-0.20	3357.6
Ocean_Cove *	2897	<b>1.09</b>	Stable	1.60	High	-0.10	2607.3	-0.10	2607.3
Stillwater_Cove	3147	0.67	Low	0.81	Stable	-0.10	2832.3	-0.10	2832.3
Timber_Cove *	3681	<b>0.43</b>	Extremely low	1.19	High	-0.20	2944.8	-0.20	2944.8
Fort_Ross_&_Reef_Campground *	2366	<b>0.68</b>	Low	NA		-0.10	2129.4	-0.10	2129.4
Jenner	963	0.67	Low	NA		-0.10	866.7	-0.10	866.7
Bodega_Head	263	0.67	Low	NA		-0.10	236.7	-0.10	236.7
Tomales_Point	561	0.67	Low	0.64	Low	0.00	561	0.00	561
Point_Reyes	31	0.67	Low	NA		-0.10	27.9	-0.10	27.9
Other_Marin	124	0.67	Low	1.30	High	-0.20	99.2	-0.20	99.2
Median SPR ratio		0.67							
Totals	25,907						22,003		22,003
							-15.1%		-15.1%

## **Appendix D. Technical considerations in using catch-MSY and LB-SPR**

This appendix is structured as a series of questions and answers related to technical aspects of the LB-SPR method for estimating spawning potential ratio (SPR) from length frequency data.

### **Question 1: Does LB-SPR produce reliable SPR estimates under steady-state conditions?**

We conducted simulation testing to evaluate the extent to which the SPR produced by the more complex operating model of red abalone agreed with the SPR estimates produced by LB-SPR. The input parameters needed for LB-SPR matched those used in the simulated data produced by the operating model, which allowed us to ask whether the simpler structural equations used in LB-SPR would produce reliable SPR estimates. We simulated equilibrium length distributions using the red abalone operating model that corresponded to “true” simulated SPR levels of 0.3, 0.4, 0.5, 0.6, and 0.7. Finally, we fit these simulated length frequencies (i.e., observed lengths sampled from multinomial distribution with effective sample size of 200 individuals, actual sample size 400 individuals). We repeated this process 100 times.

*Answer: Yes, LB-SPR produces reasonably reliable SPR estimates.*

At low simulated SPR, the estimated SPR tends to be negatively biased, but nevertheless often correctly indicates the overly depleted state of the stock. There is a positive bias at high simulated SPR, which likely reflects differences between the LB-SPR estimation routine and the red abalone operating model, namely in terms of the stock-recruitment relationship (i.e., steepness = 0.6) (Fig. D1).

### **Question 2: Do changes to length frequency data lag behind actual changes to underlying spawning biomass (or SPR)?**

It is well established that size-based indicators respond slowly to changes in fishing mortality, which can sometimes lead to delays in triggering TAC changes (Punt et al., 2001; Shin et al., 2005; Wayte and Klaer, 2010). As a demonstration of this effect, we simulated a 100-year projection using the HDT with only the LB-SPR indicator. This simulation was carried out under completely deterministic conditions (i.e., no stochastic recruitment and no temporal environmental variation or life history variation)

*Answer: Yes.*

Cyclic behavior of spawning biomass can emerge from delays in changes to length frequency distributions, which are then picked up and acted upon by the harvest control rule (Fig. D2).

**Question 3: How do dynamically changing recruitment, growth and survival affect SPR estimation?**

This is a complex question that was best addressed using the simulated outcomes of MSE. In the MSE, we retained SPR estimates that were calculated at each time step and we also recorded the true simulated SPR. Thus, we compared how SPR trends were estimated with respect to changing stock size as well as in response to environmental fluctuations.

*Answer: Some care must be taken in employing LB-SPR, but the careful integration of this approach with other indicators and an appropriate harvest control can produce reasonable management outcomes.*

We first simulated stable but low historical abundance, followed by stock rebuilding in years 15 through 40. Similar patterns were generated for 100 simulation runs (Fig. D3). This scenario highlights that at low simulated SPR, the estimated SPR tends to be slightly negatively biased, but nevertheless most often correctly indicates the overly depleted state of the stock. There is a positive bias at high simulated SPR, which likely reflects differences between the LB-SPR

estimation routine and the red abalone operating model, namely in terms of the stock-recruitment relationship (i.e., steepness = 0.6).

We then simulated declining historical abundance, followed by stock rebuilding in years 15 through 40 for 100 simulation runs (Fig. D4). During very rapid stock declines, changes in biomass outpace changes in length composition, and consequently biased SPR estimates are produced. The stabilizing of the length composition does result in reasonable SPR estimates, reduction of TACs, and stock rebuilding. Again, the examination of assessment methods in isolation inevitably will identify challenges facing any data limited assessment method. We therefore stress the need to consider the assessment pieces within the integrated harvest control rule and whether the integration and subsequent decision-making meets target management objectives.

**Question 4: Does increased natural mortality cause a decrease in SPR and is this decrease detected by the LB-SPR fitting routine?**

We simulated a stock in a stable state for 10 years, followed by an increase in  $M$  on all length classes of  $0.1 \text{ year}^{-1}$ . We then returned the natural mortality to its baseline rate for a subsequent 10 years. During this 30-year time period, fishing mortality was held constant at  $F_{MSY}$ . The resulting trends in stock dynamics demonstrate a simulated decline in SPR, followed by a rebounding once natural mortality was returned to its baseline rate (Fig. D5). We then simulated the observation of length frequency data at various points during this 30 year duration and used the LB-SPR fitting routine to estimate SPR (following the procedure used in the MSE for estimating SPR). We plotted the percent bias in SPR between the estimated values and true simulated values.

*Answer: Yes.*

SPR changes in response to periodic changes in natural mortality and this effect is detected using the LB-SPR fitting routine (Fig. D5). During the initial 10-year stable state, a negative bias is evident, as we have demonstrated in other plots in this appendix. During stock decline, the SPR estimate lags behind the changes in stock size, but later adjusts. Importantly, when natural mortality increases are driving changes in stock size, these changes will be picked up by the LB-SPR method. As noted previously, the examination of assessment methods in isolation inevitably will identify challenges facing any data limited assessment method. We therefore stress the need to consider the assessment pieces within the integrated harvest control rule and whether the integration and subsequent decision-making meets target management objectives.

**Question 5: Does catch-MSY produce reasonably reliable estimates of the harvest ratio?**

We conducted simulation testing to evaluate the extent to which the harvest ratio produced by the operating model matched the harvest ratio estimated using catch-MSY approach. Using the reconstructed abundance trends from 2002 to 2016, we subjected the corresponding simulated catches to catch-MSY estimation. We then estimated bias between the estimated harvest rate ratio and the “true simulated” harvest rate ratio from the operating model. In an extension, we ran the same analysis, but using catches that were under-reported by 50%. This scenario reflected some estimates that poaching could be as high as 100% of the reported catch (that is, total catch 2-times reported catch).

*Answer: Yes, catch-MSY produces reasonable estimates of the harvest ratio for red abalone, and does so (for complex reasons) under the under-reporting scenario that we simulated.*

On average, catch-MSY produces unbiased estimates of the harvest ratio (Fig. D6). To understand how underestimates and over-estimates are likely to affect decision tree performance, we categorized each estimate according to whether the correct indicator category of was chosen. We found that in 41% of estimates were assigned to the correct category, 39% of estimates were assigned to a higher harvest rate than they should have been, and 20% of estimates were assigned to a lower harvest rate category. It is important to consider the performance of the catch-MSY metric in the context of overall management strategy performance (see main text). However, when erroneous status determinations were made, these errors tended to be made in a direction that worked with the SPR indicator to maintain current catches or to reduce catches.

When we conducted the same analysis but with 50% under-reporting, we found consistent results (Fig. D6). In this case, 42% of harvest ratio estimates were correctly assigned, 38% of estimates were assigned to a higher category that they should have been, and 20% were assigned to a lower category. The reason for similar performance reflects the way in which scaling of vulnerable abundance occurs in the Schaefer surplus production model. The  $BO$  parameter is scaled according to the catch history input. When under-reporting occurs (at the same rate each year) a lower  $BO$  parameter is estimated by the model that would have been obtained had “true” catches been used as input. Because harvest rate is a ratio of catches to current stock size, we still obtain a reasonable estimate of this ratio, even during under-reporting. Readers should be cautioned that patterns of poaching can be more complex than those that we simulated and can affect estimator reliability in a manner that is different than we reported here.

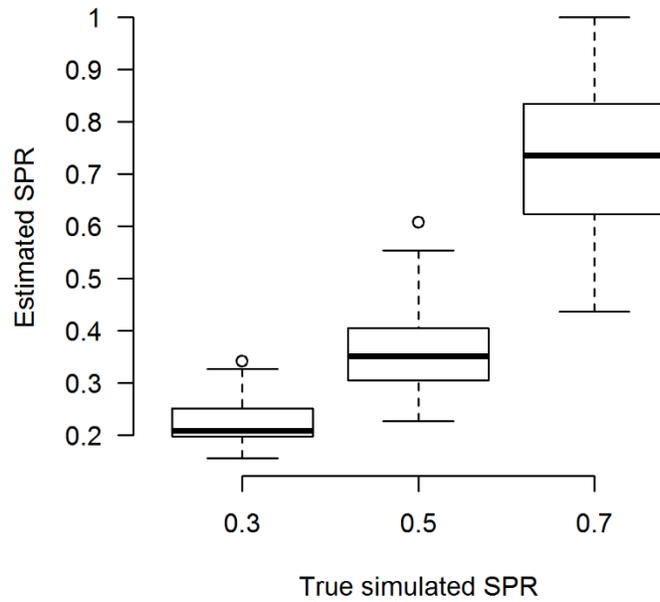


Figure D1. Comparison of “true” simulated SPR to estimates obtained from the LB-SPR fitting approach under steady-state or equilibrium conditions.

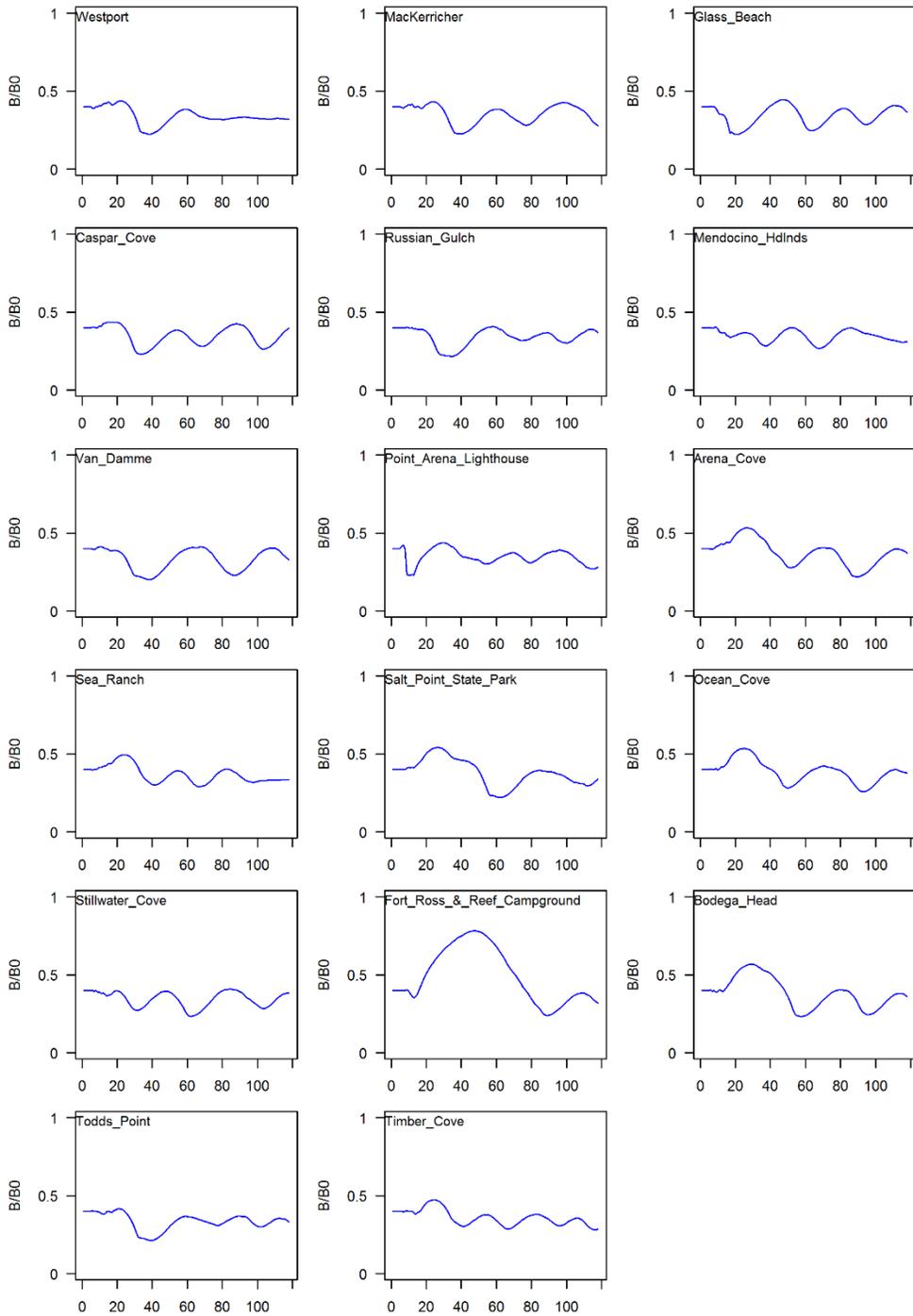


Figure D2. Depletion trends (spawning  $B / B_0$ ) based on deterministic projections using only the LB-SPR indicator in the decision tree.

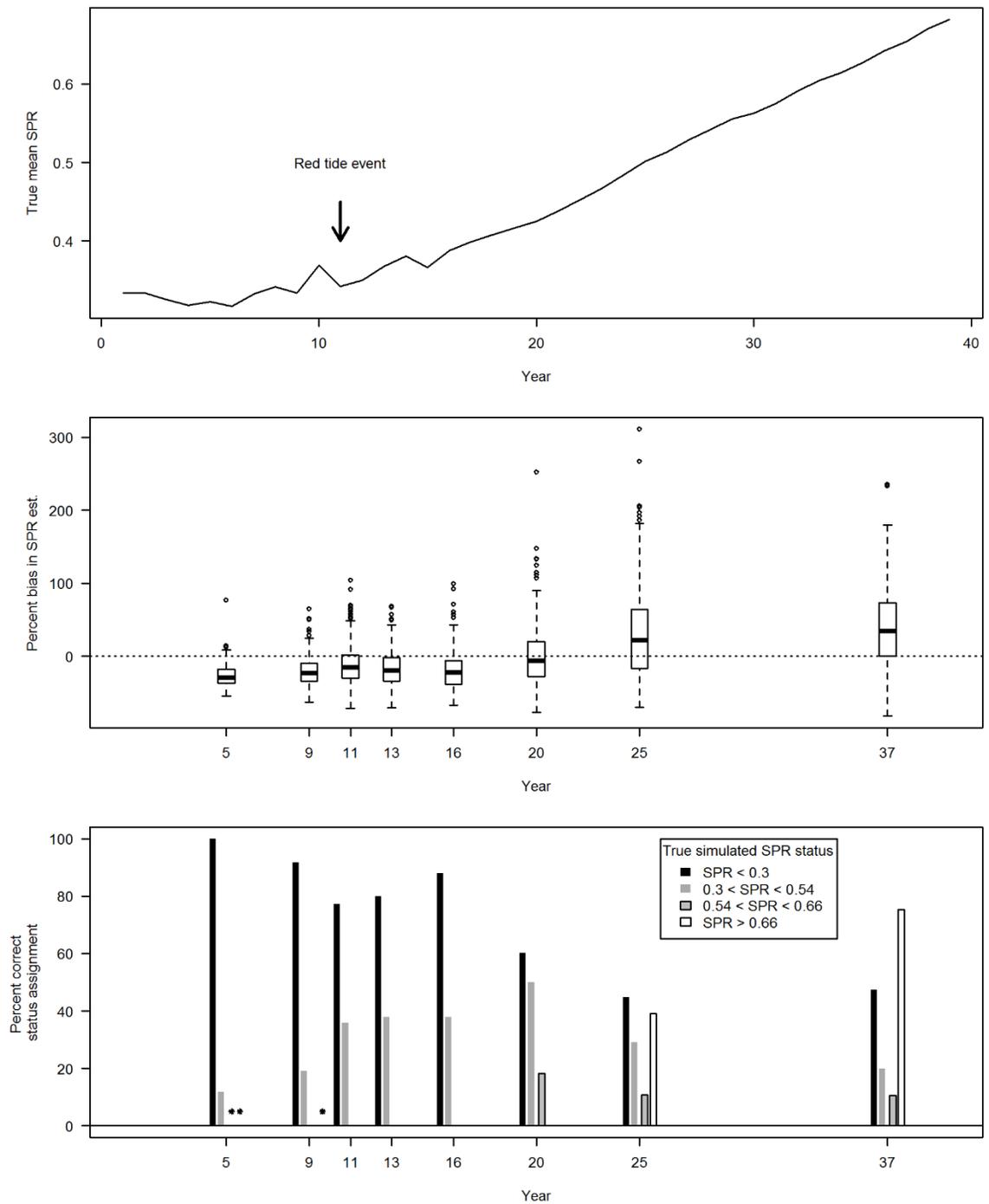


Figure D3. Summary of SPR estimation reliability under historically low but stable abundance. Upper panel is average SPR trend in 100 simulation runs, middle panel is percent bias boxplots in select years of simulation runs, and lower panel is the percent of instances of SPR estimates being correctly assigned to a status category. Asterisks indicate no true simulated instances of a status category.

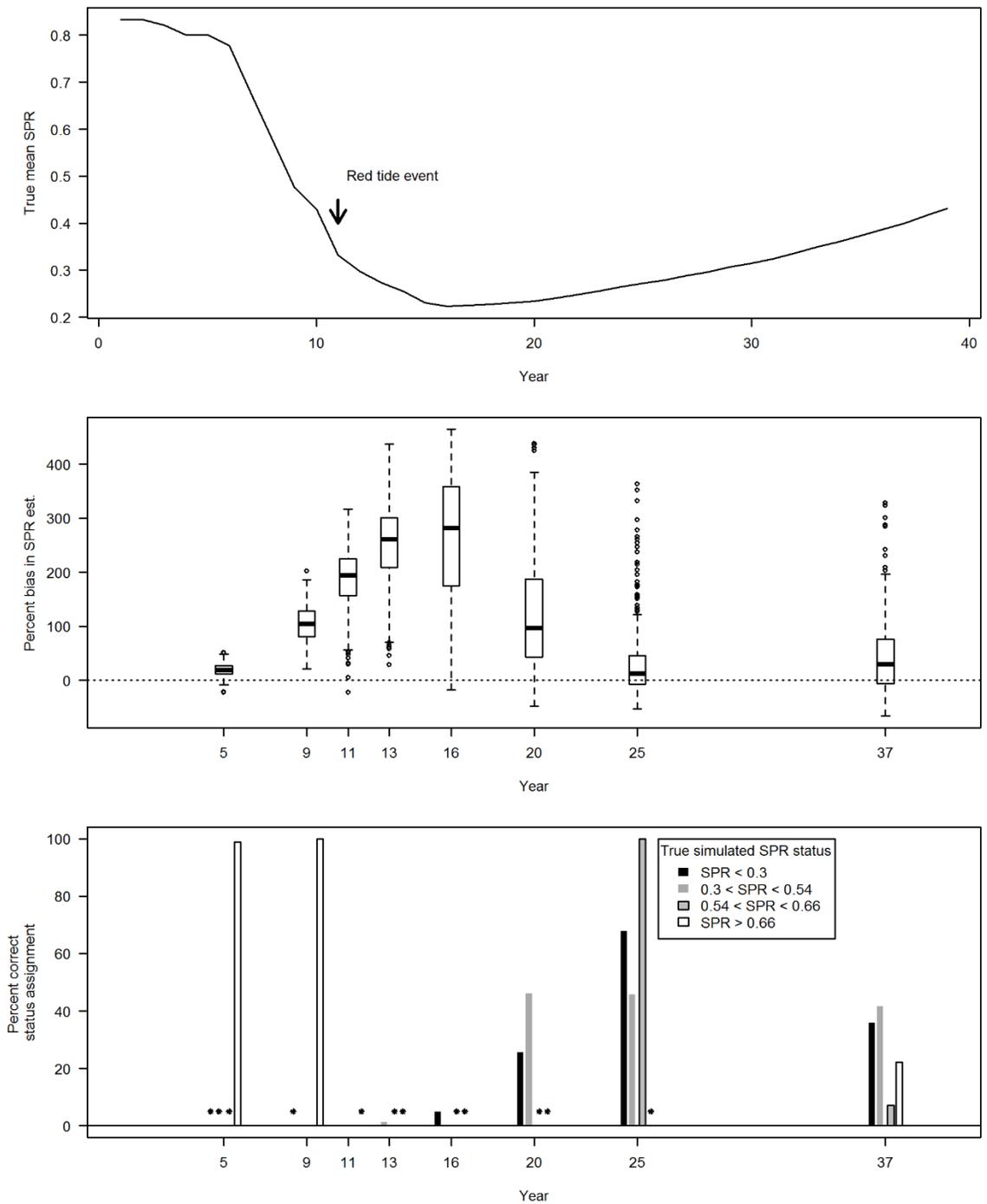


Figure D4. Summary of SPR estimation reliability under historically declining abundance. Upper panel is average SPR trend in 100 simulation runs, middle panel is percent bias boxplots in select years of simulation runs, and lower panel is the percent of instances of SPR estimates being correctly assigned to a status category. Asterisks indicate no true simulated instances of a status category.

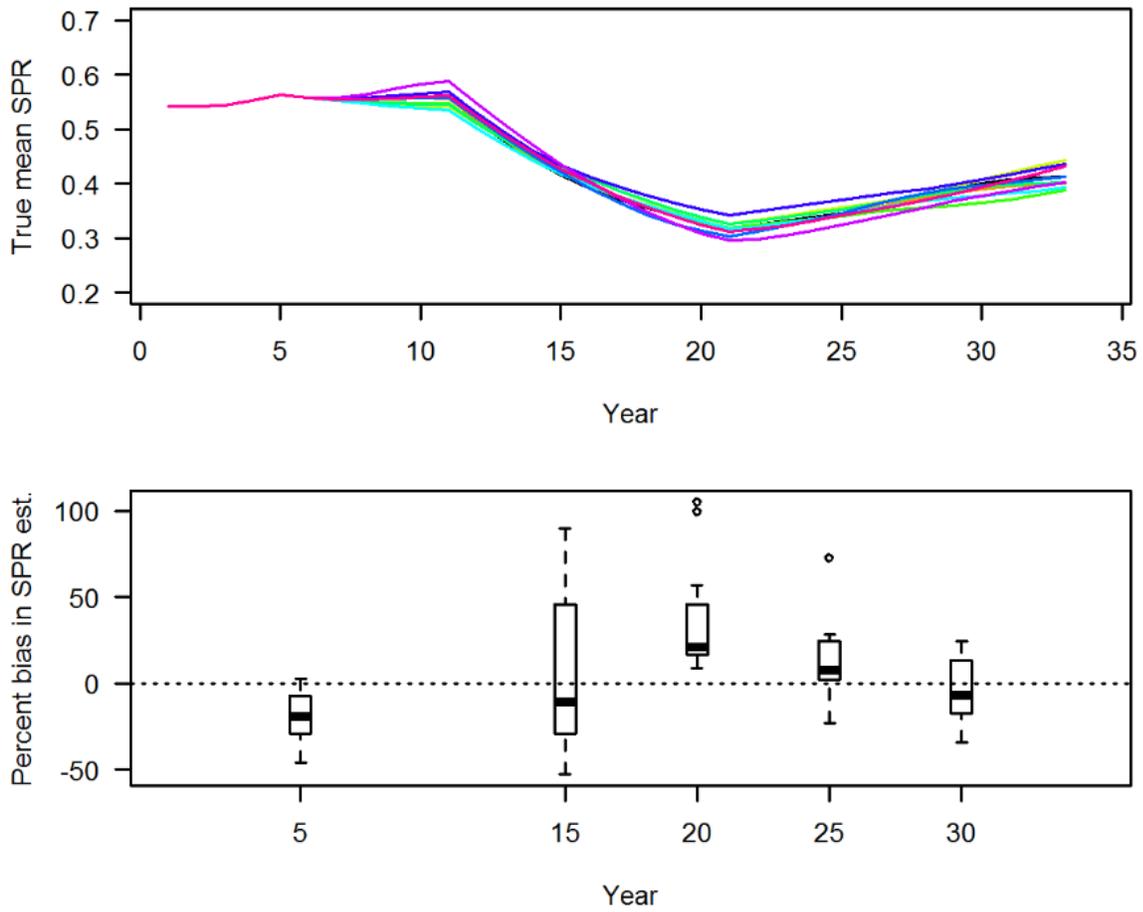


Figure D5. Upper panel shows 10 simulated stock trends that reflect increased natural mortality between years 11 and 20. The lower panel shows the corresponding bias in SPR estimation via the LB-SPR fitting method at years 5, 10, 20, 25, and 30.

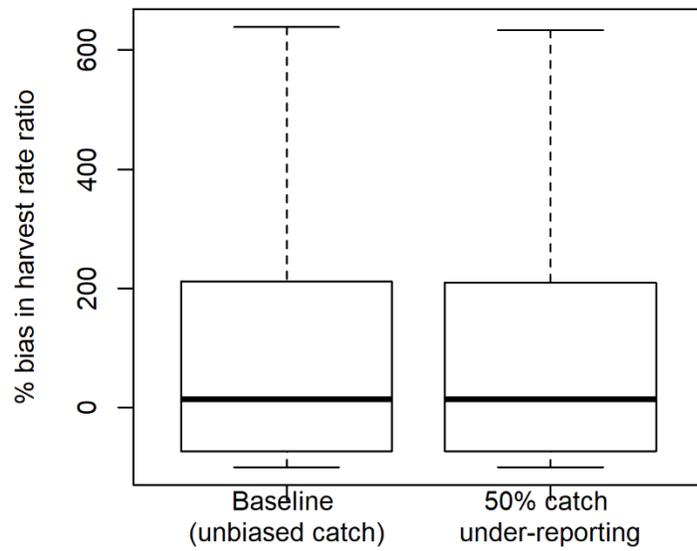


Figure D6. Percent bias in estimated harvest rate ratio (via catch-MSY) relative to the “true simulated” harvest rate ratio. Shown are two scenarios, unbiased catch and 50% under-reporting to represent poaching.

## Appendix E. Sensitivity runs

Through feedback from scientists, stakeholders, and decision-makers, several poignant concerns were raised about the candidate management strategies and about the technical aspects of conducting MSE. The aim of the sensitivity analysis was to implement the MSE under alternative modeling assumptions. These sensitivity runs were each implemented utilizing or modifying (as appropriate) the faster rebuild HCR under the operating model combination of the *historical natural mortality baseline* and typical future ENSO anomalies. Sensitivity runs were:

- **Separate performance contribution of each status indicator.** Aimed at understanding how each status indicator (catch-based or length-based) contributes to performance. Results suggested that the SPR ratio – aimed at guiding the stock towards  $B_{MSY}$  – did effectively produce catches near MSY. This decision tree variant produced biomass levels that, on average, were close to  $B_{MSY}$ , but accordingly, an increased risk of the stock being below  $B_{MSY}$  is also evident (Fig. E1). The harvest rate ratio, calculated from catch histories, produced more conservative catches when compared to the SPR approach and resulted in higher average biomass. As we have noted in the main text, the harvest rate ratio tended to err on the side of cautionary catch reductions in instances where estimated quantities deviated from “true simulated” values.
- **Monitoring length frequency at all sites.** All 56 sites were monitored to provide a conceptual baseline for performance improvement relative to current sampling of 15 sites. Results suggested that relatively minor performance gains could be obtained by improving spatial coverage of length frequency sampling (Fig. E2). Our simulations did not impose systematic spatial gradients in life history that might persist, for example, systematic growth

differences along latitudinal gradients. Thus, it is still advisable to ensure that sampling coverage is widely dispersed along the entire coastline.

- **Minimum harvest length.** Minimum harvest length was changed from seven to eight inches. Because changing minimum harvest length immediately modifies size availability of red abalone to fishers, we must view the effects of a size limit change in terms of its short-term effects and its longer-term effects. Accordingly, the use of transition dynamics illuminates how size limit changes affect catches, fishing mortality, abundance, and SPR status of the stock. Results demonstrate that a size limit increase will immediately (in the short-term) reduce the availability of abundance to be caught by the fishery because 7" to 8" abalone are no longer available. Thus, continued removal of the TAC requires fishing mortality to increase on the remaining part of the stock > 8 inches (Fig. E3). Because fishing mortality is inadvertently increased in the short-term, rebuilding (i.e., increase in SPR) is delayed. In the longer-term, we see increases in catches greater than 10 inches (trophy size) because 7- 8" red abalone live longer (i.e., are subject to lower total mortality rate), thus, more red abalone persist to larger sizes (in the longer term). There are a few caveats to this sensitivity run. First, the higher fishing mortality that occurred in the short-term to achieve TACs could be reduced by simultaneously introducing a TAC reduction. This approach would lead to a more rapid increase in SPR and lower overall fishing mortality rate. Second, the 25-year simulations do not reflect the stock becoming stable at a new steady state condition under this change from 178 mm to 203 mm minimum harvest length. Raising the size limit to eight inches (203 mm) would likely lead to higher red abalone catches once the size structure of the stock becomes stabilized. Third, increasing the minimum harvest length may reduce harvest in ways not included in this sensitivity run. For example, it is possible that overall

catches will decrease because it will be harder for fishermen to catch the less available and less accessible larger red abalone. Increasing the minimum harvest length for red abalone may disproportionately affect a facet of the fishery known as rock picking or shore-picking, which commonly capture the smallest legal size red abalone (~7 inch (178 mm)). This facet of the recreational fishery could potentially be excluded or significantly reduced upon any increase made to the minimum harvest size. At the same time, this increase in minimum harvest length could result in lower actual catches and reduce un-intentional mortality caused by bar cuts.

- **Systematic bias (underreporting) of catches.** In this sensitivity run, total catches were commensurate with those used in baseline model runs, but the reporting of catches (i.e., simulated observation of catches) was reduced by 50%. Thus, true catches were twice as high as the reported catches. Results suggested that MSE performance outcomes were nearly identical between the instance where total catches were accurately reported and those where the same catch levels were systematically under-reported by 50% (Fig. E4). The reason for similar performance reflects the way in which scaling of vulnerable abundance occurs in the Schaefer surplus production model, which is used in the catch-MSY method. The  $B_0$  parameter is scaled according to the catch history input. When under-reporting occurs (by the same percentage each year) a lower  $B_0$  parameter is estimated by the model than that which would have been obtained had “true” catches been used as inputs. Because harvest rate is a ratio of catches to current stock size, we still obtain reasonable estimates of this ratio, even during under-reporting. Readers should be cautioned that alternate approaches to simulating poaching patterns thru time can be more complex than those that we simulated and can affect estimator reliability in a manner that is different than we reported here. In general, we note

that poaching remains a major concern for the red abalone fishery. Because harvest control rules only regulate legal catches, poaching can only be controlled through enforcement and no viable harvest control rule should be expected to prevent fishery collapse in the face of high poaching, not our decision-tree nor any density-based harvest control rule. Only through the combined strategies of effective fishery regulation and diminished poaching via enforcement can we expect to ensure fishery sustainability.

- **Systematic bias in LB-SPR input parameters.** Inputs to LB-SPR of  $M/K$ ,  $L50/L\infty$ , and  $L50$  were separately varied by  $\pm 25\%$ . In each instance, only one parameter was biased, while the others were held at their constant values ( $M/K$ ,  $L50/L\infty$ ) or were estimated from simulated observations ( $L50$ ) during MSE. This sensitivity run was carried out using the two-indicator decision tree as well as the single-indicator length-based decision tree. Positive bias in  $M/K$  ratio had the most negative effect on producing sustainable biomass, while negative bias in the  $L50/Linf$  ratio and positive bias in  $L50$  had negative effects on obtaining high catches (Fig. E5).

Historical natural mortality baseline  
ENSO-driven life history variation

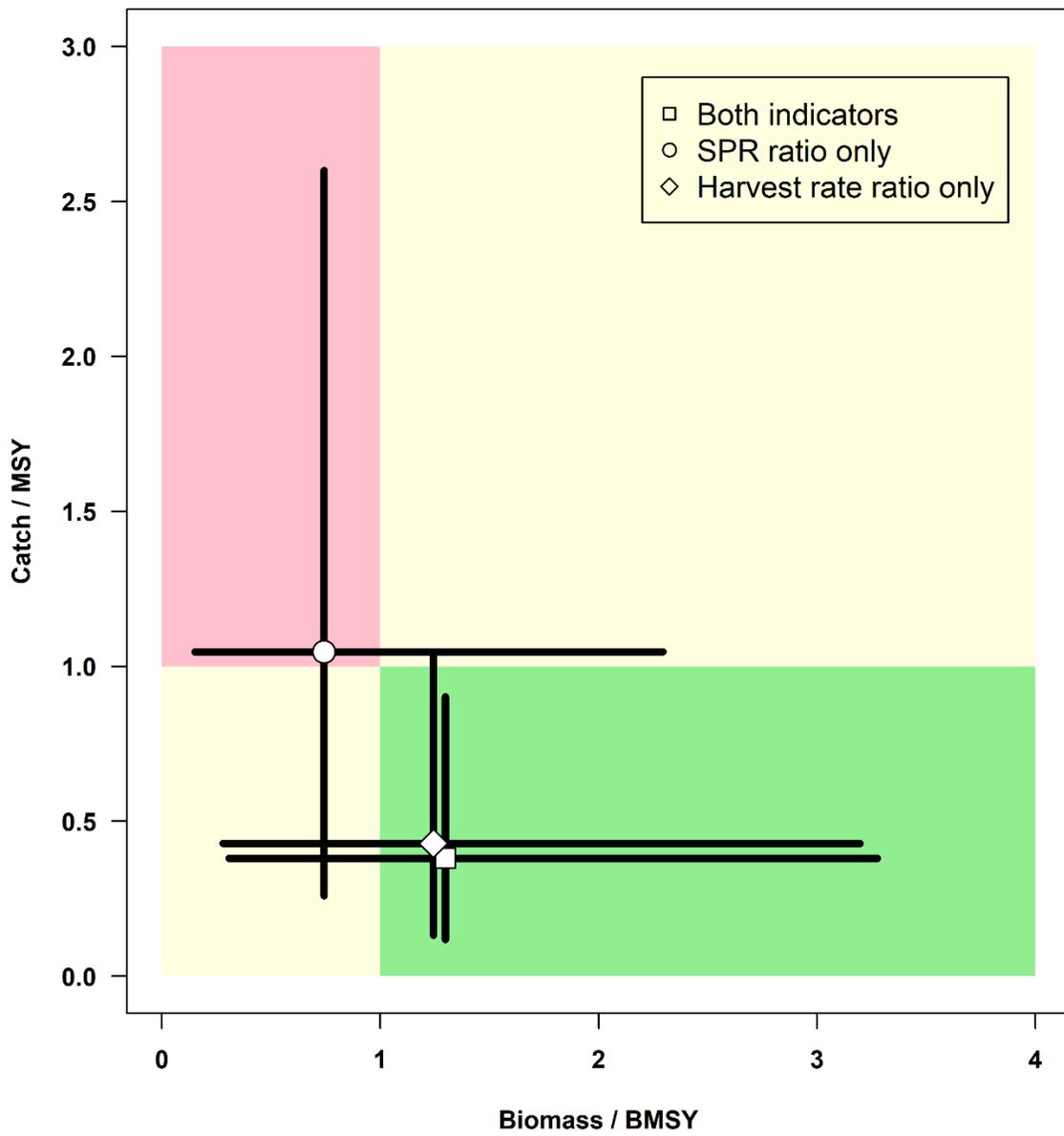


Figure E1. Performance at end of year 25 of forecasts for sensitivity runs that included the use of only a single status indicator (SPR ratio or harvest rate ratio).

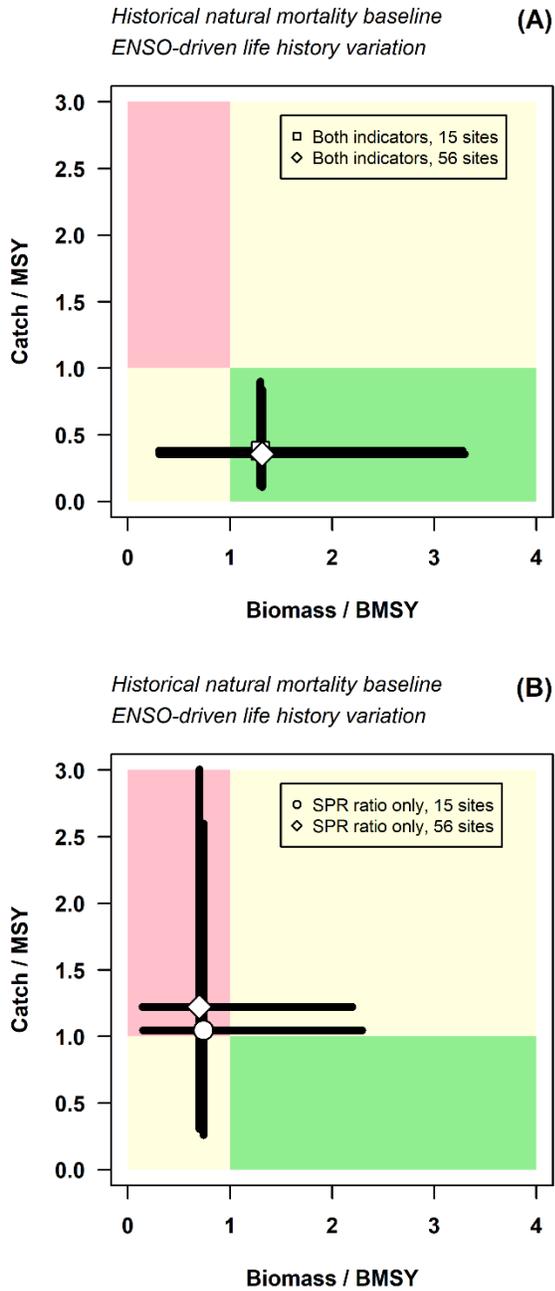


Figure E2. Performance at end of year 25 of forecasts for sensitivity runs that included length frequency sampling at all 56 sites. (A) is performance of fast rebuild decision tree, and (B) is same decision tree except with only the SPR ratio (no harvest ratio included).

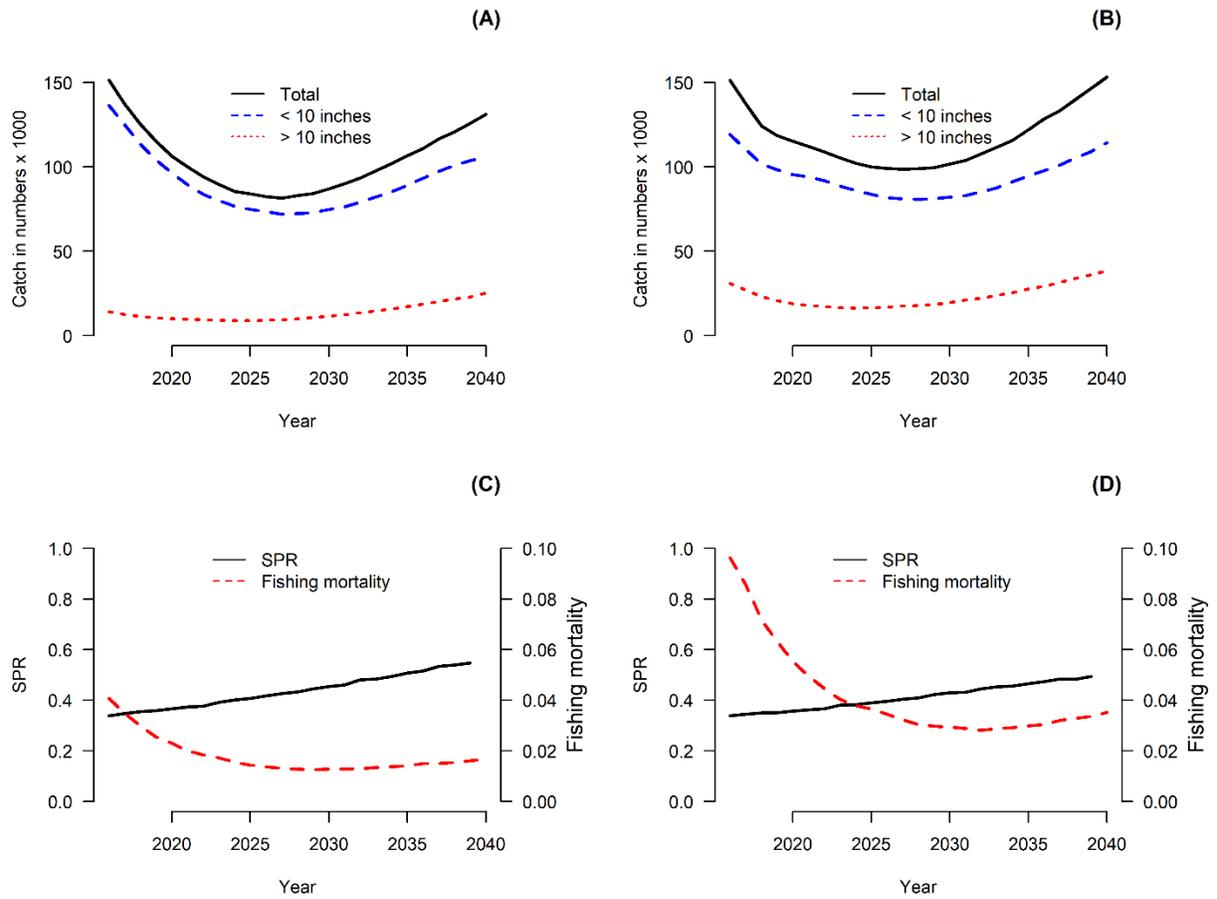


Figure E3. Transition dynamics for sensitivity runs that included changing minimum harvest size from 178 mm (A & C) to 203 mm (B & D). Median trend lines across 250 simulation runs are shown.

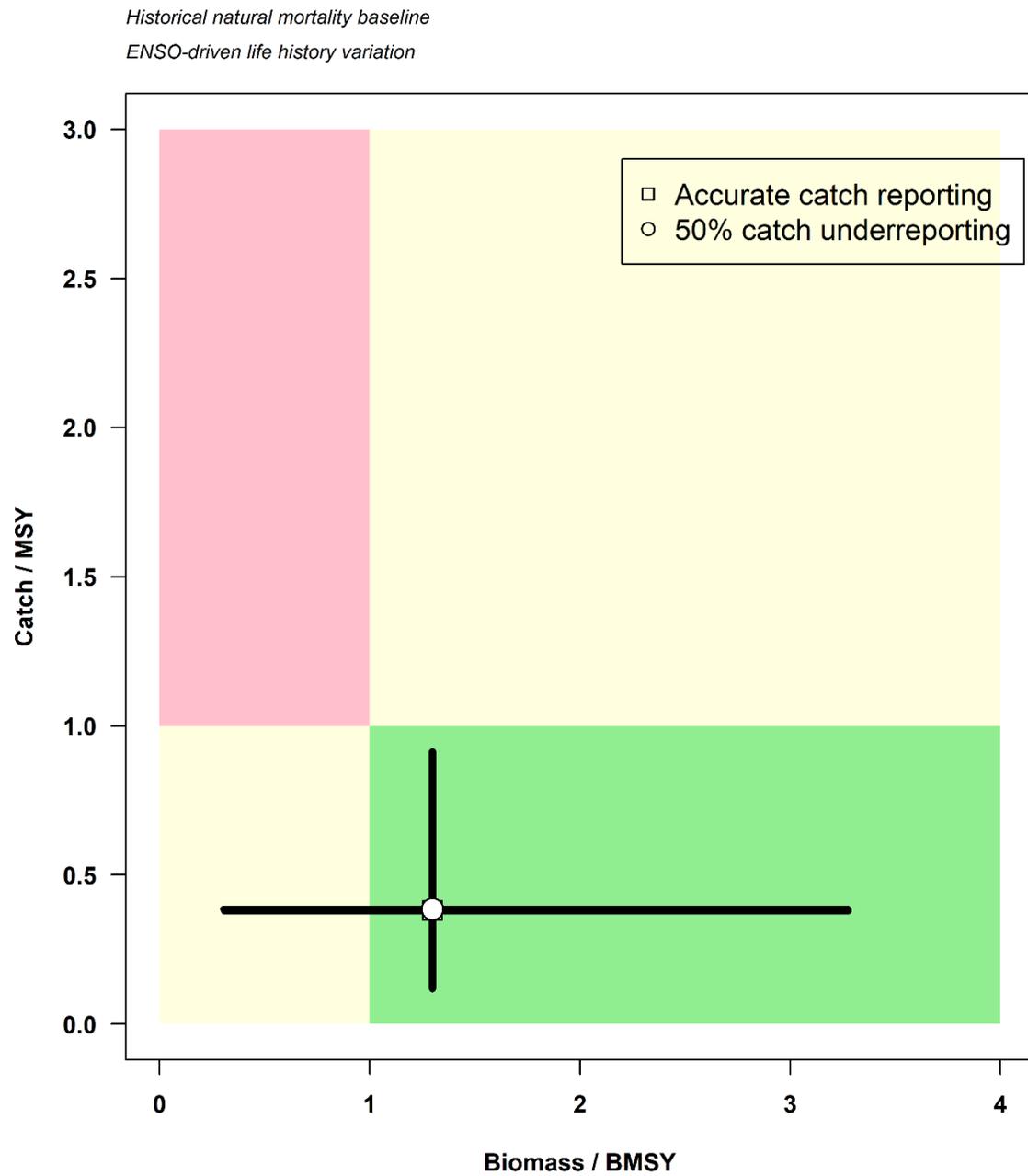


Figure E4. Performance at end of year 25 of forecasts for sensitivity runs that included under-reporting of catches by 50%.

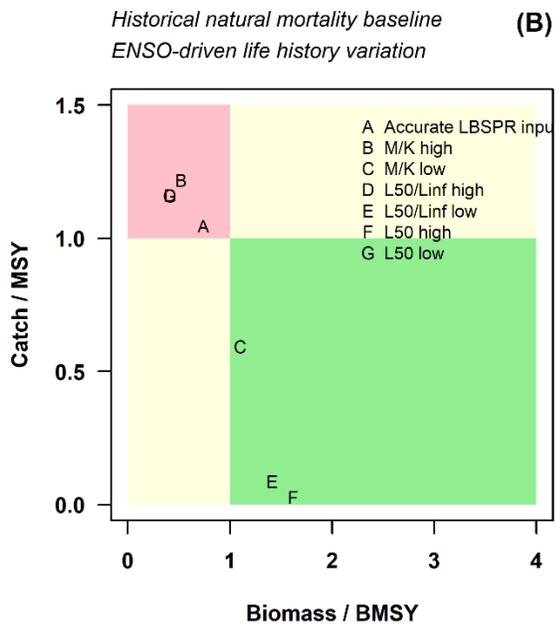
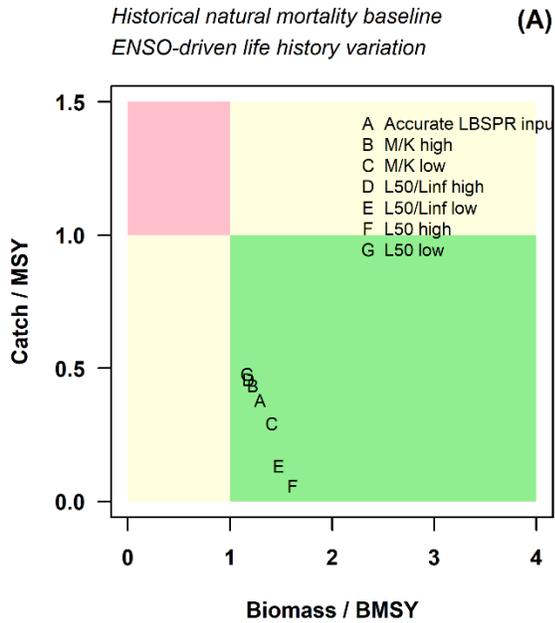


Figure E5. Performance at end of year 25 of forecasts for sensitivity runs that included bias in LB-SPR input parameters. (A) is performance for the fast rebuild decision tree that included both LB-SPR and catch-MSY indicators; (B) is performance for the fast rebuild decision tree that uses only a single indicator, LB-SPR.