A Status Review of Pinto Abalone (Haliotis kamtschatkana) Along the West Coastof North America: Interpreting Trends, Addressing Uncertainty, and Assessing Risk for a Wide-Ranging Marine Invertebrate

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A STATUS REVIEW OF PINTO ABALONE (*HALIOTIS KAMTSCHATKANA*) ALONG THE WEST COAST OF NORTH AMERICA: INTERPRETING TRENDS, ADDRESSING UNCERTAINTY, AND ASSESSING RISK FOR A WIDE-RANGING MARINE INVERTEBRATE

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ABSTRACT Pinto abalone (*Haliotis kamtschatkana*), the widest ranging abalone species in North America, occurs from Alaska, United States to Central Baja California, Mexico. The species has been observed in intertidal and subtidal habitats from 0 to 40 m depth. The best available data indicate that pinto abalone abundance has declined in many areas throughout the species' range due to fisheries harvest. Subsistence and personal use fisheries in Alaska and a commercial fishery in Mexico persist. Preliminary data from 2008 to 2016 indicate signs of recovery for some pinto abalone populations along the British Columbia coast due to multiple contributing factors including a reduction in illegal harvest, natural recovery following fishery closure, and low predation pressure. By contrast, pinto abalone populations at the San Juan Islands in Washington are experiencing recruitment failure and continuing to decline, despite closure of the fisheries and no evidence of poaching. Throughout the remainder of the species' range, trends are less clear, due to the lack of regular, long-term monitoring surveys for pinto abalone. The limited data from surveys and/or opportunistic sightings indicate that pinto abalone populations are small, patchily distributed, and/or fluctuate episodically in Alaska, California, and Mexico, with evidence of recent recruitment in a number of locations within these three areas. Baseline abundance and trend data for the species before the advent of commercial fisheries and, in some areas, the local extirpation of sea otters is lacking. Without a clear baseline with which to compare the current abundance levels and trend information, it is difficult to interpret what these levels mean for the status and viability of the species. Threats to pinto abalone were evaluated and characterized using a qualitative rating (i.e., low, moderate, high, very high) based on the threats' scope, severity, and persistence and the sufficiency of the data to support the rating. Several threats that posed a moderate level of risk to pinto abalone were identified including the following: low densities as a result of historical overfishing; the potential threat posed by ocean acidification; and illegal take because of poaching and inadequate law enforcement. The overall risk that pinto abalone face throughout their range was evaluated, and it was determined that they have a low to moderate level of extinction risk now and in the foreseeable future (over both the 30-y and 100-y time horizons). There is a high level of uncertainty regarding demographic factors, in particular regarding whether abundance and productivity levels are sufficient to support the persistence and recovery of the species in the face of continuing and potential future threats. Although recruitment failure may be occurring in some areas (e.g., San Juan Islands Archipelago), in other areas throughout the range recurring and/or recent recruitment events have been observed, despite low densities, and have even resulted in increased densities (across all size classes) at several index sites in British Columbia. Limitations in using demographic data to guide conservation actions and help ensure species persistence could be overcome by conducting consistent monitoring of pinto abalone populations throughout their range.

KEY WORDS: *Haliotis kamtschatkana*, status review, demography, abundance trends, threats, extinction risk

INTRODUCTION

The pinto abalone (*Haliotis kamtschatkana* Jonas, 1845) is a prosobranch gastropod mollusc ranging from approximately Salisbury Sound (north of Sitka), Alaska, to Bahia Tortugas, Baja California, Mexico (Fig. 1) and occupies intertidal and subtidal kelp forest habitats from 0 to 40 m depth. This species is held in high economic, cultural, and ecological regard by both indigenous and nonindigenous peoples along the West Coast of North America. Concern regarding declines of pinto abalone populations in some areas led to the decision to add pinto abalone to the National Marine Fisheries Service (NMFS) ''species of concern'' list, a list of species whose status is

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Figure 1. Geographic distribution of pinto abalone (*Haliotis kamtschatkana*) along the West Coast of North America from Salisbury Sound, AK to Bahia Tortugas, Baja California, Mexico.

alarming, but for which limited information is available (69 FR 19975). In 2013, NMFS received two petitions to list pinto abalone as federally endangered in the U.S. under the Endangered Species Act (ESA) because of historical and ongoing declines in their abundance. As a result, an effort was made to compile and evaluate the best available information and data on pinto abalone demography, abundance trends, and threats to assess the species' risk of extinction throughout its range. Specifically, the following were examined: (1) historical and current range, distribution, and habitat use of the species; (2) long-term trends in abundance throughout the species' range; (3) historical and current estimates of population size and available habitat; (4) knowledge of various life history parameters (size/age at maturity, fecundity, length of larval stage, larval dispersal dynamics, etc.); (5) potential short- and long-term risk factors that the species faces throughout its range (e.g., overharvesting, natural predation, disease, habitat loss, etc.); and (6) overall level of extinction risk based on threats and demographic risk factors over 30- and 100-y time frames.

TAXONOMY, MORPHOLOGY, AND GENETIC SPECIES IDENTIFICATION

The pinto abalone has been taxonomically subdivided into two subspecies: *Haliotis kamtschatkana kamtschatkana* ranging from Sitka, AK to Point Conception, CA; and *Haliotis kamtschatkana assimilis* ranging from Monterey, CA to Bahia Tortugas, Baja California, Mexico (Fig. 1; McLean 1966). Initially, these subspecies were described as separate species by Jonas (*Haliotis kamtschatkana*) in 1845 and Dall (*Haliotis assimilis*) in 1878. McLean (1966) argued that the two previously described species were unique forms, or subspecies, that represented geographic extremes of a single species. According to McLean (1966), the only morphological characteristics that distinguish these subspecies are shell shape and pattern. The multicolored shell of *H. k. kamtschatkana* attains a maximum length of approximately 160 mm and has three to six open respiratory pores that are raised, oval-shaped, and medium-sized. The outer surface of the shell is characterized by irregular lumps. Paralleling the respiratory pores is a deep groove (Stevick 2010). Typically, the interior of the shell is pearly white with hints of multicolored iridescence and no muscle scar (COSEWIC 2009). The shell of *H. k. assimilis* is also multicolored, but is rounder and more convex than that of *H. k. kamtschatkana*, has four to seven open respiratory pores that are somewhat elevated, and attains a maximum shell length of approximately 190 mm. For both *H. k. kamtschatkana* and *H. k. assimilis*, the epipodium (the circular fringe of skin around the foot) and tentacles are mottled yellow to dark tan with vertical banding patterns and a lacy edge. The muscular foot is tan and is used to adhere to hard substrate and for locomotion.

McLean (1966) suggested that the differences observed in shell morphology between these subspecies might be related to varying environmental conditions, possibly water temperature, along a latitudinal gradient encompassing the species' range from Alaska to Mexico, with a restricted merging zone from Monterey to Point Conception in Central California (Fig. 1). Futuyma (1986) defined subspecies as allopatric populations (i.e., populations occurring in different geographic areas or in isolation from one another) with a fixed character that does not yet represent discrete, evolutionary lineages and could interbreed at the periphery of populations. This information supported the idea that subspecies do not have to be defined by a distinct geographic boundary but instead can merge at the limits of their ranges, in this case Monterey to Point Conception. Geiger (1999) upheld the subspecies classification scheme based on the morphological descriptions of shells provided by McLean (1966) and maintained the subspecies range descriptions as Sitka, AK to Point Conception, CA, for *Haliotis kamtschatkana kamtschatkana*, and Monterey, CA to Bahia Tortugas, Baja California, Mexico, for *Haliotis kamtschatkana assimilis*. Counter to the description of a restricted merging zone offered by McLean (1966), Owen and Raffety (2017) present evidence suggesting a broader merging zone: (1) the northern shell morphology, although more frequently encountered in northern portions of the species' range, is represented in shell collections from Alaska to Baja California, Mexico (Fig. 2); and (2) the southern shell morphology, although most commonly encountered south of Point Conception, is represented in shell collections from as far north as Monterey.

Recently, two lines of evidence have raised uncertainty regarding the subspecies classification. Genetic tools have been developed to confirm whether there are genetic bases for differences among abalone species and among populations within species. To date, none of these tools (nuclear and mitochondrial single-gene sequencing and microsatellite genotyping) have been able to identify genetic differences between *Haliotis kamtschatkana kamtschatkana* and *Haliotis kamtschatkana assimilis* (Gruenthal & Burton 2005, COSEWIC 2009, Supernault et al. 2010). With newer methods (e.g., restriction site–associated DNA sequencing or RADseq, genotyping-in-thousands by sequencing or GTseq) that develop or genotype markers, such as single nucleotide polymorphisms (SNPs), scattered throughout the entire genome, geneticists may discover genetic support for the subspecies delineation in the future.

One highly conserved portion of the genome that has been investigated, and that geneticists would have expected to be different between subspecies, is the area that controls the production of the reproductive proteins lysin and vitelline envelope receptor for lysin (VERL). Supernault et al. (2010) developed high fidelity size and sequence polymorphic markers in the reproductive proteins lysin and VERL for forensic analyses of northeastern Pacific abalone species. Lysin (in sperm) and VERL (on the egg) are cognate pairs of gamete recognition proteins that coevolved to mediate fertilization, and they are species-specific, making them ideal targets for species identification (Galindo et al. 2002). Results indicated that all species recognized on the basis of morphological differences have been confirmed to be distinct on the basis of genetic sequences, with only the two subspecies, *Haliotis kamtschatkana kamtschatkana* and *Haliotis kamtschatkana assimilis*, indistinguishable through molecular analysis. Gruenthal and Burton (2005) had similar results, concluding *H. k. kamtschatkana* and *H. k. assimilis* were statistically indistinguishable at sequenced portions of the mitochondrial genes of cytochrome oxidase subunit one (CO1) and cytochrome b, as well as VERL, although the sample sizes were small. Straus (2010) also found no statistically significant differences in either CO1 or lysin, stating that the two subspecies share identical sequences at both

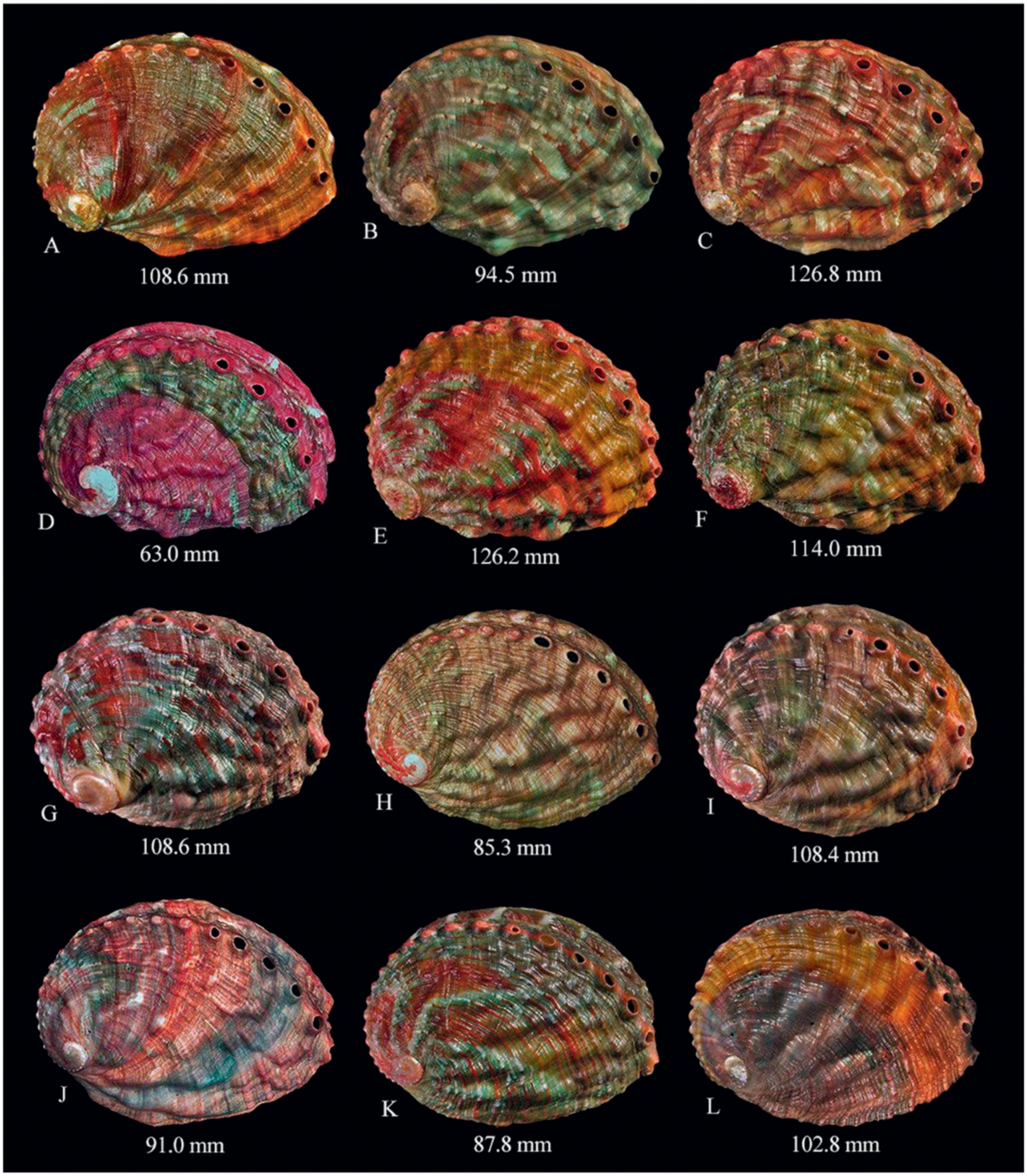

Figure 2. Examples of pinto abalone shells (63–126 mm) exhibiting the northern shell morphology from throughout the species range: (A) Alaska; (B andC) British Columbia, Canada; (D–F) northern California, United States; (G–L) Baja California, Mexico. Adapted from Owen and Raffety (2017).

mitochondrial and nuclear loci and cannot be differentiated. Most recently, Schwenke and Park (unpublished data) constructed bootstrapped neighbor-joining trees of new and archived mitochondrial CO1 and VERL sequences, finding that VERL is currently the best marker available to resolve the most closely related abalone species found along the northeastern Pacific coast (white, pinto, flat, and red), whereas CO1 separates this group from the remaining species (i.e., black, pink, and green; Schwenke personal communication). Again, however, neither marker provided subspecies-level resolution. To specifically address the subspecies issue, an expanded representation across the *Haliotis kamtschatkana* species range of genomic variation using SNPs is planned, with a particular focus on the region of overlap between the two putative subspecies.

In addition to the fact that the two subspecies remain indistinguishable at the molecular level to date, Owen and Raffety (2017) present evidence that the shell morphology representative of *Haliotis kamtschatkana assimilis* occurs throughout at least 40% of the species' range and the shell morphology representative of *Haliotis kamtschatkana kamtschatkana* occurs throughout 100% of the species' range (Fig. 2). Given this evidence, the range overlap between the two putative subspecies is much more extensive than was previously thought and this degree of overlap does not meet the definition of subspecies as allopatric populations provided by Futuyma (1986).

For the purpose of this review, pinto abalone will be referred to as one species throughout its range. This is a parsimonious decision given the degree of overlap between the subspecies, no evidence to date for species divergence at the molecular level, and the fact that there are other examples of marine gastropods with broad geographic ranges (e.g., ribbed limpet, black turban snail) and/or pronounced morphological plasticity (e.g., periwinkle snails) extending on the order of 1,000s of kilometers (Ricketts et al. 1985). Although it is possible that pinto abalone subspecies exist, without some genetic, geographic, or ecological justification for treating these subspecies as separate species, the status and risks to pinto abalone throughout its range from Alaska to Mexico was examined. From this point forward, the species will be referred to as pinto abalone or *Haliotis kamtschatkana* unless published information specifically calls out one of the recognized subspecies.

DISTRIBUTION, DEPTH RANGE AND HABITAT ASSOCIATIONS

Of the seven species of abalone found along the West Coast of North America (Geiger 1999), pinto abalone have the broadest latitudinal range extending from Salisbury Sound, AK, to Bahia Tortugas, Baja California, Mexico (Fig. 1; Campbell 2000), and it is the predominant abalone found in Washington and Alaska, and in British Columbia, Canada. According to Geiger (2000, http://www.vetigastropoda.com/ABMAP/NEPacific.html), which visually depicts the geographic distributions for *Haliotis kamtschatkana kamtschatkana* and *Haliotis kamtschatkana assimilis* based on specimen and literature records, pinto abalone range from southeast Alaska to approximately Punta Eugenia and possibly Bahia Tortugas, Baja California, Mexico, with an absence of records from Puget Sound proper and from between the Neah Bay, WA to Cape Mendocino, CA.

In the northern portion of its range, pinto abalone occurs in intertidal and subtidal habitats (0–20 m depth, most commonly 0–10 m depth; Rothaus et al. 2008) that vary with respect to exposure and contain hard substrate (bedrock and boulders/cobble) with ample quantities of benthic diatoms, and micro and macro-algae, and are often associated with crustose coralline algae that is thought to serve as a settlement cue (Roberts 2003). Small juvenile (<10 mm) pinto abalone are difficult to find in the field but are occasionally observed under boulders and on smooth bedrock or boulders that are bare or encrusted with coralline algae, mostly at deeper depths (e.g., 5–15 m) than adults are typically found (Breen 1980a). Other grazers (e.g., sea urchins, chitons, limpets, and adult abalone) may be important in maintaining coralline encrusted rock (Sloan & Breen 1988). Typically, the species is found in areas with little freshwater influence (salinity ≥30 parts per 1,000) and can tolerate wide ranges in temperature, from 2°C to 24°C, based on laboratory experiments (COSEWIC 2009).

In Alaska, pinto abalone are primarily found in rocky substrates and kelp beds in the lower intertidal and subtidal surge zones on the outer coast of southeast Alaska (Fig. 3). They are also found in the Inside Passage of southern Southeast Alaska. The Alaska Department of Fish and Game (ADF&G) has documented pinto abalone occurrence as far north as Salisbury Sound, north of Sitka (ADF&G public comments to NMFS, January 17, 2014).

In Washington, pinto abalone occur from Little Patos Island in the northern San Juan Islands Archipelago (SJA) to just offshore of Cape Flattery in the west entrance of the Strait of Juan de Fuca (SJF) and north of a line formed by Point Wilson and the Keystone Jetty in North Puget Sound (Fig. 3). Pinto abalone likely occur outside this range, but observations were not reported from credible sources. Vertical depth distribution in Washington is 3–20 m depth relative to mean lower low water. No comprehensive surveys of pinto abalone habitat have been conducted to determine the entire range of vertical distribution of pinto abalone in Washington. No pinto abalone were observed during a 2010 Washington Department of Fish and Wildlife (WDFW) remotely operated vehicle survey in SJA at 165 locations ranging from 5 to 300 m (WDFW unpublished data).

Very little is known about the occurrence of pinto abalone along the Oregon coast. There were no specimens available from this region at the time of the Geiger (2000) review of the distribution and biogeography of this species. From 2009 to present, a small number of isolated pinto abalone have been reported on Orford Reef (Fig. 3; the biggest offshore rocky reef in Oregon; Groth unpublished data).

In the southern portion of the range, pinto abalone occur in subtidal habitats (approximately 12–40 m; Geiger & Owen 2012) commonly on open-rock surfaces (Fig. 3). The animals appear to be patchily distributed in areas along the southern California mainland, and distribution may be correlated with substrate type (flat rock preferred over uneven), relief (low relief with scattered rock and boulders preferred over high relief), the presence of intermittent sand channels that may accumulate drift kelp, and algal composition in that *Pelagophycus porra*, *Laminaria farlowii*, *Agarum fimbriatum*, *Pterygophora californica*, and coralline algae (articulated and crustose) are often present in areas where pinto abalone occur (Bird unpublished data, Hagey et al. unpublished data, Neuman et al. unpublished data). In Mexico (Fig. 4), a recent study reported that *Haliotis kamtschatkana assimilis* and *Haliotis sorenseni* occurred at depths ranging from 11 to 25 m (Boch et al. 2014). Most abalone were found between 13–15 m and 19–21 m, but this may reflect a bias toward the depths that were most visited (Boch et al. 2014).

POPULATION STRUCTURE

Withler et al. (2001) provide the only published assessment of population structure in *Haliotis kamtschatkana*. Genetic

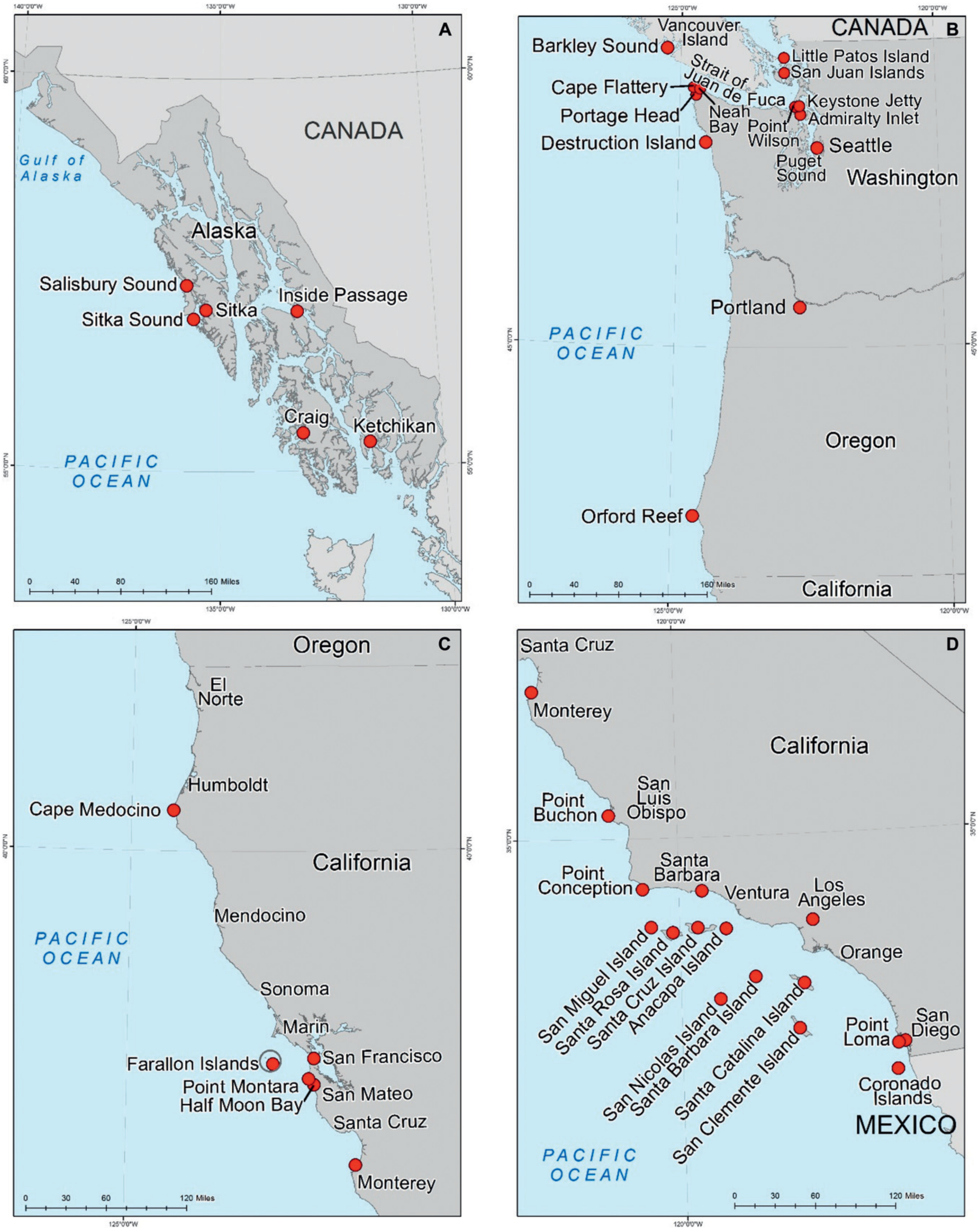

Figure 3. Geographic distribution of and points of reference for pinto abalone (*Haliotis kamtschatkana*) in: (A) Alaska; (B) Washington and Oregon; (C) Northern California; and (D) Southern California.

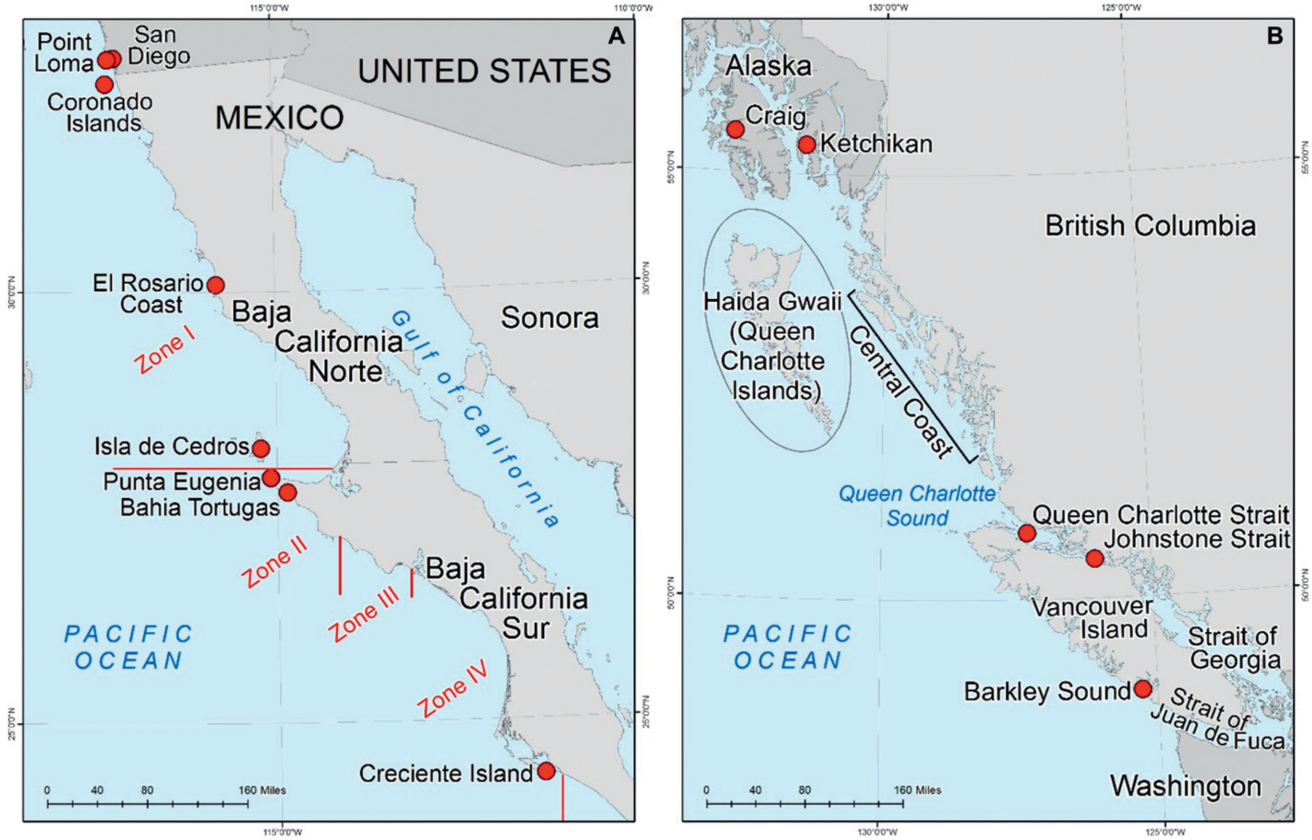

Figure 4. (A) Geographic distribution of and points of reference for pinto abalone (*Haliotis kamtschatkana)* in Baja California, Mexico. Four abalone fishery management zones (I–IV) are delineated by red lines and encompass 22 cooperatives. Adapted from Sierra-Rodrigues et al. 2006; (B) Geographic distribution of and points of reference for pinto abalone (*Haliotis kamtschatkana)* in British Columbia, Canada.

variability at 12 microsatellite loci was estimated for a total of 18 sites located throughout coastal British Columbia and at one site in Sitka Sound, AK. Only 0.2% of variation was attributable to differences between a grouping of Haida Gwaii (formerly Queen Charlotte Islands; Fig. 4) with Sitka and the remaining sites, and another 0.2% of variation between Haida Gwaii and Sitka. Overall, 99.6% of variation was within populations, indicating a lack of differentiation among the sampled sites. In addition, there was only weak evidence for isolation by distance. These results suggest gene flow among populations within this region, with little restriction in dispersal. Research on populations throughout the remainder of the species range or using other genetic marker types has not been performed to date. Future work will focus on the use of new genetic markers (SNPs) that include variation in known genes of functional importance to pinto abalone (Timmins-Schiffman et al. 2013). One goal of these studies is to determine the level of genetic differentiation that exists within and between populations in British Columbia to ascertain whether the spatial scale of the current management zones is appropriate for achieving recovery and conservation goals (Curtis personal communication).

MOVEMENT

Little is known about movement patterns of larval or juvenile pinto abalone anywhere in their range. Based on laboratory observations, posthatch larvae are phototactic and swim upward, where they are available for transport by water currents (Olsen 1984, Calderwood 1985). The planktonic larval stage is short (approximately 5–6 days; Olsen 1984), and thus dispersal is likely to be limited and determined primarily by patterns of water movement in nearshore habitats near spawning sites. Individual larvae may be able to influence movement to some degree by adjusting vertical position in the water column, but the ability of pinto abalone larvae to move in this way has not been documented.

There is no published information on direct observations of movement behavior of small (<20 mm) juvenile pinto abalone in the field. Distribution patterns of juveniles and adults indicate an ontogenetic shift in habitat use, with small juveniles [<10 mm shell length (SL)] occupying highly cryptic habitats in deeper waters and migrating to shallower depths and more exposed habitats as they increase in size (Sloan & Breen 1988). In surveys in British Columbia, the proportion of exposed abalone increased from 60% for juveniles 10–70 mm in size to 90% for individuals 70–90 mm in size (Boutillier et al. 1985). Almost all individuals greater than 90 mm in size were found exposed on rock surfaces (Boutillier et al. 1985). This shift may be associated with changes in diet (Sloan & Breen 1988) and predation risk (Griffiths & Gosselin 2008) with size.

Movement generally decreases as individuals grow in size and age. Tagging studies and observational surveys conducted in British Columbia indicate that although adult pinto abalone have the ability to move several meters a day and tens of meters in a year, they typically exhibit minimal movement, likely staying within close proximity to their settlement habitat (Sloan & Breen 1988). Large adults have been observed in home scars (bare rock underneath an individual that matches its shell shape), indicating relatively little movement, particularly if sufficient food (drift algae) is available (Breen 1980a). Pinto abalone can climb up kelp to graze and have been observed to climb back down, rather than just drop off the kelp, when disturbed (Sloan & Breen 1988). A tagging study by Quayle (1971) found lateral movements of less than 50 m in a year and little vertical movement in the 0–10 m depth range. Breen (unpublished data; cited in Sloan & Breen 1988) recorded a maximum movement of approximately 20 m in 1 y in a study of approximately 400 tagged abalone on the west coast of Vancouver Island (Fig. 4). Emmett and Jamieson (1988) recorded a maximum movement of 125 m in 1 y by adults, although they observed little emigration overall from placement sites. In more recent adult aggregation studies along the coast of British Columbia, translocated adult abalone tended to stay within their new habitat rather than migrating out of the aggregation sites (DeFreitas personal communication, Harding personal communication, Lessard personal communication; cited in COSEWIC 2009). Movement rates of 10 pinto abalone, were measured using acoustic telemetry near San Diego, CA (Fig. 6). Although aggregation behavior was not observed, five animals moved estimated maximum distances of 6.3–34.1 m. Thus, pinto abalone may have the ability to decrease the nearest neighbor distances over a few months, thereby increasing the likelihood of reproductive success. There were no apparent spatial or temporal patterns in movement, no relationship between distances moved and proximity to other abalone at the start of the study, nor an effect of tagging on abalone survival (Neuman et al. 2017).

Laboratory and field observations indicate that individuals tend to be more active at night (Sloan & Breen 1988) and during the spawning season (spring through summer months). Breen and Adkins (1980) observed a natural spawning event off the Queen Charlotte Islands (Fig. 4) in July 1979, in which spawners were aggregated and stacked on top of each other and tended to migrate to the highest point available (e.g., boulder tops and up kelp stipes). This behavior was also observed in spawning events in the laboratory (Quayle 1971). The reason for this behavior is unknown, but may serve to increase fertilization rates by aggregating spawners and increasing the amount of time that the broadcast gametes spend in the water column, and thus, increasing the opportunity for fertilization (Sloan & Breen 1988).

DIET

After a short 5–6 day lecithotrophic (feeding on stored yolk) larval phase (Olsen 1984), juveniles settle and immediately begin feeding (Morse 1984, Morse & Morse 1984). Laboratory observations and gut content analyses of hatchery-reared juveniles show that postmetamorphic juveniles graze on minute benthic diatoms, microalgae, and bacteria associated with encrusting coralline algae and rock surfaces (Olsen 1984, Norman-Boudreau et al. 1986). Juveniles may also feed on the crustose coralline algae itself (Garland et al. 1985). These observations are consistent with microhabitats within which small juveniles are found in the wild (smooth or crustose coralline encrusted bedrock and boulders) (Breen 1980a).

Juveniles begin to shift their diet from benthic diatoms, microalgae, and bacteria to macroalgae between 2 and 5 mm SL. By the time juveniles have reached 20 mm SL, the transition to a diet of primarily macroalgae is complete (Won et al. 2010). Adults have been observed to feed directly on attached macroalgae (Sloan & Breen 1988), but drift macroalgae is believed to be the primary food resource (Breen 1980a). Laboratory studies indicate that adults prefer *Macrocystis* and *Nereocystis* but will feed on diatoms and brown, red, and green algae, including *Laminaria*, *Pterygophora*, and *Costaria* (Paul et al. 1977; Breen unpublished data, Gee & Lee unpublished data; cited in Sloan & Breen 1988). Adults avoided *Fucus distichus* and *Agarum cribrosum* (Paul et al. 1977; Gee & Lee unpublished data; cited in Sloan & Breen 1988). Diet composition likely varies by location within the species range, depending on what is available. In British Columbia, drift brown algae makes up an important food resource for pinto abalone (Breen 1980a, Breen 1980b).

REPRODUCTION

Fecundity

Pinto abalone become emergent (i.e., visible to divers) and are generally reproductively mature at a size of about 50 mm SL (about 2–5 y in age), with all abalone mature at a size of about 70 mm SL (Leighton 1959, Ault 1985, Campbell et al. 1992). Size at maturity can vary by location depending on factors such as water temperature and food availability and quality. Fecundity generally increases with age and shell size and may also be affected by food availability, water temperature, and local environmental conditions [California Department of Fish and Wildlife (CDFW); formerly known as the California Department of Fish and Game 2005]. Campbell et al. (2003) estimated that ovaries of individual *Haliotis kamtschatkana* females (101–135 mm SL) may hold from 2.6 to 7.5 million eggs, with a maximum 11.6 million eggs in a 139-mm female. Fewer eggs are likely released per spawning event; however, reports range in the order of 30,000 to 2.3 million eggs released per event, indicating that pinto abalone may be capable of a number of spawning episodes per spawning season, although this has not been documented.

Spawning Period and Length

Quayle (1971) found ripe pinto abalone year round at several sites in British Columbia, with spent gonads documented from April through June, and observed spontaneous spawning of laboratory-held abalone in May. Breen and Adkins (1979) observed ripe gonads and spawning at Haida Gwaii, British Columbia, in mid-July after collecting and replacing wild abalone for a tagging study. Campbell et al. (2003) report a spawning season from April through July, with a few abalone ripe throughout the year, whereas CDFW (2005) reports the spawning season as April through June. More recently, Seamone and Boulding (2011) documented aggregation and spawning in June and July in Barkley Sound, British Columbia. In the laboratory, spawning has been induced from April through December and volitional spawning has been observed from

May through July, with ripe individuals found throughout the year (Bouma personal communication).

Spawning Density

Broadcast-spawning marine invertebrates with separate genders, such as pinto abalone, must spawn in temporal and spatial synchrony to maximize the probability of successful fertilization. Standard population models predict that a reduction in adult density should be associated with a decrease in intraspecific competition, leading to an increase in growth rate, survival, and gamete production. These advantages, however, may be countered by decreases in the rate of successful fertilization, if individuals are sparsely distributed (Levitan 1995, Levitan & Sewell 1998, Gascoigne & Lipcius 2004). Fertilization success is a limiting factor for reproduction, and hence, recruitment, especially for sessile or semisessile broadcast-spawning species (Smith & Rago 2004).

Critical density thresholds have been identified for broadcast-spawning species across a broad taxonomic range (NMFS 2009). Prince et al. (1988), McShane (1992), and Morgan and Shepherd (2006) have demonstrated correlations between densities of adult and newly recruited juvenile abalone at study locations on the southern Australian coast. Babcock and Keesing (1999) estimated critical density thresholds at 0.15–0.20/m^2 for greenlip abalone (*Haliotis laevigata*), a Southern Australian species that shares habitat, depth range, and spawning season characteristics with pinto abalone. These patterns are consistent with models in which larval dispersal from natal populations is limited, a relatively common circumstance in broadcast-spawning marine invertebrates with lecithotrophic larvae. It follows that if abalone populations are below critical density thresholds, spawners do not move, and gamete/larval immigration from distant populations is unlikely, then populations will not be sustainable. This research has been relied on by other investigators to set recovery thresholds for a variety of subtidal abalone species along the West Coast of North America (CDFW 2005, DFO 2007, NMFS 2008). The validity of applying this threshold to pinto abalone has not been tested.

Despite apparent risks of local extinction when critical density thresholds for abalone seem to be violated, there are several examples in California where combinations of circumstances have allowed populations to recover to densities above the critical thresholds (e.g., black, green, pink, and red abalone; VanBlaricom personal communication, Kushner personal communication). These circumstances may include aggregation behavior (i.e., movement) resulting in decreased distances between spawners, lengthened larval period (>10 days), and/or conditions that allow for gamete and/or larval dispersal over long distances (on the order of tens of meters). Thus, for most abalone species, key data gaps to fill are as follows: (1) the identification of critical threshold nearest-neighbor distance (or another appropriate aggregation metric); (2) the capacity for adult abalone to move closer to their nearest neighbor during spawning season; (3) a better understanding of larval longevity under natural conditions; and (4) circumstances by which gametes and/or larvae can disperse across longer distances (see "Larval Dispersal" below).

VanBlaricom (unpublished data) measured nearest neighbor distances for black abalone, an intertidal abalone found from Northern California to Mexico, at a California Channel island from 2004 to 2007. Despite dramatic disease-induced reductions in population density beginning in 1992, most animals in surveyed populations remain in close proximity to conspecifics. The data indicate that black abalone have remained strongly aggregated despite the following: (1) reductions in density averaging nearly 99% at nine separate sites on the island since 1992; and (2) densities below the threshold for successful reproduction (0.34/m^2; Neuman et al. 2010) identified for this species at all but one site on the island. The mechanism for maintaining close proximity to other animals may involve behavioral responses to the presence of conspecifics or alternatively may reflect active selection of particular microhabitat types that facilitates aggregation as a side effect. Regardless of the cause, it appears that persistent gregarious distributions in black abalone have the potential to forestall negative population-level effects of drastic reductions in density. These data emphasize the value of assessing the variance structure in density data and the mean values when considering critical density thresholds.

Density variance structure data are lacking for pinto abalone across much of the species' range. Only Seamone and Boulding (2011) studied aggregation characteristics of pinto abalone during one spawning season in Barkley Sound, British Columbia. Nearest neighbor R ratios (i.e., mean distance between individual pinto abalone) were significantly less than one, indicating aggregation, and densities ranged from 0.12 to 0.64 abalone/m^2. Aggregations were independent of gender, and therefore, the probability of encountering an individual of the opposite gender increased with increasing density. Indications overall are that pinto abalone in this region were sufficiently aggregated during spawning to potentially increase fertilization rates and compensate for low densities.

FERTILIZATION

Female pinto abalone produce 0.3–2.3 × 10^6 eggs per spawn, depending on animal size (Campbell et al. 2003). Bouma (2007) states that sperm densities recommended for fertilization in a hatchery setting have ranged from 200 to 10,000 per egg. Sperm density is not as important as the time over which sperm is allowed contact with the egg, which should be a maximum of 2–5 min to prevent polyspermy. Clavier (1992) evaluated the relationship of sperm concentrations per unit volume of sea water to fertilization rate for the ormer (*Haliotis tuberculate*), a species that shares a similar depth distribution with pinto abalone (0–20 m). Fertilization was not observed at sperm concentrations less than 10^3 cells/mL. Fertilization rate was found to increase steadily as sperm concentration increased above 10^3 cells/mL, approaching 100% fertilization at approximately 10^5 cells/mL. Fertilization rates fell to zero at sperm concentrations above 10^6 cells/mL.

LARVAL DISPERSAL

Direct measurement of larval travel patterns typically is not tractable for broadcast-spawning marine invertebrates. Planktonic larvae are so small and fragile that effective methods for marking and direct tracking of movements do not exist (e.g., McShane et al. 1988). Three indirect alternative methods are used to estimate larval dispersal distances empirically, and all have been applied to the problem of determining dispersal distances in abalone. The first is to use objects such as drift cards or drift bottles, labeled with appropriate identifying information, as surrogates for larvae and collecting data on recovery times and locations to make inferences about larval dispersal distances (e.g., Tegner & Butler 1985, Chambers et al. 2005). The second approach is to use molecular tools to establish relatedness of adult populations and newly recruited cohorts, allowing inferences about dispersal pattern (e.g., Hamm & Burton 2000, Chambers et al. 2006). Most recent research on dispersal distances in marine invertebrate larvae relies on molecular tools. The third approach is to examine the spatial relationship of newly recruited cohorts to known aggregations of breeding adults (e.g., Prince et al. 1988). Dispersal distance is estimated based on the range of distances measured between sites occupied by adults and sites occupied by groups of juveniles. This approach is most likely to provide accurate results along linear coastlines with relatively simple nearshore current patterns, with increasing errors of interpretation as the complexity of the habitat and dispersal distance capability of the species increases. Each of these methods include biases and sources of error that must be considered when interpreting the results.

Because specific studies for pinto abalone are limited, information that is available regarding dispersal distances for other abalone species is considered here. Overall, the three indirect methods for assessing larval dispersal distance in abalone point to consistent results, indicating limited larval dispersal distances along the coasts of southern Australia and California (Prince et al. 1987, 1988, McShane et al. 1988, McShane 1992, Hamm & Burton 2000, Chambers et al. 2005, 2006, Gruenthal 2007, Gruenthal et al. 2007). Given that most abalone larvae are in the plankton for a period of about 3–10 days before settlement and metamorphosis (e.g., McShane 1992), it seems to follow that abalone in general should have limited capacity for dispersal over distances beyond a few kilometers. Nonetheless, observations in southern California suggest that dispersal of abalone larvae (pinto, red, pink, and black) over a few kilometers has occurred in multiple areas and on multiple occasions (Kushner personal communication). A recent review of larval abalone dispersal modes provides evidence for short-, long-, and mixed-distance dispersal modes for the North Pacific abalones depending on a variety of biological factors including timing of spawning, larval duration, and vertical behavior of larvae (Miyake et al. 2017). Available information on the genetic structure of pinto abalone populations suggests that long-distance dispersal events occur frequently enough to maintain high gene flow among populations over distances of at least 1,000 km (Withler et al. 2001).

SETTLEMENT

Keough and Downes (1982) define settlement of the larvae of marine benthic invertebrates, including abalone, as contact with the postlarval substratum, resulting in metamorphosis from the larval to the postlarval form. Morse (1990) presented evidence that abalone larvae require exogenous chemical induction for settlement and metamorphosis, and Morse and Morse (1984) suggested that settlement cues associated with crustose coralline algae are related specifically to certain chemicals produced by them and present only on their surfaces (Morse 1992). These studies collectively involve a number of abalone species, and there is sufficient empirical evidence to suggest that the mechanisms described for the haliotids apply in general form to the settlement process in pinto abalone.

The sequence of studies and aforementioned discoveries suggests that availability of crustose coralline algae in appropriate habitats may be significant to the success of the larval recruitment process in pinto abalone. Although crustose coralline algae are ubiquitous in rocky benthic habitats along the West Coast of North America, a mechanistic understanding of processes that sustain these algal populations has not been established. If the presence of pinto abalone serves an important role in facilitating abundance of crustose coralline algae, it follows that the question of critical density thresholds (see aforementioned discussion of ‘‘Spawning Density’’) takes on a new dimension. That is, the critical density concept may apply to minimum densities needed to maintain community structure and function, including promotion of crustose coralline algal abundance, as well as to maintain minimum local abalone densities required for fertilization rates adequate to forestall local extinction. Field observations along the British Columbia coast indicate differential distribution of juveniles and adults, with juveniles observed at deeper depths, suggesting that settlement of larvae occurs in deeper habitats (Sloan & Breen 1988). Thus, settlement may be influenced by other environmental factors independent of the presence and/or density of adults.

RECRUITMENT

In the context of this review recruitment is defined as the appearance in one or more locations of measurable numbers of new, postmetamorphic abalone. This definition recognizes the possibility that recruitment may occur regardless of the local presence of breeding adults. This definition is ecologically based and should not be confused with the traditional and familiar recruitment definition used in the context of fisheries management. In the latter, recruitment is defined as the appearance in one or more harvested populations of new individuals that have reached a size large enough for legal harvest.

As noted previously, Prince et al. (1987, 1988), McShane et al. (1988), and McShane (1992) have presented evidence that recruitment of abalone is most likely to occur in relatively close spatial proximity to aggregations of breeding adults, at least in part a consequence of the relatively short duration of the planktonic larval phase in abalone. McShane (1992) reviewed literature emphasizing the significance of coastal current regimes, including eddies and other regional-scale flow patterns, in distributing planktonic larvae and influencing locations of settlement and recruitment. Some flow features apparently serve to concentrate larvae and may foster the development of recruited cohorts at high density in appropriate benthic habitat (e.g., McShane et al. 1988). By contrast, flow patterns that disperse larvae of benthic species to inappropriate habitats, such as the open sea, may ultimately increase larval mortality rates and diminish recruitment rates (e.g., Strathmann 1985). Predation and starvation may also influence numbers and distribution of planktonic larval abalone (Strathmann 1985), but no data were available to provide a basis for associating mortality rates to either of these processes.

An overriding problem in quantifying movement and fate of planktonic phase larvae of abalone is the virtual absence of data on abalone larval distribution in the water column, for any abalone species in any location on any scale (e.g., McShane 1992). As a result, effects of larval-phase population dynamics on recruitment processes for abalone can be evaluated only on the basis of various indirect measures. As noted previously and in previous sections, abalone recruitment appears to be influenced by the distribution of breeding adults, densities of adults on a local scale; availability of benthic recruitment substrata that provide appropriate chemical cues for settlement and metamorphosis of larvae; regional and local flow regimes that control larval dispersal from natal sites; and possibly predation and starvation of larvae.

Some information is available regarding recruitment in pinto abalone populations. As described in the ''Fishery-Independent Information'' portion of this review, data from index site surveys indicate that populations in Washington are experiencing recruitment failure, whereas populations in British Columbia have had successful recruitment despite continued declines in overall densities. A study by Zhang et al. (2007) estimating stock recruitment relationships for populations at Haida Gwaii and along the central coast of British Columbia found that poaching, rather than lack of recruitment, is an important factor limiting recovery in Canada. This is possible based on preliminary results from 2011 and 2012 surveys in these areas, showing an increase in population densities in areas where poaching has been reduced (Lessard personal communication). A more recent examination of the time series data (2012 to 2016) suggest that mature abalone densities above the threshold necessary for successful reproduction, favorable settlement conditions, and reduced predation pressure [from sea otters and/or sunflower seastars (*Pycnopodia helianthoides*)] may be contributing to higher densities of new, postmetamorphic abalone (Chandler et al. 2017). There is also evidence of recent recruitment events in California and Mexico based on the observation of animals less than 50 mm SL (Boch et al. 2014; Bird personal communication, Parnell personal communication). Alaska Department of Fish and Game has observed mixed age classes in some areas in Southeast Alaska, including juveniles (Walker personal communication). The cryptic nature of juvenile pinto abalone makes the detection of recruitment events difficult. Small juveniles (<10 mm SL) have occasionally been observed under boulders and on smooth bedrock or boulders that are bare or encrusted with coralline algae (Breen 1980a). Juveniles tend to occupy highly cryptic habitats in deeper waters compared with adults (Sloan & Breen 1988). In surveys along the coast of British Columbia, only 60% of juveniles 10–70 mm in size were exposed, compared with 90% of individuals 70–90 mm size and almost all individuals greater than 90 mm in size (Boutillier et al. 1985). Thus, recruitment events may be occurring but going undetected in regions that are not surveyed on a regular, consistent basis.

In a few areas of Southern California where kelp forest monitoring (KFM) has occurred somewhat regularly over the course of a decade or more (San Diego, CA and Northern Channel Islands), pinto abalone recruitment rates appear to be variable resulting in episodic pulses of successful recruitment interspersed with periods of no recruitment (Parnell personal communication). This pattern has been observed in the absence of fishing pressure, predation by sea otters, and low densities of reproductively mature pinto abalone. As recognized in the preceding paragraph, one explanation for this may be that the frequency and spatial coverage of sampling is not adequate to capture pinto abalone recruitment events that are occurring consistently. Another plausible explanation that is supported based on studies focused on pinto abalone (Breen 1986) and other abalone species (VanBlaricom unpublished data) is that successful recruitment is variable in space and time and is dependent on naturally occurring, long-term fluctuations in ocean conditions.

GROWTH RATE AND MAXIMUM SIZE

In a conservation context, growth rate of abalone is important because of linkages to size, age, and reproductive potential. Growth is also important to understand because body size may be an important determinant of vulnerability to predation. Growth may also be a useful indicator of abalone health and may reflect patterns of temperature, food supply, and other environmental features that can be monitored in the interest of abalone conservation. Growth rate data are clearly of value in the context of fishery management as well (e.g., Day & Fleming 1992).

As noted previously, young postmetamorphic abalone are often cryptic in coloration and habitat use, making direct measurements of growth rate in the field difficult. The problem is compounded by the small size of young animals, such that tag attachment is largely intractable and, if attempted, may cause disturbance, injury, or death of the subject animal.

Abalone growth models have been developed primarily based on data from populations in British Columbia (Schnute & Fournier 1980, Breen 1986). Growth in pinto abalone, as measured by SL, appears to vary by location and season depending on factors such as exposure to wave action, temperature, and the availability and quality of food (Sloan & Breen 1988). Pinto abalone were found to grow faster in moderately exposed areas with giant or bull kelp, compared with highly exposed areas with *Pterygophora californica*; this is likely due to reduced capture efficiency of drift algae in such habitats (Sloan & Breen 1988). Laboratory studies by Paul and Paul (1981) show that the growth was the highest at 13.5°C, whereas growth was inhibited at 5.5°C. This corroborates laboratory and field observations indicating that pinto abalone growth is the greatest during May to August, when temperatures range between 13°C and 14°C (Paul et al. 1977, Larsen & Blankenbeckler 1980). Growth can also vary with age and maturity. For example, both Paul et al. (1977) and Larsen and Blankenbeckler (1980) found that growth slowed with the onset of sexual maturity. Larsen and Blankenbeckler (1980) studied tagged pinto abalone in southeast Alaska and found that growth decreased from about 19.1 mm per year in abalone less than 50 mm SL and about 12.6 mm per year in abalone from 50–74 mm SL to about 6.2 mm per year in abalone 75–99 mm SL and about 4.3 mm per year in abalone greater than 100 mm SL. Thus, estimates of age at size can vary considerably, for example, from 2–5 y of age for a 50-mm SL individual to 6–9 y (or more) of age for a 100-mm SL individual (Sloan & Breen 1988). Estimated longevity of at least 15–20 y is reasonable for pinto abalone (Shepherd et al. 2000).

In summary, available data on pinto abalone growth in captive settings suggest that young animals reach sizes of about 22 mm SL (range 8–32 mm SL) in their first year (Olsen 1984), then grow at rates of approximately 18 mm per year for the next several years (Sloan & Breen 1988). Growth begins to slow at lengths of about 50 mm, corresponding to the onset of sexual maturity. Growth can vary based on many factors besides age, including water temperature, season, food availability and quality, and exposure to wave action. The maximum recorded SL for pinto abalone is 165 mm (Breen 1980a).

ABUNDANCE

There are two types of data that can be examined to provide a better understanding of variation in pinto abalone abundance over time: fishery-dependent and fishery-independent data. Because of the general lack of formal data, information from peer-reviewed publications and published reports, as well as from observational reports from individuals or groups of people was included. Also, because both abundance levels and the availability of information vary by region, the information is summarized by the following general regions: Alaska, British Columbia, Washington (SJA), Oregon, California, and Mexico.

FISHERY-DEPENDENT INFORMATION

Alaska Fisheries

Harvest of pinto abalone in Alaska occurred in the commercial, sport, personal use, and subsistence fisheries. Data from these fisheries are limited to commercial landings by weight and general information for the other fisheries, but indicate an overall trend of declining catch in the 1980s and 1990s in both the commercial and subsistence fisheries. The commercial fishery was closed in 1995 and the sport fishery in 2012; both remain closed to date. The personal use and subsistence fisheries remain open and harvest is believed to be low. In 2016, ADF&G established index sites and collected baseline information for pinto abalone to determine the status of local abalone populations (with and without sea otters present) and inform region-wide management decisions and potential recovery efforts (Donnellan & Hebert 2017).

Commercial Abalone Fishery

Commercial harvest of pinto abalone, with a minimum size restriction of 76 mm SL, was occurring in southeast Alaska by 1962 (Woodby et al. 2000). Although pinto abalone in southeast Alaska can be picked by hand from the intertidal zone during extreme low tides, most of the commercial fishing effort used scuba or hookah diving gear in the subtidal zone (Rumble & Hebert 2011). Commercial harvest of pinto abalone in southeast Alaska was characterized by a significant increase in effort and harvest in the late 1970s and early 1980s, followed by a steep decline in catch in the late 1980s and 1990s (Fig. 5; Hebert & Walker unpublished data). The increase in effort can be attributed in large part to an increase in value from less than one dollar per pound in the early 1970s to greater than six dollars per pound by 1993 to 1994 (Woodby et al. 2000).

The observed harvest rate declines were likely due in part to declines in pinto abalone abundance and changes in regulations to limit the fishery (Table 1; Rumble & Hebert 2011). The season was open year-round before 1979. From 1979–1980, the harvest season was reduced to 287 days between September 1 and May 31. Size restrictions varied in the southeast Alaska fishery between years and fishing district with a general trend of increasing size minimums to decrease the harvest rate of mature abalone because of concerns regarding abalone abundance (Woodby et al. 2000). Additional fluctuating reductions in season length occurred in the remaining years of the fishery (Table 1; Rumble & Hebert 2011). A commercial harvest limit (Guideline Harvest Range) was first put in place by ADF&G during the 1980 to 1981 season (250,000 pounds; Rumble & Hebert 2011). Catch per unit effort dropped significantly through the late-1980s and 1990s (Fig. 5). The commercial harvest limit was reduced multiple times over the following 15 y, but adjustment of the season length was the primary factor limiting the total annual harvest (Woodby et al. 2000). In 1981 to 1982, the annual harvest of 370,894 pounds far exceeded the ADF&G Guideline Harvest Limit of 100,000–125,000 pounds, even though the season was curtailed to 59 days following closure by emergency order (Rumble & Hebert 2011).

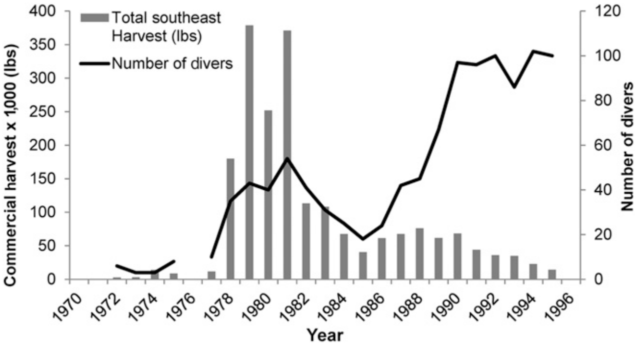

Figure 5. Commercial harvest amount and number of divers in the pinto abalone dive fishery in Southeast Alaska from 1970 to 1996. Adapted from K. Hebert and S. Walker presentation, March 2014.

Before 1996, the commercial dive fisheries in Southeast Alaska were open access. As new markets opened and ex-vessel value increased, fishing effort expanded to levels that made it difficult for the ADF&G to manage each fishery (Rumble & Hebert 2011). Legislation passed in 1996 capped the number of participants in the four dive fisheries (i.e., geoduck, sea cucumber, red sea urchin, and pinto abalone) (Rumble & Hebert 2011). The commercial fishery for pinto abalone was closed in 1995 (Woodby et al. 2000), and remains closed. Legislation imposed a 4-y moratorium on entry of new participants into the remaining dive fisheries (Rumble & Hebert 2011). The new, limited-entry fisheries were implemented by the Commercial Fisheries Entry Commission for the red sea urchin in November 2000, and for geoduck and sea cucumber in May 2001 (Rumble & Hebert 2011).

Area closures were implemented in the pinto abalone commercial fishery beginning as early as 1977, with closed area expansions though 1985, to reduce conflicts with sport, personal use, and subsistence fisheries in nearby communities. Sitka Sound was also closed to commercial harvest of pinto abalone. These area closures remained in place until the entire commercial fishery was closed in 1995.

Sport Abalone Fishery

Abalone harvest has occurred in the sport abalone fishery (for nonresidents), but data on trends in harvest are not available. In the sport fishery, the daily bag limit was 5 abalone per day (minimum size: 3.5 inches), with no closed season.

TABLE 1.

Registration Area A (southeast Alaska) commercial abalone harvest, effort, value, and season length, 1970/1971 through 1996/1997 (Rumble & Hebert 2011).

Season	Guideline harvest range (lb × 1,000)	Southern southeast harvest (lb)	District 13 harvest (lb)	Total southeast harvest (lb)	Number of divers	Exvessel value	Season length (days)
1970/1971	–	–	–	–	–	–	365
1971/1972	Confidential data—less than three divers reporting landings						365
1972/1973	–	65	2,610	2,675	6	$2,675	365
1973/1974	–	–	3,000	3,000	3	$4,500	365
1974/1975	–	–	13,826	13,826	3	$20,739	365
1975/1976	–	55	8,497	8,552	8	$17,104	365
1976/1977	Confidential data—less than three divers reporting landings						365
1977/1978	–	805	10,861	11,666	10	$14,816	365
1978/1979	–	130,607	49,320	179,927	35	$253,697	365
1979/1980	–	316,952	61,733	378,685	43	$408,980	287
1980/1981	250	233,589	18,382	251,971	40	$420,792	273
1981/1982	100–125	338,305	32,589	370,894	54	$445,073	59
1982/1983	100–125	100,458	12,826	113,284	41	$240,162	36
1983/1984	100–125	99,294	8,735	108,029	31	$302,481	126
1984/1985	100–125	59,237	8,379	67,616	25	$165,659	151
1985/1986	25–58	32,817	7,720	40,537	18	$117,963	71
1986/1987	25–58	47,404	13,820	61,224	24	$168,366	146
1987/1988	25–58	57,209	10,406	67,615	42	$208,930	36
1988/1989	25–58	65,928	10,172	76,100	45	$307,444	33
1989/1990	25–58	57,784	4,020	61,804	67	$330,651	40
1990/1991	25–58	62,779	5,607	68,386	97	$374,071	9
1991/1992	25–58	35,987	8,095	44,082	96	$267,578	35
1992/1993	25–58	26,905	9,083	35,988	100	$386,151	19
1993/1994	25–58	27,680	7,172	34,852	86	$487,928	7
1994/1995	25–58	15,055	7,824	22,879	102	$330,373	8
1995/1996	0–16	8,524	5,828	14,352	100	$126,526	6
1996/1997	Closed						

Scuba and hookah gear were allowed until 1996. The Alaska Board of Fisheries closed the sport abalone fishery in 2012 and it remains closed at present.

Personal Use Abalone Fishery

Abalone harvest still occurs in a personal use abalone fishery (only open to residents of the State of Alaska), but data on trends in harvest are not available. In 2012, the Alaska Board of Fisheries reduced the daily bag limit to 5 pinto abalone per person. These regulations are currently unchanged. Before 2012, the daily bag limit for personal use harvest of pinto abalone was 50 per person, except in one area, around Sitka, where there was a daily bag limit of 20 abalone per person. The minimum size limit of 3.5 inches remains unchanged from past years. There is no closed season. Scuba and hookah diving was legal before 1996 but is now prohibited for harvesting abalone for personal use. Alaska Department of Fish and Game believes that personal use harvest of pinto abalone in Alaska is low (ADF&G comments to NMFS, January 17, 2014).

Subsistence Abalone Fishery

The ADF&G Division of Subsistence has monitored the past subsistence harvest of pinto abalone in southeast Alaska and found a significant decline (98% decrease) in the subsistence harvest from 1972 to 1997 (Bowers et al. 2011). In 2012, the Alaska Board of Fisheries reduced the daily bag limit for subsistence harvest to 5 pinto abalone per person, with no closed season and no annual limit (Bowers et al. 2011). In the past, the daily bag limit for subsistence harvest was 50 abalone per person. The minimum size limit remains unchanged at 3.5 inches (Bowers et al. 2011). Scuba and hookah diving is prohibited for harvesting pinto abalone for subsistence use but was legal before 1996. Current legal harvest methods include use of snorkel equipment, abalone irons, or collection by hand. The ADF&G believes that the subsistence harvest of pinto abalone in Alaska remains low (ADF&G comments to NMFS, January 17, 2014).

Other Dive Fisheries in Southeast Alaska

Commercial dive fisheries for red sea urchin, sea cucumber, and geoduck clams still occur in southeast Alaska and are managed by the ADF&G. Monitoring by ADF&G for these existing dive fisheries suggest continued declines of pinto abalone. Additional data are presented below in the ''Fishery Independent Information'' section.

British Columbia Fisheries

Historically, pinto abalone in British Columbia were harvested in commercial, recreational, and traditional First Nations food, social, and ceremonial fisheries. All pinto abalone fisheries were closed in December 1990 because of concerns regarding population declines. Because there is very limited information on the recreational and First Nations fisheries, the review is focused primarily on the commercial fishery and harvest data.

Before the advent of scuba gear around 1960, harvest occurred primarily at low tide by "shore picking" (Farlinger & Campbell 1992), although some First Nations such as the Haida, used a two-pronged spear to take abalone as deep as 2 m below the lowest tide (Jones 2000). Recreational fishers also originally took abalone by shore picking; however, after the advent of scuba gear this sport or recreational fishery became widespread along the coast, including in remote areas with the operation of dive charters (Farlinger & Campbell 1992). No landing statistics are available for either the First Nations or recreational fisheries (Sloan & Breen 1988, Farlinger & Campbell 1992).

Although small, local, and sporadic commercial abalone fisheries began in British Columbia as early as 1889 (Mowat 1890, p. 261), the commercial dive fishery first began to expand significantly in 1972. Several sources (Fedorenko & Sprout 1982, Breen 1986, Sloan & Breen 1988, Farlinger & Campbell 1992, Muse 1998, Harbo & Convey 2006) have provided reviews of the history of the pinto abalone commercial fishery in British Columbia.

Landings in the British Columbia commercial abalone fishery from 1952 to 1990 are presented in Figure 6 (Quayle 1971, Fedorenko & Sprout 1982, Sloan & Breen 1988, Campbell 1997). Before the early 1970s, a small domestic market sustained a relatively small commercial fishery. A high landing record of nearly 60 metric tons (mt) occurred in 1972 and landings remained near this level until 1976 when they increased dramatically to 273 mt, topping out at over 480 and 400 mt in 1977 and 1978. A quota was first imposed on this open-access fishery in 1979 and catch that year dropped to about 200 mt. Various explanations for the occurrence of this "gold-rush" fishery (Breen 1986, Sloan & Breen 1988) have included the following: (1) advent of scuba and dry-diving suits that allowed more diver submergence time; (2) advent of on-board boat freezers; (3) emergence of a market in Japan for pinto abalone; (4) a tripling of the price per pound between 1972 and 1976 to over $3.00 (Canadian dollar) per pound; (5) redirection of fishing effort toward pinto abalone because of restricted access to salmon and herring fisheries; and (6) the unrestricted access to the abalone fishery before 1977 (Sloan & Breen 1988, Farlinger & Campbell 1992). Numerous management actions influenced the fishery after 1977 (Sloan & Breen 1988), and landings leveled out to between 44 and 47 mt under quota management. Breen (1986) estimated that at the beginning of 1976 the abalone stock stood at 1,800 mt, in areas that were then open to harvest. By the end of 1980, the stock size had been reduced to just 450 mt (Breen 1986). During the peak of the commercial fishery in 1977 to 1979

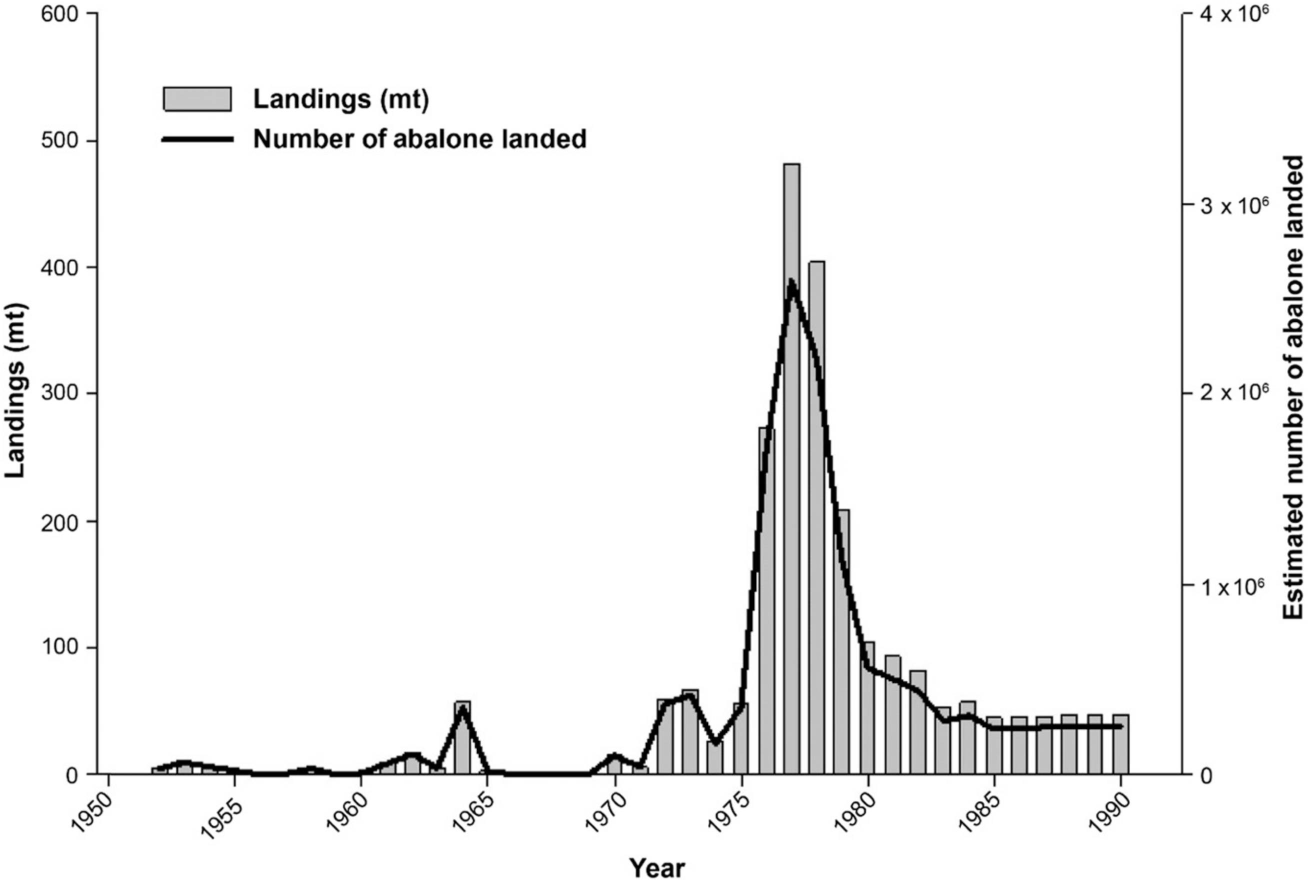

Figure 6. Biomass (mt) and estimated number of pinto (aka northern) abalone landed in the British Columbia commercial dive fishery from 1952 to 1990. Estimated number of individual abalone landed are based on the predicted mean weight of a legal-sized northern abalone of 159.7 g from 1952 to 1976 (≥90 mm SL) and 185.3 g from 1977 to 1990 (≥100 mm SL) (Quayle 1971, Fedorenko & Sprout 1982, Sloan & Breen 1988, Campbell 1997).

most abalone landings came from Central Coast (CC) areas and Haida Gwaii (Figs. 8 and 10; Breen 1986, Sloan & Breen 1988).

The number of individual pinto abalone landed each year from 1952 to 1990 in the commercial fishery based on landed biomass, and the predicted mean weight of legal-sized northern abalone is reported in Figure 6 (≥90 mm SL from 1952 to 1976, and ≥100 mm SL after 1976). It is estimated that as many as 2.5 million abalone were harvested in 1977, with at least a million abalone being taken each year from 1976 to 1979 and more than 240,000 being harvested each year during the last decade of the fishery (Fig. 6). After 1990, DFO banned all harvest of pinto abalone in British Columbia as a result of rapidly declining densities in survey sites and overall low population levels (Egli & Lessard 2011). The pinto abalone is currently listed as endangered (i.e., facing imminent extirpation or extinction) in British Columbia under the Canadian Species at Risk Act.

Washington Fisheries

There has never been a commercial fishery for pinto abalone in Washington State. Local indigenous peoples, and later, other early residents of Washington State reportedly harvested intertidal abalone; however, the magnitude and extent of this subsistence fishery are not well documented (WDFW 2014). Pinto abalone in Washington were first recognized as a recreationally harvestable shellfish with a daily possession limit of three abalone by Washington Administrative Code orders, first published in 1959. The daily possession limit was set at 5 per person and a 3.5 inch (~90 mm) minimum size limit (measured in horizontal line across longest portion of the shell) was implemented in 1980 (shell and body must remain intact in the field). In 1985 the retention of the first five legal-sized abalone harvested became required (i.e., upgrading was not allowed) and removal of undersized abalone from the water was banned (Blewett 2007). In 1990 the use of ''curved irons, knives, or other sharp instruments'' to harvest abalone was banned, and the harvest was limited to hands or round-edged ''abalone irons'' of specific dimensions to reduce injury to abalone that were removed from the substrate but not retained (Blewett 2007, WDFW 2014). In 1992, the daily possession limit was reduced from five to three, the minimum longest shell dimension was increased to 4 inches (101.6 mm SL), and it became a requirement to possess ''a 4-inch caliper and use it to determine if the abalone is of legal size before it is removed from its attachment'' (Blewett 2007, WDFW 2014). The Washington recreational pinto abalone fishery was closed in 1994 and it became ''unlawful to fish for or possess abalone taken for personal use the entire year'' (Blewett 2007, WDFW 2014).

Records of pinto abalone harvest in the Washington recreational fishery were not collected (Rothaus et al. 2008). Farlinger and Campbell (1992) cite a personal communication with Burge that this recreational ''harvest was probably about 12 t annually.'' Approximately 91% of the abalone harvest occurred in the North Sound region, which includes the SJA, and the remainder occurred in the SJF and just north of Admiralty Inlet. Gesselbracht (1991) conducted interviews with sport divers from September 1989 to August 1990 and reportedly estimated that 40,934 abalone were harvested annually. WDFW (2014) cautioned, however, that these self-reported harvest data may underestimate true recreational exploitation rates and also do not reflect cumulative harvest that has occurred over several decades.

California Fisheries

Human exploitation of abalone has occurred in the southern California Islands since the late Pleistocene (~10,500 y ago) and in central California for about 5,000 y. To date it has not been possible to reconstruct species-specific harvest information based on abalone shells observed in middens (Erlandson et al. 1996). Commercial abalone fisheries in California began in 1898 and during that same year the first cannery was built in Monterey County. From 1913 to 1928, commercial and recreational dive fisheries developed, with red (*Haliotis rufescens*), pink (*Haliotis corrugata*), and green (*Haliotis fulgens*) abalone being the targeted species, but only red abalone were documented in records of commercial landings before 1940 (Rogers-Bennett et al. 2002). By 1930, after the realization that significant declines in abalone landings were occurring, several laws restricting fishing methods, areas, sizes, numbers, and season were imposed on the abalone fishery (Croker 1931, Lundy 1997). Commercial fishing peaked at an annual harvest of more than 2,500 mt in 1957, and by 1994, the annual harvest had declined to approximately 140 mt (CDFW 2005).

Landings data, reported by the CDFW Abalone Recovery and Management Plan, indicate that pinto abalone were landed at the Farallon Islands, Point Montara, Point Buchon, Point Conception, the northern and southern Channel Islands, Santa Barbara, San Diego, and the offshore banks from 1950 to 1997 (Fig. 7; CDFW 2005). The highest yields were reported at Point Conception and Point Loma based on CDFW landing receipts from 1950 to 1997. The peak of the fishery occurred in 1974 when approximately 10,000 pounds (4.5 mt) of pinto abalone were landed (CDFW 2005). Pinto abalone made up approximately 13% of the abalone population in Northern California in the 1970s according to (Gotshall et al. 1974). Cox (1962) reported that the species were often patchily distributed and that dense patches could occasionally be found in deeper waters. Overall, pinto abalone was not considered a major component of the California commercial or recreational catch (CDFW 2005); however, increased fishing pressure and over harvest led to landings less than 500 pounds annually (0.2 mt) by the 1980s. California Department of Fish and Wildlife closed all commercial and recreational abalone fisheries south of San Francisco in 1997. In 1999, CDFW effectively excluded pinto abalone from the red abalone recreational fishery in Northern California by increasing the minimum legal size limit to 178 mm for all species (Rogers-Bennett et al. 2002).

Rogers-Bennett et al. (2002) estimated baseline abundance (i.e., abundance before overfishing) for *Haliotis kamtschatkana assimilis* using landings data from the peak of the commercial and recreational fisheries (1971 to 1980). The baseline minimum estimate of abundance for *H. k. assimilis* before overexploitation was 21,000 animals. After 1980, only 66 animals were landed suggesting a decline of 99.6% over a 10-y period. This estimate provides a historic perspective on patterns in abundance, defines a relevant baseline abundance against which to compare modern-day trends, and helps to assess the species' current status and risks. It is important to note that this method

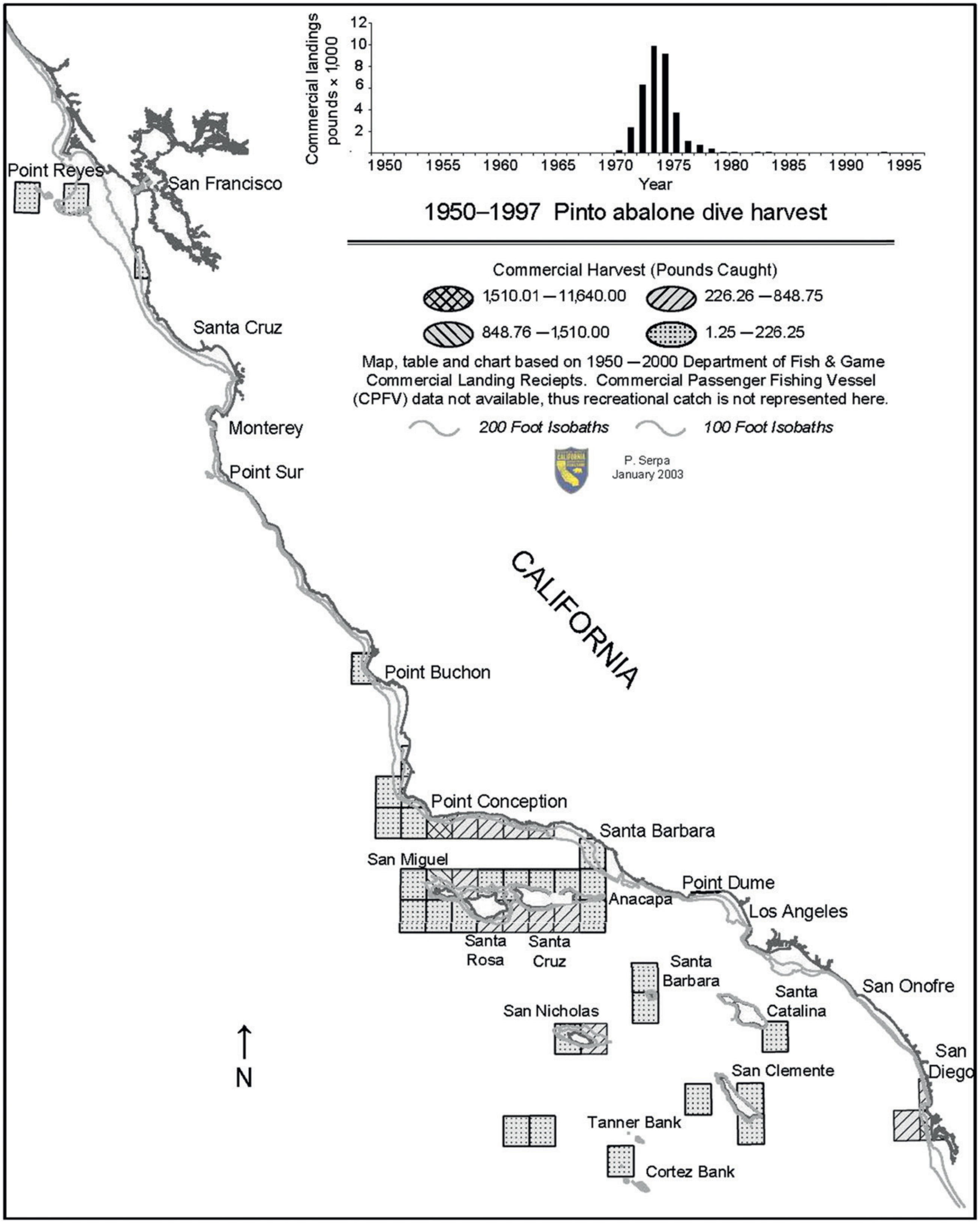

Figure 7. Abalone landings data for pinto abalone recorded by the CDFW 1950 to 1997 (CDFW 2005).

assumes that the population was at least as large as the number taken in the fishery, that the fishery "sampled" all size classes, and that no new individuals were added to the population during the 10-y peak of the fishery. In addition, this estimate was based on data from a time period when pinto abalone abundances may have been higher than usual due to the decline of sea otters along the California coast; thus, this estimate may not reflect the true baseline abundances that existed before the abalone fishery and the exploitation of sea otters.

Mexican Fisheries

The abalone fishery in Mexico dates to approximately 1860 (Bonnot 1930, Lundy 1997), but modern commercial harvests did not develop until the 1940s. The fishery is pursued by 22 fishing cooperatives, distributed across four management zones on the Pacific coast of the Baja California Peninsula (Fig. 4). Five cooperatives are present in the management zone 1, which is the northernmost of the zones and extends from the U.S.–Mexico border and the Coronado Islands, to just south of Isla Creciente, Baja California Sur, and includes Isla Cedros. Although green and pink abalone historically have been the primary catch in Baja California, both pinto and white abalone have been relatively abundant and harvested (Boch et al. 2014).

FISHERY-INDEPENDENT INFORMATION

Population Trends in Alaska

There is little trend information available on abalone populations in Alaska before the significant commercial fishery, and there has been no rigorous long-term monitoring of pinto abalone since the 1995 commercial fishery closure. Observations of pinto abalone made by ADF&G biologists from 1991 to 2012 while conducting dive surveys to monitor other benthic invertebrate species for management purposes suggested a continued decline of pinto abalone since the fishery closure (Fig. 8). Alaska Department of Fish and Game noted increases in empty abalone shells and areas devoid of live abalone where they were once common while conducting sea urchin surveys between 2001 and 2012 (Hebert personal communication). More recently, Donnellan and Hebert (2017) reported continued declines in abundance (abalone count/minute) based on pinto abalone–targeted timed swims at the same sites sampled in 1979 to 1981, 1986, 1997, and 2016 (Fig. 9; Donnellan & Hebert 2017).

From 2015 to 2016, biologists from ADF&G, the Sitka Sound Science Center and University of California Santa Cruz established index sites in southeast Alaska to address knowledge gaps on pinto abalone population status and impacts of sea otter predation, and to inform management and conservation actions. At study sites near Ketchikan and Craig, average absolute adult (≥50 mm) densities were low ($0.17/m^2 \pm 0.03$ SE to $0.29/m^2 \pm 0.08$ SE) despite evidence of ongoing recruitment (i.e., juveniles <50 mm SL were present at all but one site) and the cause(s) for low numbers was uncertain (Fig. 10; Donnellan & Hebert 2017). At most study sites near Sitka, average absolute adult densities were above critical levels thought necessary for successful reproduction ($0.2/m^2$; Babcock & Keesing 1999) and young recruits (<20 mm SL) were present at nearly all sites (Bell et al. in prep). These initial studies highlight that there is no apparent relationship between adult densities and recruitment and that high levels of spatial variability in abalone densities exist in southeast Alaska with no clear indication of the forces driving that variation. A better understanding of the dynamics of these populations and their present and future risk factors (i.e., expanding sea otter

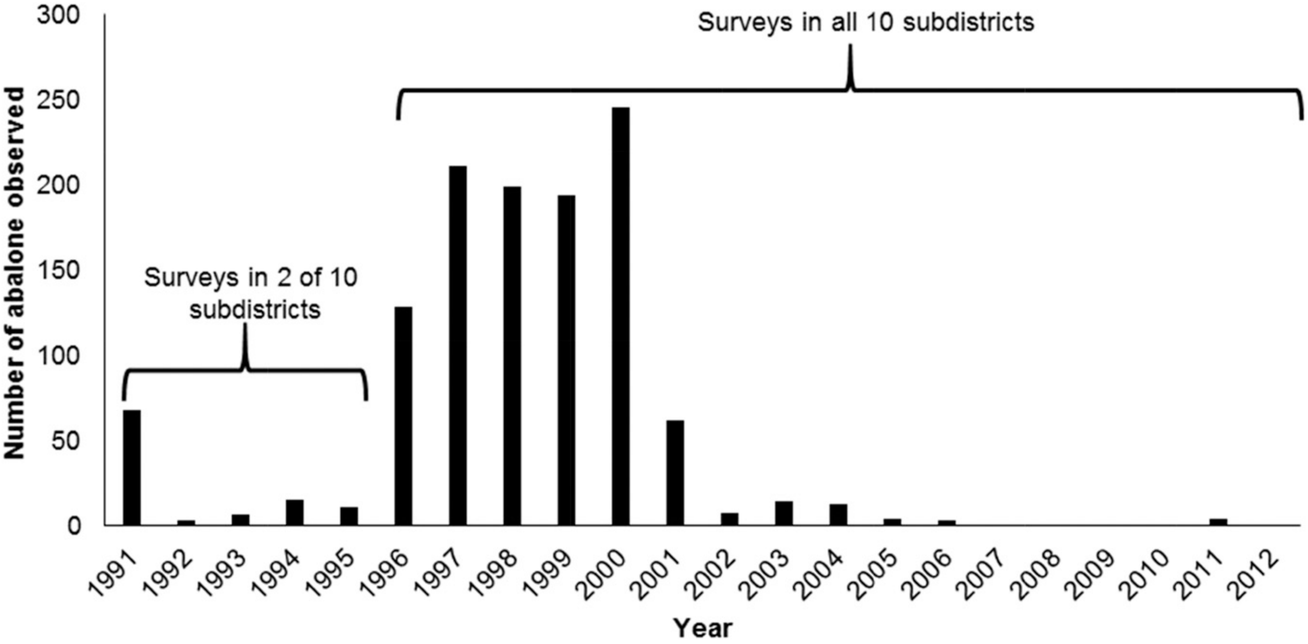

Figure 8. Observations of abalone, summed over transects and subdistricts for a given year, recorded by ADF&G during red sea urchin surveys. Two of 10 subdistricts were observed from 1991 to 1995 and all subdistricts (*n* = 10) were observed from 1996 to 2012. Data were obtained by chance encounters with abalone during surveys designed to estimate red sea urchin density and was recorded on a volunteer basis by survey divers. (ADF&G abalone observations 1991 to 2012; ADF&G comments to NMFS, January 17, 2014).

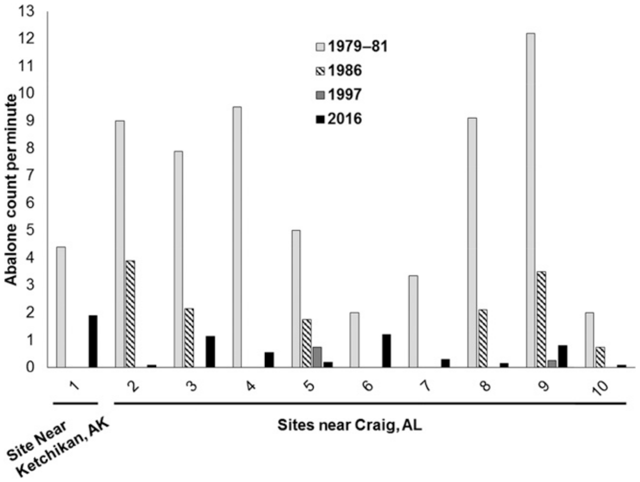

Figure 9. Relative density of pinto abalone (all sizes combined) by survey year for all sites surveyed using timed swims and for which historical data exist. Historical data compiled from Blankenbeckler and Larson (unpublished data), Shepherd et al. (2000) and Woodby et al. (2000). Adapted from Donnellan and Hebert (2017).

populations and unmonitored personal use and subsistence fisheries) could be achieved by continuing to monitor index sites and by using conservative management to prevent further declines (Donnellan & Hebert 2017, Bell et al. in press).

Population Trends in British Columbia

Early observations of pinto abalone (aka northern abalone) in British Columbia were reported by Thompson (1914), particularly around Haida Gwaii and in Queen Charlotte Sound, although no estimates of abundance were given (Fig. 4). A private, exploratory survey for abalone at 26 sites in southeastern Haida Gwaii was conducted in 1955 by a diver with "hard hat" gear and found that pinto abalone were present at 54% of the sites; present but relatively rare at 15% of the sites; and absent from 31% of the sites (Quayle 1962, Sloan & Breen 1988). Breen and Adkins (1979) sampled the same area in 1978 and, by contrast, found pinto abalone present at almost all sites and recorded an overall mean density of 2.5 abalone/m^2.

The Department of Fisheries and Oceans Canada (DFO) began conducting index site surveys in 1978 in British Columbia to monitor populations of pinto abalone. Over the years, most of these abalone surveys have occurred in southeastern Haida Gwaii and along the CC of British Columbia (Fig. 4) where pinto abalone were most abundant and most of the commercial harvest was taken (Sloan & Breen 1988, Egli & Lessard 2011). Index survey sites were selected based on the presence of harvestable densities of abalone, with surveys beginning in 1978 to 1980 and conducted on a 5- y rotation since 2001. Occasional surveys have also been conducted in the Strait of Georgia, Johnstone Strait, Barkley Sound, and the far north coast (Fig. 4; Cripps & Campbell 1998, Lucas et al. 2002a,

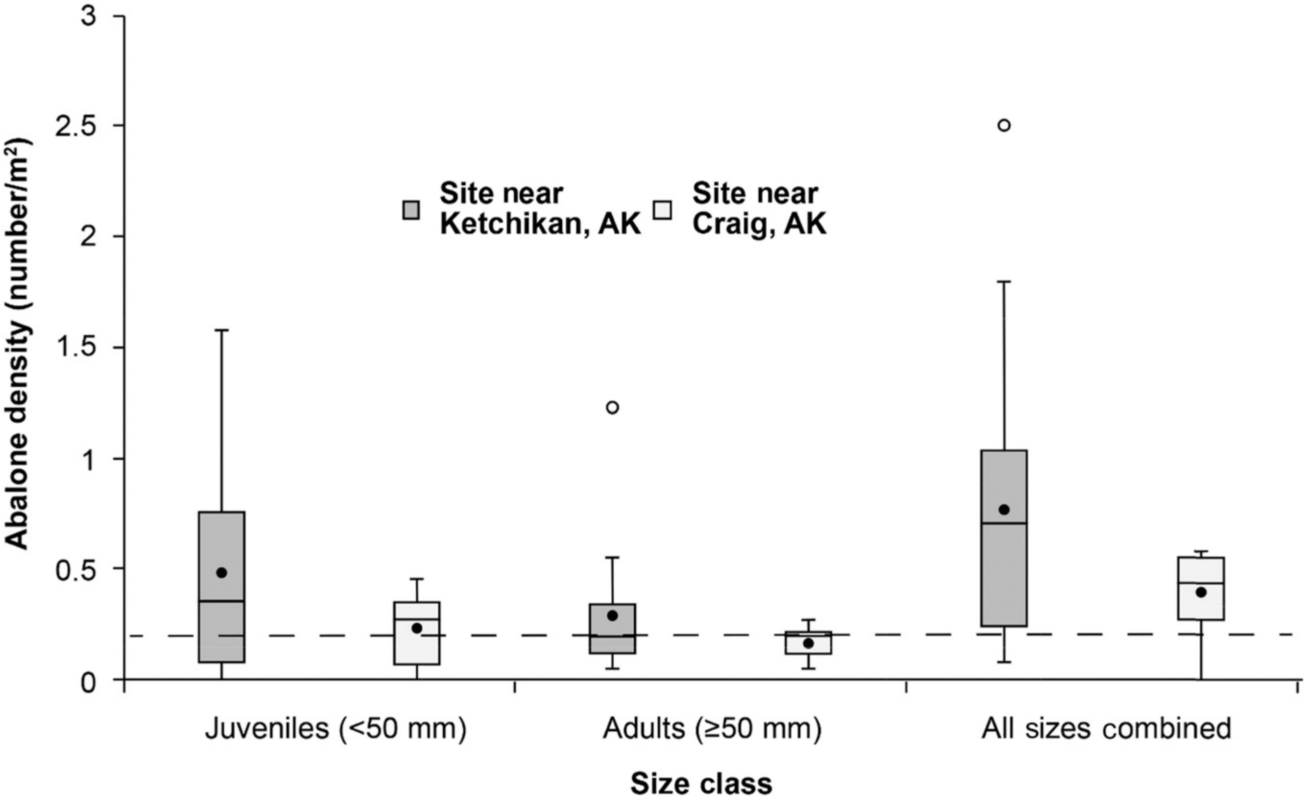

Figure 10. Box and whisker plots of absolute density of abalone by size class and study area (replicate = site). The dashed horizontal line represents the mean density threshold density for recruitment failure of 0.2/m^2 estimated by Babcock and Keesing (1999). The bottom and top of each box are the first and third quartiles, respectively, and the horizontal line inside the box is the second quartile (i.e., the median). The mean is indicated by an "x", and the positive and negative whiskers extend to the maximum and minimum quadrat density values, respectively. Adapted from Donnellan and Hebert (2017).

2002b, 2002c, 2002d, 2002e, Lessard et al. 2004). Department of Fisheries and Oceans abalone surveys began in 2003, 2004, and 2008 on the West Coast of Vancouver Island, Queen Charlotte Strait (QCS), and Haida Gwaii, respectively (Chandler et al. 2017). Survey sites were chosen randomly within general areas where abalone were known to be present.

Two fishery-independent survey methods have been used by DFO since index sites were established in 1978: (1) the " Breen survey" (Breen & Adkins 1979) whereby divers select a site predetermined to contain pinto abalone habitat, place a 1 m^2 quadrat on top of that habitat, and then proceed to count and measure emergent abalone and score habitat within a 7 m × 16 m area; and (2) the "transect survey method" (Cripps & Campbell 1998) whereby transects of varying lengths are randomly selected, 1 m × 1 m quadrats are searched by divers moving from deep to shallow water on either side of the transect, emergent abalone are measured and counted, and macroalgal cover is noted (Lessard & Egli 2011, DFO 2016). Below, the available information regarding pinto abalone abundance from these index site surveys and additional surveys conducted within the sites is summarized.

Haida Gwaii

Since the first survey in 1978, 10 abalone density surveys have been conducted at index sites using the Breen survey methodology in the Haida Gwaii area of British Columbia (Hankewich et al. 2008; Lessard personal communication). Results of these surveys have been presented in numerous reports published by DFO (Breen & Adkins 1979, Boutillier et al. 1985, Carolsfeld et al. 1988, Thomas et al. 1992, Winther et al. 1995, Campbell et al. 2000, Atkins et al. 2004, Hankewich et al. 2008, Chandler et al. 2017). Before the main impact of the commercial dive fishery in eastern Haida Gwaii, Breen and Adkins (1979) found abalone densities that averaged 16 abalone/m^2 and that ranged from zero to as high as 28 abalone/m^2 (Sloan & Breen 1988). Adkins and Stefanson (1977) also reported high densities of 4.4–10.0 abalone/m^2 in unharvested areas in Haida Gwaii in 1976. In 2007, the mean size of surveyed pinto abalone in Haida Gwaii was 61.5 mm, the smallest mean SL observed since 1978, and abalone were observed at 80.5% of the 82 total sites surveyed (Hankewich et al. 2008). The mean densities of large adult and mature abalone (0.03 and 0.15 abalone/m^2), the percentage of sites with large adult abalone, and the percentage of quadrats containing abalone fell below the short-term targets in the DFO National Recovery Strategy (2007) (Hankewich et al. 2008).

The mean densities of abalone (all sizes combined; 1978 to 2016) from survey index sites on the east and west coasts of Haida Gwaii are shown in Figure 11. The number of immature abalone went up significantly between 2007 and 2016, indicating that recruitment occurred, which may be linked to the recent cold phase of ocean conditions in the northeast Pacific (Lessard & Curtis, personal communication). The mean density of mature abalone also increased over this time period. As of 2012, the density of mature abalone (≥70 mm SL) at five of the nine index survey sites in Haida Gwaii was at or above the short-term recovery objective of 0.32 abalone/m^2 (Lessard, personal communication). The estimated mortality rate of mature abalone in Haida Gwaii decreased from a previous estimate of 0.32–0.26 between 2007 and 2012 (Lessard, personal communication). By contrast to portions of the CC region of British Columbia, sea otters are currently not present in Haida Gwaii (Lessard, personal communication).

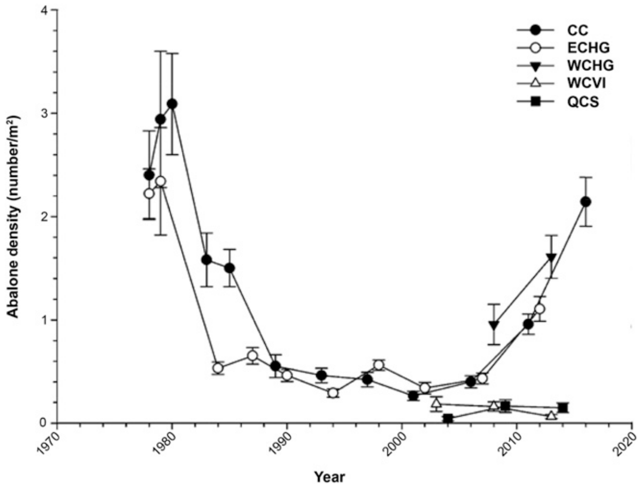

Figure 11. Total density of pinto abalone (all sizes combined) for each index site region 1978 to 2016: CC, East Coast Haida Gwaii (ECHG), West Coast Haida Gwaii (WCHG), West Coast Vancouver Island (WCVI), and QCS. Values are mean ±SE. Adapted from Chandler et al. 2017.

Central Coast

Since the first survey in 1978, 10 abalone density surveys have been conducted using the Breen survey methodology in the CC area of British Columbia (Hankewich & Lessard 2008; Lessard personal communication). Results of these surveys have been presented in numerous reports published by DFO (Breen & Adkins 1980, 1982, Boutillier et al. 1984, Farlinger & Bates 1986, Farlinger et al. 1991, Campbell & Cripps 1998, Campbell et al. 1998, Cripps & Campbell 1998, Lucas et al. 1999, 2000, 2002c, Lessard et al. 2007, Hankewich & Lessard 2008).

The most recent survey in the CC region in 2016 indicated that the mean total density (all sizes combined) increased more than 5-fold since 2006 and that recruitment occurred based on the large number of small individuals observed (Fig. 11; Table 2). Although the density of mature abalone (≥70 mm SL) at most of the index sites was at or above the short-term recovery target of 0.32 abalone/m^2 (Fig. 11), the size structure of the population was highly skewed toward smaller individuals (Table 2). Sea otters currently occur and are expanding in some areas within the central part of the CC. The estimated mortality rate of mature abalone in the CC region has decreased from a previous estimate of 0.33–0.25 between 2006 and 2011 (Lessard personal communication).

Strait of Georgia

Pinto abalone have been reported in the Strait of Georgia, but they are relatively rare (Quayle 1971). Their distribution may be limited to depths of more than 7 m in this area because of higher water temperatures and lower salinities at shallower depths (Sloan & Breen 1988). During timed swims on the southwest end of Vancouver Island in 1996 to 1997, Wallace (1999) found 211 abalone (0.77 abalone/min); however, in 2005 only a single abalone was seen across four sites surveyed in this region (DFO 2007). Across all 19 sites surveyed in 2005 on the south end of Vancouver Island only three abalone were found for a density estimate of 0.0098 abalone/m^2 (DFO 2007).

TABLE 2.

Representative SL data from the CC Index Sites (1978 to 2016).

Year	SL (mm) Mean	SE	Median	Min	Max	Total count
1978	94.9	1.0	97	30	149	454
1979	81.6	0.7	83	17	139	804
1980	68.2	0.7	70	14	123	959
1983	80.9	0.6	82	8	131	1,076
1985	82.5	0.8	85	6	129	702
1989	81.3	1.4	82	20	140	227
1993	78.4	1.1	80	16	126	421
1997	80.7	1.4	84	20	142	317
2001	77.6	1.4	81	29	122	230
2006	69.4	1.1	74	2	122	433
2011	63.8	0.8	66	3	124	1,137
2016	52.9	0.5	50	2	132	2,771

Adapted from Chandler et al. 2017.

By contrast to the southern Strait of Georgia, Egli and Lessard (2011) reported that abalone distribution in the northern Strait of Georgia "has been anecdotal and has not been systematically determined." Lucas et al. (2002d), using the transect survey method in 2000 and 2001, found a total of 49 abalone across a total of 21 transects for a mean density of abalone (all sizes) of 0.06/m^2. A conservative mean total population estimate of emergent (90–110 mm SL) abalone in the northern Strait was 1,577 (Lucas et al. 2002d). In October 2009, sites in the northern Strait of Georgia were surveyed and the overall density was found to be 0.013 ± 0.007 abalone/m^2 (Egli & Lessard 2011).

West Coast Vancouver Island

Atkins and Lessard (2004) surveyed abalone density and size in 2003 at 32 sites on the exposed outer coast of north-west Vancouver Island. Surveys were repeated and expanded in 2009 and 2014 (Curtis personal communication). In 2004, no abalone were found on the exposed outer coastline and most abalone in more protected locations were found at depths less than or equal to 4 m, indicating that surf conditions are too severe for abalone to occur in the shallow subtidal, where they are fully exposed to the open Pacific (Atkins & Lessard 2004). Mean abalone densities within two protected embayments were 0.212/m^2 and 0.038/m^2, respectively. Between 2004 and 2013, and by contrast to the other regions discussed previously, the densities in these embayments declined by almost 50% (Fig. 11).

Tomascik and Holmes (2003) found a mean density of 0.15 abalone/m^2 at 22 sampling locations across five island groups and two depth zones (2–5 m and 6–9 m) in Barkley Sound. Although some evidence of abalone recruitment was seen (42% of the sampled population were juveniles), densities were about four times lower than were seen before the commercial fishery closure by Emmett and Jamieson (1988) (Tomascik & Holmes 2003).

Queen Charlotte Strait and Johnstone Strait

In 1977, during the commercial fishery, Breen et al. (1978) surveyed 34 sites identified by previous fishing activity and found that abalone were abundant enough to sustain commercial fishing pressure at only two of the sites and possibly at a few others. Systematic surveys of abalone abundance using the Breen survey methodology were conducted in 2004 in Johnstone Strait (Davies et al. 2006) and in 2004 and 2009 in QCS (Lessard & Egli 2011). By 2008, there were over 200 sea otters in the QCS area, and sea otter predation and low recruitment were items of special concern (Lessard & Egli 2011).

In the most recent surveys (2009 and 2014), abalone were found at almost half of the index sites surveyed in QCS, and the overall density was approximately 0.1/m^2 (Lessard & Egli 2011, Chandler et al. 2017). Higher densities were observed in 2009 and 2014 compared with 2004 in QCS (nearly four times higher). Even though abalone abundance was low compared with the CC and Haida Gwaii (>1/m^2), abalone and/or their habitat were distributed throughout the area surveyed and densities appeared to be stable as of 2014 (Lessard & Egli 2011, Chandler et al. 2017).

Summary

Pinto abalone populations in British Columbia experienced large declines from the 1970s until the mid-2000s (75%–80%; Chandler et al. 2017) due to fisheries harvest and, after the fisheries closure in 1990, continued illegal harvest. Since the mid-2000s, the best available data indicate that recruitment is occurring and population density is increasing in Haida Gwaii and along the CC due to multiple contributing factors including a reduction in illegal harvest, natural recovery following fishery closure, and low predation pressure (i.e., from sea otters and sunflower seastars). In the Haida Gwaii and CC regions, short-term population and distribution objectives described in the DFO National Recovery Strategy (2007) have been met. The size structure of the populations in these areas is highly skewed toward smaller individuals, which indicates recent recruitment (Chandler et al. 2017). The persistence and reproductive contributions of animals that are currently small is critical to continued recovery and long-term population viability.

Pinto abalone populations are believed to have the capacity to recover in British Columbia, especially given that habitat does not appear to be limiting in this region. Evidence of successful juvenile recruitment throughout the years and recent increases in adult abundance and density (Fig. 11; Table 2) indicate that removing or reducing illegal harvest to minimal levels can have a positive impact on populations and promote recovery. Not all regions are showing recovery, however, especially those with sea otters. If sea otter populations expand and sunflower seastars recover, the positive trends that have been observed could slow or reverse (Chandler et al. 2017).

Population Trends in Washington

No estimates of pinto abalone biomass, population viability, or extinction risk in Washington have been made and historical levels of abundance are not well understood (WDFW 2014). Data are available from timed swim and index site surveys, as well as abalone recruitment studies, conducted in the SJA (Fig. 3). The best available data indicate that pinto abalone populations in Washington are declining despite the closure of fisheries, and local recruitment failure may be occurring.

San Juan Islands

Timed scuba swim surveys designed to quantify abundance and measure SL of pinto abalone in the SJA were conducted from 1979 to 1981 by WDFW and have been variously described by Rogers-Bennett (2007), Rogers-Bennett et al. (2011), and WDFW (2014).

Rogers-Bennett (2007) and Rogers-Bennett et al. (2011) reported on timed scuba swim surveys conducted in 2005 at 10 sites in the SJA that were selected because they were known to have had abundant pinto abalone populations in the past. Substantially fewer pinto abalone were observed in 2005 than during similar timed swim surveys conducted in 1979 by the WDFW (Rogers-Bennett et al. 2011). In 2005, only 17 pinto abalone ranging in size from 75 to 142 mm (mean SL of 107 mm) were observed during 30–40 min timed swims at the 10 sites (694 total minutes for two divers), with 82% of the animals found at just two sites. By contrast to the approximately 25.5 abalone encountered per 20-min dive during the 1979 to 1981 surveys, WDFW divers only encountered about 1.1 abalone per 20 min dive in 2010 to 2011 (about a 96% reduction in the encounter rate) (WDFW 2014). The mean SL of pinto abalone measured during timed swim surveys in 1979 was 97.6 mm ($n = 755$), whereas the mean SL of pinto abalone measured at the 10 index sites in 2013 was 118.4 mm ($n = 56$) (WDFW 2014) and in 2017 was 126.5 mm ($n = 17$) (WDFW unpublished data). This general trend from smaller, younger abalone to larger and presumably older individuals in more recent years is illustrated in Figure 12. Overall, the mean size of pinto abalone has been increasing by an average of 0.5 mm per year (WDFW 2014).

In 1992, WDFW began conducting nondestructive index site surveys at 10 locations in the SJA to monitor densities of pinto abalone. These index survey sites were established in areas known to have high pinto abalone abundance and ranged in size from 135 to 375 m^2 in area (Table 3). The 10 sites have been periodically resurveyed for abalone abundance, density, and SL, most recently in 2017 (Figs. 16 and 17). From 1992 to 2006, the mean density at the 10 index sites declined from 0.18 to 0.04 abalone/m^2, with a significantly faster decline observed in the deep stratum sites (4.4–9.0 m depth) than in the shallow stratum sites (0.5–4.3 m depth) (Rothaus et al. 2008). Washington Department of Fish and Wildlife resurveyed these same 10 index stations in 2009, 2013, and 2017 and confirmed that mean density has continued to decline; between 1992 and 2017, abundance at the SJA index sites has declined from 98% to about 0.005 abalone/m^2 in 2017 (Fig. 13; Rothaus et al. 2008, WDFW 2014; WDFW unpublished data). In addition, the percentage of emergent juvenile pinto abalone (SL <90 mm) seen during surveys has declined from 31.8% in 1979 to 17.4% in 1992, and most recently to 7.1% in 2013 (WDFW 2014). In 2017, only one of the total 17 animals encountered within all sites was emergent size (WDFW unpublished data).

TABLE 3.

Survey sites, survey area, and number of pinto abalone observed at each of 10 survey sites from 1992 to 2013 in the SJA.

		Pinto abalone index station abundance by survey year								
Index stations	Area (m^2)	1992	1994	1996	2003	2005	2006	2009	2013	2017
Site 1	152	48	36	7	8	2	4	4	1	1
Site 2	190	20	34	32	7	9	10	9	7	2
Site 3	316	46	21	8	0	1	0	1	0	1
Site 4	375	45	19	19	2	0	0	0	0	0
Site 5	135	41	23	3	1	1	0	0	0	1
Site 6	158	49	41	74	39	14	13	6	12	0
Site 7	176	49	30	18	17	10	5	5	1	1
Site 8	155	22	24	31	19	15	11	9	0	0
Site 9	229	22	29	14	18	11	12	17	0	1
Site 10	356	17	30	24	27	13	9	9	6	5
Totals	2,242	359	287	230	138	76	64	60	27	12

Adapted from WDFW (2014, 2017), unpublished data.

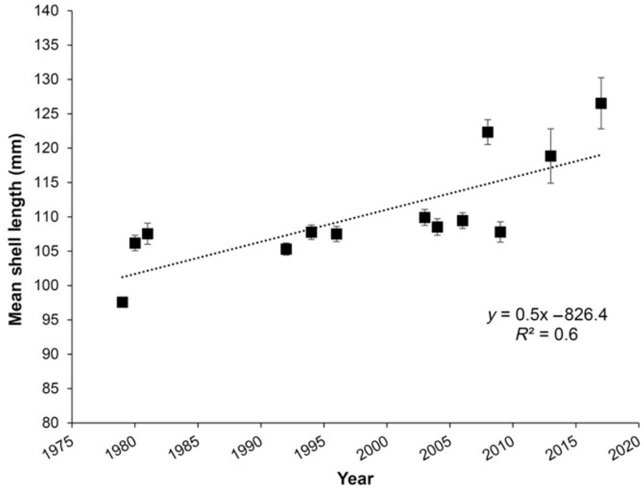

Figure 12. Pinto abalone mean (±SE of the mean) SL on timed survey transects and index station surveys in the SJA, WA. Based on SL measurements of 2,581 pinto abalone measured during timed swim surveys during 1979 to 1981 and index site surveys during 1992 to 2017. Adapted from WDFW (2014, 2017), unpublished data.

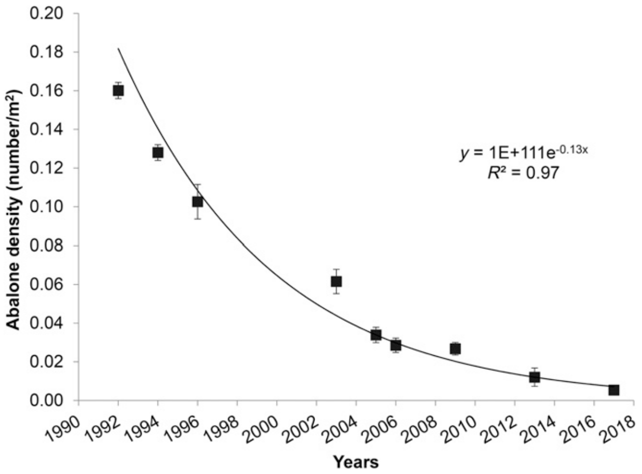

Figure 13. Mean density (abalone/m^2 ±SE of the mean) of pinto abalone from 1992 to 2017 at 10 index sites in the SJA. Half of the index stations were surveyed in each year 2004 and 2005; however, these data are presented as a single data point (2005) as described in WDFW (2014). Adapted from WDFW (2014, 2017), unpublished data.

To estimate recent juvenile pinto abalone recruitment, Bouma et al. (2012) deployed 60 ''artificial habitats,'' also known as abalone recruitment modules (ARMs) at three sites (20 ARMs per site) in the SJA that had supported large abalone populations in the past. Abalone recruitment modules at the three sites were examined by scuba divers six separate times during 2005 and 2006. Eight abalone (one adult, four emergent, and three juveniles) were observed at one of the three sites. Juvenile and emergent abalone density in the ARMs across all three sites in 2006 was 0.012/m^2 and 0.008/m^2, respectively. The scarcity of juveniles observed in the ARMs over the 2-y study suggests limited recruitment is occurring in pinto abalone populations (Bouma et al. 2012). Taken together with the observations in Rothaus et al. (2008), it is likely that local recruitment failure is occurring in the SJA (Bouma et al. 2012).

Strait of Juan de Fuca & Outer Coast of Washington

WDFW (2014) stated that pinto abalone have been observed in the SJF, but they do not have any data regarding trends in abundance in this area. Two quantitative benthic surveys for invertebrates and macroalgae were carried out along the south shore of the Strait, with data collections at 20 sites used for both surveys (USGS unpublished data). Sites were selected with a stratified random protocol. Two pinto abalone (in physical contact with one another) were seen in the 1997 survey and none were observed in 2001.

Pinto abalone have not been reported south of Portage Head on the outer Olympic Coast of Washington (WDFW 1992). Sloan and Breen (1988) stated that the species does not appear to occur in shallow waters at extremely exposed sites. Similarly, Atkins and Lessard (2004) did not find abalone on the open north-west coast of Vancouver Island but did find abalone in nearby protected sounds and bays, indicating that the shallow subtidal on extremely exposed coastlines may be too inhospitable for abalone to survive.

Population Trends in California

Northern California

Rogers-Bennett et al. (2002) estimated a baseline abundance of 153,000 *Haliotis kamtschatkana kamtschatkana* in northern California using estimates of fishery-independent densities and suitable rocky abalone habitat derived from data collected in 1971 and 1975. The 95% confidence intervals around this estimate were very large (upper 341,000 and lower 29,000) because of the patchy nature of the abundance data and limited sampling in 1971. Rogers-Bennett et al. (2002) compared this baseline estimate with a modern estimate of 18,000 abalone (upper 95% confidence interval 22,000; lower 95% confidence interval 13,000), derived from data collected in 1999 to 2000 at five sites in Mendocino County, indicating an estimated 10-fold decline in abundance between the 1970s and 1999 to 2000.

California Department of Fish and Wildlife conducted dive surveys at multiple sites from 2007–2013 in Mendocino County and 2007–2012 in Sonoma County (Rogers-Bennett unpublished data). Mean densities were higher in Mendocino County ($\sim$1.75/m^2) compared with Sonoma County ($\sim$0.65/m^2) and were higher at depths greater than approximately 10 m (see Fig. 32, NMFS 2014). The mean densities recorded for both counties at depths greater than approximately 10 m were above the critical density thresholds for successful reproduction reported for other species of abalone (Babcock & Keesing 1999, Neuman et al. 2010). In addition, smaller size classes (<50 mm SL) were better represented in Mendocino County compared with Sonoma County, suggesting that recent recruitment has likely occurred in Mendocino County (see Fig. 33, NMFS 2014).

Southern California

Generally, there are few reports of pinto abalone from Point Conception to the Mexican border from about 1980 to 1999 (Parnell personal communication, Kushner personal communication). In 1974, CDFW (Taniguchi unpublished data) conducted timed SCUBA searches in the northern Channel Islands (focusing on all abalone species that could be observed) and found a total of 53 pinto abalone off the southwest corner of San Miguel Island (range: 1.5–36.75 abalone per hour), 10 pinto abalone off the southwest corner of Santa Rosa Island (range: 3.0–4.2 abalone per hour), and 18 pinto abalone off the southwest corner of Santa Cruz Island (range: 0.63–22.5 abalone per hour). The National Park Service has collected density and/or ''recruitment'' data for all species of abalone from 1982 to present at annually monitored, permanent, long-term, KFM transects. Most of these transects are not placed in what would be considered good pinto abalone habitat. Although no pinto abalone were observed from 1982 to 1999, in 2000 pinto abalone were observed (Kushner personal communication) for the first time after nearly 2 decades of surveys. From 2000 to 2017 the KFM divers have observed six pinto abalone along density transects, 24 pinto abalone in ARMs, and 12 pinto abalone in natural habitat size frequency sampling (it is likely that several of these abalone were also counted on the density transects).

Limited SCUBA efforts conducted in habitats suitable for pinto abalone between 2008 and 2012 resulted in the identification of pinto abalone (Table 4). Very low densities, ranging from 0.0002 (San Miguel Island) to 0.0286 (San Diego County-South) pinto abalone/m^2, were reported (Taniguchi unpublished data). Size frequency distribution data from sites throughout Southern California (Table 4; 2000 to 2012) suggest that recent recruitment events occurred in at least two locations: Santa Cruz Island and San Diego County-South (Taniguchi unpublished data).

Recently, reports of pinto abalone in Southern California have been more common (Parnell personal communication, Hagey personal communication, Kushner personal communication, Witting personal communication). In most areas, reports range from a few individuals to up to several dozen. In San Diego, there appears to be a relatively large population of pinto abalone of all sizes (depths 15–40 m) that has been present since about 1997 (Parnell personal communication). An average of two to three pinto abalone have been observed on 100×2 m band transects and sometimes aggregations of 12 or more have been observed in one area (Parnell personal communication). Aggregations of up to three animals are occasionally observed and pairs are frequently observed. In addition, these observations consist of abalone of all sizes up to more than 137 mm, with small (20–40 mm) freshly dead recruit shells observed regularly. Surveys were conducted from 2014 to 2016 to characterize the demographics of pinto abalone populations in nearshore San Diego kelp beds (Bird, unpublished data;

TABLE 4.

Location, source, sampling year, estimated area surveyed, density estimates, and size ranges for pinto abalone observed during dive surveys in Southern California from 2000 to 2012 (Taniguchi, CDFW, unpublished data).

Location	Source	Year(s) for density estimate	Area for density estimate (m^2)	Density (pinto abalone/m^2)	Year(s) for size range	Size range (mm)
Santa Barbara County Mainland	PISCO	2012	720	0.0014	2010, 2012	120–150
San Miguel Island	CDFW	2008	19,680	0.0002	ND	ND
San Miguel Island	CINP	2014	720	0.0014	2014	63
Santa Rosa Island	CINP	2011, 2012, 2014	720	0.0014–0.0028	2001, 2011, 2012, 2014, 2017	70–90
Santa Cruz Island	CINP	2014	720	0.0014	2000–2007, 2011, 2012, 2015	11–135
Anacapa Island	PISCO	ND	ND	ND	2009	50
Los Angeles County Mainland (North)	PISCO	2012	720	0.0014	2012	130–140
Los Angeles County Mainland (South)	PISCO	2012	720	0.0014	2012	120
San Diego County (North)	PISCO	2012	720	0.0014	2012	140
San Diego County (South)	Button (unpublished data; see Fig. 34 in NMFS 2014)	2008	2,520	0.0286	2008	40–150

NMFS 2014). Preliminary data are summarized in the following paragraph.

Between June 2014 and December 2016, band transect dive surveys were conducted at 58 sites in San Diego, CA, covering a total of 27,471 m^2 (~7 acres) of hard-bottomed kelp forest habitat between 10 and 30 m depth. Surveys targeted what was considered to be an ideal pinto abalone habitat in the southern range of the species based on preliminary exploratory surveys. Divers identified and enumerated pinto abalone and collected size and location data for each abalone. A total of 92 pinto abalone were observed. Average density for all sites was 0.003 abalone/m^2 ranging from 0 to 0.03 abalone/m^2 per site and are far below threshold values identified for other species (Babcock & Keesing 1999, Neuman et al. 2010). Small animals were observed, however, with maximum shell lengths ranging from 13 to 151 mm (see Fig. 37, NMFS 2014). These surveys covered a very small portion of available kelp forest habitat in San Diego, but highlight the fact that this species is extremely patchy and that densities recorded on a per m^2 basis may not be the best metric for evaluating population viability (Bird, unpublished data; NMFS 2014). Further detailed analyses of these data will provide information on: (1) pinto abalone density and size frequency distributions with depth and latitude; (2) habitat characteristics associated with higher densities; and (3) aggregation sizes and spatial dispersion (Bird personal communication).

Population Trends in Mexico

There is little information from the southernmost portion of this species range in Baja California, Mexico. Reports of pinto abalone are common but often not confirmed. The most comprehensive survey conducted on abalone species for Baja California has no information on the distribution and abundance of pinto abalone (Guzman del Proo et al. 1976). A recent collaborative study designed to target green and pink abalone reported density data on pinto and white abalone in five areas surveyed off the El Rosario Coast, Baja California, Mexico in 2012 (Boch et al. 2014). Because of similarities in shell morphology and possible misidentification by observers (some surveyors had not been trained to identify abalone in the field), pinto and white abalone were grouped and referred to as a two-species complex (*Haliotis kamtschatkana assimilis*–*Haliotis sorenseni*) in this –study. The authors estimated that 75% of the observations in this group were *H. k. assimilis* (Boch, personal communication). A total of 178 *H. k. assimilis*–*H. sorenseni* were found on 24 transects each covering a 400 m^2 area between 11 and 25 m depth (Fig. 14; Boch et al. 2014). Taking into account that 75% of these were likely *H. k. assimilis*, the density estimate for *H. k. assimilis* was 0.0139/m^2. With the majority ranging in size from 40 to 180 mm, *H. k. assimilis*–*H. sorenseni* ranged from 40 to 240 mm SL (individuals >165 mm SL were likely to be *H. sorenseni*). Recent recruitment was evident in at least one area where the population consisted of primarily 40–80 mm SL animals (Fig. 15; Boch et al. 2014).

MORTALITY AND COMPETITION

Shepherd and Breen (1992) provide an excellent review of mortality in abalone and note that an understanding of

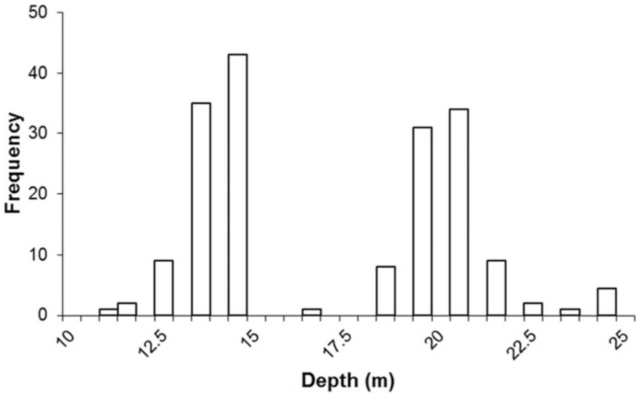

Figure 14. Depth distribution of *Haliotis kamtschatkana assimilis*/ *Haliotis sorenseni* found in El Rosario, Baja California, Mexico. Total depth distribution of $n = 178$ abalone found during the survey (note: all depths were not equally sampled). Adapted from Boch et al. 2014.

mortality rates in the three phases of natural abalone populations (larvae, juveniles, and adults) is essential to appropriate population modeling and management. As with most marine species, the various categories of mortality rates often are poorly known in abalone.

Mortalities of Abalone Larvae

Mortality rates of larval abalone are particularly difficult to estimate in the wild. The Shepherd and Breen (1992) review of abalone mortality lists a number of factors that may influence mortality rates and provides supporting references from the published literature. Among physical oceanographic factors, fluctuation beyond species-specific tolerance ranges in temperature and salinity have been identified as possible sources of mortality. As noted previously, larvae that are distributed to inappropriate habitats by ocean currents likely experience high mortality rates as a result of physiological stress, starvation or predation, although documentation of such processes is lacking. Many predators have the capacity for ingestion of abalone larvae in the marine water column, including planktivorous fishes and zooplankton, but rates of mortality associated with planktonic predation are also unknown. Given known fecundities for abalone and in consideration of the largest of estimated postmetamorphic abalone population sizes, it is likely that larval mortality rates in abalone are high even under optimal conditions.

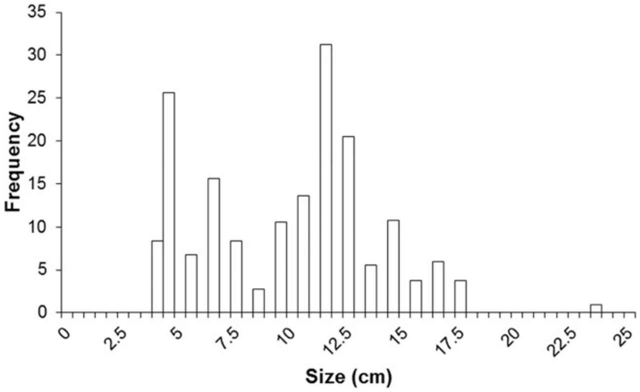

Figure 15. Size distribution of *Haliotis kamtschatkana assimilis*/ *Haliotis sorenseni* found in El Rosario, Baja California, Mexico. Total size (cm in diameter) distribution of $n = 178$ abalone found during the survey. Adapted from Boch et al. 2014.

Mortalities of Newly Metamorphosed Abalone

As described previously, available information indicates that crustose coralline algae are particularly important recruitment habitats for pinto abalone and for other abalone species. Shepherd and Breen (1992) indicate that little is known about predation on newly metamorphosed abalone in this habitat type. It is speculated that a broad range of small benthic invertebrates are capable of consuming new abalone recruits, including polychaetes, nematodes, polyclad flatworms, and anemones. Small, newly recruited abalone may also be susceptible to mortalities associated with disturbances such as substratum movement or disruption, deposition of sediment, influx of low-salinity waters associated with heavy rainfall and river discharge, and local seawater temperature anomalies associated with local weather events or larger scale oceanographic perturbations. There are no published data capable of supporting meaningful estimates of mortality rates in newly recruited juvenile abalone in response to any of the listed processes or events, or to any other form of predation or disturbance.

Mortalities of Small Cryptic Abalone (<40–50 mm SL)

As abalone grow, studying associated mortality processes become somewhat more tractable. Larger animals can be more easily located and monitored in the field as compared with smaller life history phases, allowing an understanding of certain types of mortality processes based on direct observation or tagging methods. In addition, the *postmortem* persistence of robust shells typical of larger animals often provides useful information on mortality sources and rates. Metamorphosed abalone that are still small enough to remain highly cryptic may require cracks or crevices of appropriate dimensions to provide refuge from foraging crabs (e.g., Shepherd 1973). Abalone in this size range (<40–50 mm) face predatory pressure from a number of other consumer species as well. Ault (1985), Shepherd and Breen (1992) and Hofmeister et al. 2018 list gastropods, octopuses, lobsters, sea stars, and fishes as predators capable of ingesting small abalone.

Despite the large number of identified predators on small cryptic abalone, studies that estimate mortality rates of pinto abalone in association with a predator species have not been identified. Cryptic abalone less than 40–50 mm in size may also suffer mortalities from the same range of physical disturbances listed previously for minute postmetamorphic abalone, although estimates of rates of mortality from such sources are not available for pinto abalone. In addition, abalone in this size range are large enough to experience illegal harvest by people (see ‘‘Mortalities associated with human removals’’ below).

Mortalities of Emergent Abalone (>40–50 mm SL)

Mortality patterns for large, emergent abalone (>40–50 mm in size) are reasonably well known for some species. Identified categories of mortality include predation (e.g., by sea otters, sea stars, crab, lobster, and fishes), variation in food supply (Neuman et al. 2017), physical disturbance, pollution, disease, and human removal (e.g., Shepherd & Breen 1992). Mortality from human removals, disease, and predation by sea otters are discussed in more detail below.

Competitive Interactions

Abalone and sea urchins often share habitats and food preferences. Tegner et al. (1992) noted that sea urchins can have negative effects on other herbivorous marine invertebrates in cases of limited food supply. Tegner and Levin (1982) evaluated possible competitive interactions of red abalone and red sea urchins (*Strongylocentrotus franciscanus*), finding minimal evidence for strong competition for food resources. Tegner (1989) noted that purple sea urchins (*Strongylocentrotus purpuratus*) may also be abundant in red abalone habitats in California and may be capable of destructive overgrazing of kelp populations at a level that could be nutritionally detrimental to abalone, potentially contributing to increased mortality rates.

Pinto abalone co-occur with both of these species of urchins and there are references in the literature to competition for food among them (DFO 2007). Empirical studies that examine the strength or nature of the competitive interactions that may exist between urchins and pinto abalone were not identified.

Mortalities Associated with Nonfishery Human Removals

Mortalities of pinto abalone associated with human removals fall into five major categories: (1) Subsistence harvest by indigenous peoples; (2) commercial harvest; (3) recreational harvest; (4) purposeful illegal harvest; and (5) accidental lethal injury. Categories (1–3) were discussed previously in ''Fishery-dependent Information''.

Purposeful illegal harvest (typically termed ''poaching'') has been a source of mortality for pinto abalone throughout their range since the establishment of harvesting regulations in Alaska, Canada, Washington, and California. In British Columbia, poaching has been identified as the main cause of declines in mature abalone densities, due to the species' tendency to aggregate in shallow, accessible waters; their high market value; and the remoteness of coastal areas where pinto abalone occur, which can be difficult to patrol. Poachers appear to target larger abalone (mean size of poached abalone was 115.1 mm, ±0.41 SE; Lessard personal communication; cited in COSEWIC 2009), which tend to be more fecund. Thus, removing individuals may result in the remaining animals being too far apart for successful spawning and also having less reproductive potential. Estimates of annual adult mortality rates for the period before 2007 were 0.32 for Haida Gwaii (Hankewich et al. 2008) and 0.33 for the CC (Hankewich & Lessard 2008). Both exceeded the estimated natural mortality rate of 0.15–0.2 (Breen 1986) and the estimated annual mortality rate of 0.25 that is believed to be sustainable (Zhang et al. 2007). More recent surveys in 2011 and 2012 indicate a decline in annual mortality (estimated at 0.26 for Haida Gwaii and 0.25 for the CC), primarily attributed to a reduction in poaching pressure (Lessard personal communication).

There is no evidence indicating illegal harvest is currently occurring in Washington, although several cases of illegal harvest and laundering of pinto abalone product were investigated in the late 1980s (directly linked to the legal commercial dive fisheries for sea urchins and sea cucumbers). Periodic cases of illegal sport harvest were reported after the 1994 fishery closure. It is generally believed that current populations no longer exist at commercially viable quantities, and the risk (effort) versus reward deters poaching. Washington Department of Fish and Wildlife enforcement covers the entire coast and includes at sea monitoring of commercial and recreational fisheries and periodic commercial buyer and market emphasis patrols. Outreach and education efforts to diving communities are also conducted. Despite these efforts, Vadopalas and Watson (2013) identified poaching as a major threat to abalone in Washington. Although there is the potential for illegal take of pinto abalone in other areas, records of enforcement cases or evidence for poaching were not identified.

Accidental injury is also a potential source of mortality for pinto abalone, particularly when fisheries harvest was occurring. This was identified as an issue in Washington, where WDFW eventually required abalone harvesters to carry a 4-inch caliper so that animals could be measured and determined to be of legal size before being removed from the substrate. Data assessing the effects of accidental lethal injury to pinto abalone either in the past or presently were not identified.

Mortalities Associated with Disease

No infectious diseases affecting wild pinto abalone have been reported in Alaska, Washington, or California. Two abalone pathogens have been reported in British Columbia, Canada (Bower 2010). Several catastrophic diseases have been reported in wild and cultured abalone worldwide, illustrating the importance of infectious disease to abalone populations (Bower 1987a, 1987b, 1987c, 1989, 2000, 2003, OIE, World Organization for Animal Health 2012, Crosson et al. 2014). Several abalone diseases, some previously unknown, have emerged in recent years highlighting the need for health examinations before animal movement to reduce the risk of pathogen introduction with animal movements (OIE, World Organization for Animal Health 2012).

Diseases affecting pinto abalone in British Columbia include the labyrinthulid protist, *Labyrinthuloides haliotidis*, which caused high losses of young farmed abalone <5 mm in SL during the early 1980s (Bower 1987a, 1987b, 1987c, 2000). In 1991, the renal coccidian *Margolisiella* (= *Pseudoklossia*) *haliotis* (Friedman et al. 1995, Desser & Bower 1997) was introduced into barrel culture in Bamfield, British Columbia with imports of red abalone from California. In 2005, one broodstock that had been collected in Bamfield and held in captivity for over a year had heavy renal (kidney) coccidian infections. The current distribution of this parasite in British Columbia is not known. The coccidian appears nonpathogenic to adult pinto abalone (Friedman et al. 1993).

Four significant abalone diseases have emerged over the past several decades and include withering syndrome (WS), ganglioneuritis (and the related amyotrophia), vibriosis, and shell deformities (sabellidosis). Withering syndrome is a rickettsial disease caused by ''*Candidatus Xenohaliotis californiensis*,'' a rickettsiales-like organism (RLO; Friedman et al. 2000). The disease has caused mortality in abalone populations ranging from 0% to 100% (see review by Friedman et al. 2014). Elevated temperature plays an important role in WS-RLO transmission and disease development (Crosson et al. 2014). Pinto abalone are highly susceptible to WS as evidenced by recent studies of

Crosson and Friedman (in press) during which 100% of the exposed pinto abalone died, whereas only 50% of exposed pink and 55% of exposed red abalone died. In addition, more rapid transmission of the WS-RLO and a lower thermal threshold for development of clinical WS was observed in pinto abalone relative to red and pink abalone. In the early 1990s, California abalone farms became infested with a sabellid polychaete, *Terebrasabella heterouncinata* (Kuris & Culver 1999), that an abalone farmer unintentionally introduced along with abalone from South Africa. Although the worms appear benign in natural populations, they slowly disfigure and weaken the shell of farmed California and South African (*Haliotis midae*) abalone. Before its introduction into California, this sabellid polychaete was unknown (Fitzhugh & Rouse 1999).

Mortalities Associated with Foraging Sea Otters

Sea otters (*Enhydra lutris*) ranged historically throughout the North Pacific Rim from northern Japan to the Pacific coast of Baja California, Mexico (Kenyon 1969), encompassing more than the entire geographic range of pinto abalone. Sea otters were hunted to near extinction during the maritime fur trade period from 1743 until the late 1960s (Kenyon 1969, Riedman & Estes 1990, VanBlaricom 2015). The sea otter population in California was listed as ''threatened'' in 1977 pursuant to the ESA of 1973 as amended (42 FR 2965, January 14, 1977) and as ''depleted'' pursuant to the Marine Mammal Protection Act of 1972 as amended (16 U.S.C. 1361 *et seq*).

When otter harvest rates began declining early in the 20th century, sea otters began recovering (either naturally or through reintroductions) in a number of areas throughout their range, a process leading to conflicts with nearshore, marine shellfish fisheries, particularly for abalones, sea urchins, clams, and crabs, whose populations had expanded although sea otter numbers and distribution contracted (Estes & VanBlaricom 1985). Conflicts have been particularly acute in Prince William Sound and in the Alexander Archipelago of Alaska, off Vancouver Island in British Columbia, near Cape Flattery and Neah Bay on the coast of Washington, and at several locations along the mainland CC of California. Within the geographic range of pinto abalone, contemporary sea otter populations are present; (1) in southeast Alaska (numbering ~25,712 individuals in 2014), (2) in two discrete population segments off British Columbia, (3) from Cape Flattery to Destruction Island off Washington (3-y average of 1,753 individuals in 2017; Jeffries et al. 2018), (4) from Half Moon Bay to near Point Conception on the mainland California coast (3-y average of 3,104 individuals in 2017; Tinker & Hatfield 2017), and (5) at San Nicolas Island off southern California (3-y average of 82 individuals in 2017; Tinker & Hatfield 2017). Sea otter populations remain regionally extirpated in the marine waters of Oregon and Baja California, Mexico.

Through the application of innovative behavioral attributes and the use of stones as hammers during foraging, sea otters are capable of imposing significant constraints on abalone abundances, size frequencies, and microhabitat distributions. Sea otters are known to feed on pinto abalone, but the quantitative ecological strength of the interaction has not been directly investigated and remains poorly understood. Quantitative information on impacts at the population level has been obtained only for red and black abalone. Available data suggest that predation on red abalone by sea otters typically reduces red abalone density by ~90% (Ebert 1968, Lowry & Pearse 1973, Cooper et al. 1977, Hines & Pearse 1982, Ostfeld 1982, Wendell 1994, Fanshawe et al. 2003) and eliminates viable commercial and recreational harvests of red abalone (Wild & Ames 1974, Estes & VanBlaricom 1985). Effects of sea otter predation on black abalone remain equivocal at present (e.g., VanBlaricom 1993, Crosson et al. 2014).

Although interactions between sea otters and pinto abalone are not well understood, a few recent studies and preliminary results from abalone monitoring provide insight in British Columbia. Based on modeling of sea otter and pinto abalone population dynamics, illegal harvest rates, and a series of conservation objectives for abalone restoration, Chadés et al. (2012) concluded that achievement of abalone population recovery in the presence of illegal abalone harvests and sea otter predation is unlikely. Reduction of poaching rate by 50% from current levels was effective in initiating modeled abalone population restoration; more so with a hypothetical removal of sea otters. It was suggested that simultaneous accomplishment of recovery goals for pinto abalone and sea otters in British Columbia waters will be difficult but possible if illegal harvest rates can be reduced substantially. In fact, in 2011 the density of mature abalone was greatest at one area along the CC where sea otters were present, exceeding the long-term objective of 1 abalone per m^2 (Lessard, personal communication), providing evidence that the corecovery of otters and abalone may be possible on a broader scale.

The sea otter population in northern and southern southeast Alaska is growing between 12% and 14% annually (USFWS 2014). The dramatic increase in sea otter numbers and range has caused significant concern about benthic invertebrate fisheries in southeast Alaska. Observations by divers for the ADF&G on the outer coast of southeast Alaska suggest that sea otters preferentially select red sea urchins and pinto abalone as prey when foraging in rocky subtidal habitats (Rumble & Hebert 2011). Otter predation on abalone is not considered the major factor in the decline of abalone in the 1980s because sea otter expansion occurred after high harvests by commercial fishing (Woodby et al. 2000). The increase in the otter population and predation on abalone will likely affect the recovery of abalone and the potential for a future commercial fishery (Rumble & Hebert 2011). Current research efforts include joint projects by the University of Alaska and the US Fish and Wildlife Service to examine the effects of sea otter recolonization in Southeast Alaska (https://www.sfos.uaf.edu/people/profile.php?uid=2080, http://seagrant.uaf.edu/research/projects/10/otter/, and http://project.nprb.org/view.jsp?id=6af6bc0e-6f46-4795-95aa-aaff483505b9). In a recent study of Northern sea otter diet in southeast Alaska, only three pinto abalone were observed in more than 6,100 otter-foraging dives (699 foraging bouts), suggesting a low presence of abalone in this region (Hoyt 2015).

Concern about interactions of sea otters and shellfisheries often involves significant economic issues, in many cases leading to polarizing controversy among scientists, managers, and stakeholders (VanBlaricom et al. 2013, Carswell et al. 2015). In some cases the intensity of concern about the effects of predation by sea otters on shellfish, and the associated hyperbole, leads to failure of management authorities to consider and manage other natural and anthropogenic sources of shellfish mortality that may be of equal or higher importance to shellfish conservation. Abalone fisheries have been historically characterized by initially high harvests, almost invariably unsustainable and followed, over varying time scales, by collapse. Failures of abalone fisheries have been ascribed to poor fishery management practices, illegal harvests, habitat degradation, changes in regulatory frameworks, diseases, and consumption by natural predators including sea otters (Estes & VanBlaricom, 1985, Neuman et al. 2010). In no case has local extinction of any abalone population or species in the northeastern Pacific been documented as a result of predation by sea otters.

Impacts of Climate Change

Laboratory and field research have found that many organisms, especially calcifiers, respond negatively to ocean acidification (Hall-Spencer et al. 2008, Kroeker et al. 2010, Busch & McElhany 2016). These changes include decreases in growth and calcification, dissolution of shells and hard parts, reductions in survival, altered gene and protein expression and physiological impairment, including negative impacts to acid-base balance and energy metabolism (Pörtner 2008, Kroeker et al. 2010, Parker et al. 2013, Swezey et al. 2017). Responses to acidification vary greatly among species and even within populations of a single species (Kroeker et al. 2010, Parker et al. 2011, Kelly et al. 2013). Some primary producers (e.g., seagrasses, macroalgae, and phytoplankton with low-efficiency CO_2 concentrating mechanisms) may in fact exhibit increased growth under ocean acidification through carbon fertilization effects (Palacios & Zimmerman 2007, Swanson & Fox 2007, Reinfelder 2011). The fast rate of change in ocean carbon chemistry that is predicted over the next century raises the question of whether species that are negatively impacted by ocean acidification possess sufficient standing level genetic variation to acclimatize and/or adapt under increasingly acidic ocean conditions. Geologically induced ocean acidification events are contemporaneous with extinction events in many taxa during the history of the Earth (Kump et al. 2009, Clarkson et al. 2015). Given that the current rate of CO_2 emissions exceeds the rate of emissions observed in many past extinction events, this would suggest that ocean acidification may overwhelm evolutionary processes and reorganize ecosystems (Hautmann et al. 2008, Kump et al. 2009, Pelejero et al. 2010). Marine communities near natural CO_2 vents are significantly different from neighboring communities that are not exposed to elevated CO_2 levels (Hall-Spencer et al. 2008, Fabricius et al. 2011, Kroeker et al. 2011, Enochs et al. 2015). Furthermore, ecosystem modeling suggests that trophic interactions can magnify the direct impacts of ocean acidification on sensitive species by indirectly affecting species to which they are trophically linked (Busch et al. 2013, Gaylord et al. 2015, Marshall et al. 2017). Although the ocean acidification literature continues to grow, in large part, there is a lack of understanding regarding how the vast majority of economically and ecologically important species in the California Current Ecosystem will respond to ocean acidification and how acidification will affect species interactions (but see Busch & McElhany 2016, Marshall et al. 2017). It is known, however, that production of Pacific oyster (*Crassostrea gigas*) larvae in Pacific Northwest shellfish hatcheries has been negatively affected by changes in ocean carbon chemistry that have already occurred (Barton et al. 2012), and that marine molluscs show an elevated sensitivity to the effects of ocean acidification when broadly compared with other marine invertebrate groups (Kroeker et al. 2013).

At present, atmospheric CO_2 levels exceed 410 ppm and are expected to climb throughout the century to approximately 800 ppm if emissions are not kept in check (ECOS 2011). These increasing emissions will directly reduce surface ocean pH, and will affect upwelling activity along the West Coast of North America. Upwelling refers to surface wind driven transport of deep ocean, low pH waters into coastal habitats. As part of this process, low pH waters with pCO_2 values in excess of 1,200 µm have recently been observed in regions throughout coastal California and Oregon (Feely et al. 2008). The incidence of these upwelling events is expected to increase as atmospheric CO_2 concentrations continue to rise, with seasonal onsets of pervasive low pH conditions expected along many regions of the California coasts by 2050 (Hauri et al. 2009, Gruber et al. 2012).

Effects of ocean acidification on early life stages of pinto abalone are beginning to be investigated. Laboratory studies on pinto abalone indicate that reduced larval survival and shell abnormalities or decreased shell size occur at 800 and 1,800 ppm (ppm) CO_2, compared with 400 ppm CO_2 (Crim et al. 2011). Friedman (unpublished data) has also found reduced pinto abalone larval survival at elevated pCO_2 and is currently studying the synergistic effects of increased pCO_2, varying temperature, and disease on early life stages of pinto abalone. Low pH has resulted in slower growth and lower survival of greenlip and blacklip abalone (Harris et al. 1999; reviewed in Morash & Alter 2015), as well as impacts larval size and survival in the European abalone (Wessel et al., in review) indicating that other abalone species are highly sensitive to decreases in ocean pH.

Other climate change–related effects that may impact pinto abalone include increased water temperatures and decreased salinity (due to freshwater intrusions). Bouma (2007) studied cultured pinto abalone and found that laboratory rearing temperatures of 11°C, 16°C, and 21°C did not affect postlarval survival. Larvae tolerated temperatures of 12°C–21°C, with mortality at 24°C. Captive adult pinto abalone in Alaska showed no behavioral abnormalities at 2°C–24°C but high mortality at 0.5°C and 26.5°C. Low salinity intrusions from freshwater inputs to Puget Sound and the SJA may also have negative effects on pinto abalone recruitment. In laboratory experiments, low salinity water reduced larval and postlarval survival of pinto abalone (Bouma 2007).

Overall, some information is available regarding the potential effects of ocean acidification, elevated water temperatures, and low salinity intrusions on pinto abalone. There is currently a high degree of uncertainty regarding the risk these threats pose because there are a limited number of studies involving pinto abalone and spatial variability in predictions regarding climate change impacts.

EXTINCTION RISK ASSESSMENT METHODS AND RESULTS

Measuring or documenting extinction risk factors for pinto abalone is limited and the available information is often not quantitative. Therefore, in assessing risk, both qualitative and quantitative information were considered and the assessment was based on the approaches used in previous abalone status reviews to organize and summarize the professional judgment of the coauthors. Conclusions about the risk of extinction faced by pinto abalone were made under the assumption that present conditions will continue into the future (recognizing that existing trends in factors affecting populations and natural demographic and environmental variability are inherent features of ''present conditions'').

Threats Assessment Methods

The potential role that different stressors have had on wild populations of pinto abalone in the United States, Canada, and Mexico was examined (Fig. 16). Different stressors and specific sources for those stressors (terms defined below) were identified. For each stressor/source combination, a qualitative rating for the following criteria were assigned: the scope and severity of the stressor/source; the level of data available to assess the stressor/source (data sufficiency); and the time frame over which the stressor/source is affecting the species (threat persistence). The time frame of future threat persistence varied for each source/stressor combination, but generally was between 30 and 100 y. Each of these criteria, the qualitative rating levels (e.g., low, medium, high, very high), and the numerical scores associated with each rating level are defined below and in more detail in the status review report (NMFS 2014).

The terms used in Figure 16 are defined as follows (including definitions for the criteria, the qualitative rating levels used for the criteria, and the associated numerical scores for each qualitative rating level):

Stressors: The specific condition that causes stress to the organisms (e.g., elevated temperature, predation, disease).

Sources: Natural or anthropogenic processes that create stressful conditions for organisms (e.g., climate change, sea otter recovery, pathogen introduction).

Scope

The proportion of the species' population that has been or can reasonably be expected to be affected by the threat. This refers to the spatial extent of the impact of the threat, not of the threat itself. Specific ratings for this criterion are defined as follows:

Very high: The effects of the threat are or are likely to be very widespread or pervasive in scope, affecting most of the species' population (the area over which >90% of the species' population occurs). Numerical score = 1.

High: The effects of the threat are or are likely to be widespread in scope, affecting much of the species' population (the area over which 51%–90% of the species' population occurs). Numerical score = 0.7.

Medium: The effects of the threat are or are likely to be restricted in scope, affecting the species across some of the species' population (the area over which 11%–50% of the species' population occurs). Numerical score = 0.3.

Low: The effects of the threat are or are likely to be very narrow in scope, affecting the species across a small proportion of the species' population (the area over which 0%–10% of the species' population occurs). Numerical score = 0.05.

Severity

Within the scope, the level of damage to the habitat or population from the threat that has been or can reasonably be expected given the continuation of current circumstances and trends. Specific ratings for this criterion are defined as follows:

Very High: Within the scope, the threat has or is likely to destroy or eliminate the habitat or affected population, or reduce its habitat or affected population by greater than 90%. Numerical score = 1.

High: Within the scope, the threat has or is likely to seriously degrade habitat or reduce the affected population by 51%–90%. Numerical score = 0.7.

Medium: Within the scope, the threat has or is likely to moderately degrade habitat or reduce the affected population by 11%–50%. Numerical score = 0.3.

Low: Within the scope, the threat has or is likely to only slightly degrade habitat or reduce the affected population by 0%–10%. Numerical score = 0.05.

Data Sufficiency

The quality of data available on which to assign a rating. In other words, are the available data sufficient to support a credible threats assessment? Specific ratings for this criterion are defined as follows:

High: An abundance of data is available for the threat and its effects on the species, and the reviewer has no reservations in reaching a rating decision. Numerical score = 1.

Medium: Data are available for the threat and its effects on the species, and a rating can be assigned but additional data are desired. Numerical score = 0.75.

Low: Ratings are based on expert opinion, based on biological concepts or inferences from data or information on other species or areas. Numerical score = 0.5.

Threat Persistence: Historical, Current, and/or Future

The relative time frame(s) over which the threats and/or their impacts were/are/will occur. Historical (H) threats and/or impacts are those that occurred in the past and may or may not be occurring presently. Current (C) threats and/or impacts are those occurring presently. Future (F) threats and/or impacts are those likely to affect the species. Numerical scores were assigned as follows:

Historical threat (H) = 0
Future threat (F) = 0.5
Current threat (HCF, CF, or HC) = 1

Overall Rating for Sources

The overall score/rating for each source, based on the mean (across all coauthors) of the product of scope, severity, data sufficiency, and threat persistence. The products (numerical scores) were converted into categorical ratings as follows:

Very High (VH): Mean score = 0.9–1.0
High (H): Mean score = 0.49 to <0.9
Medium (M): Mean score = 0.07 to <0.49
Low (L): Mean score = <0.07

Coefficient of Variation (CV) Values

Coefficient of variation values were calculated as a measure of the variation in scores among coauthors. The CV values were converted into categorical ratings as follows:

Very High (VH): CV value >2
High (H): CV value >1–2

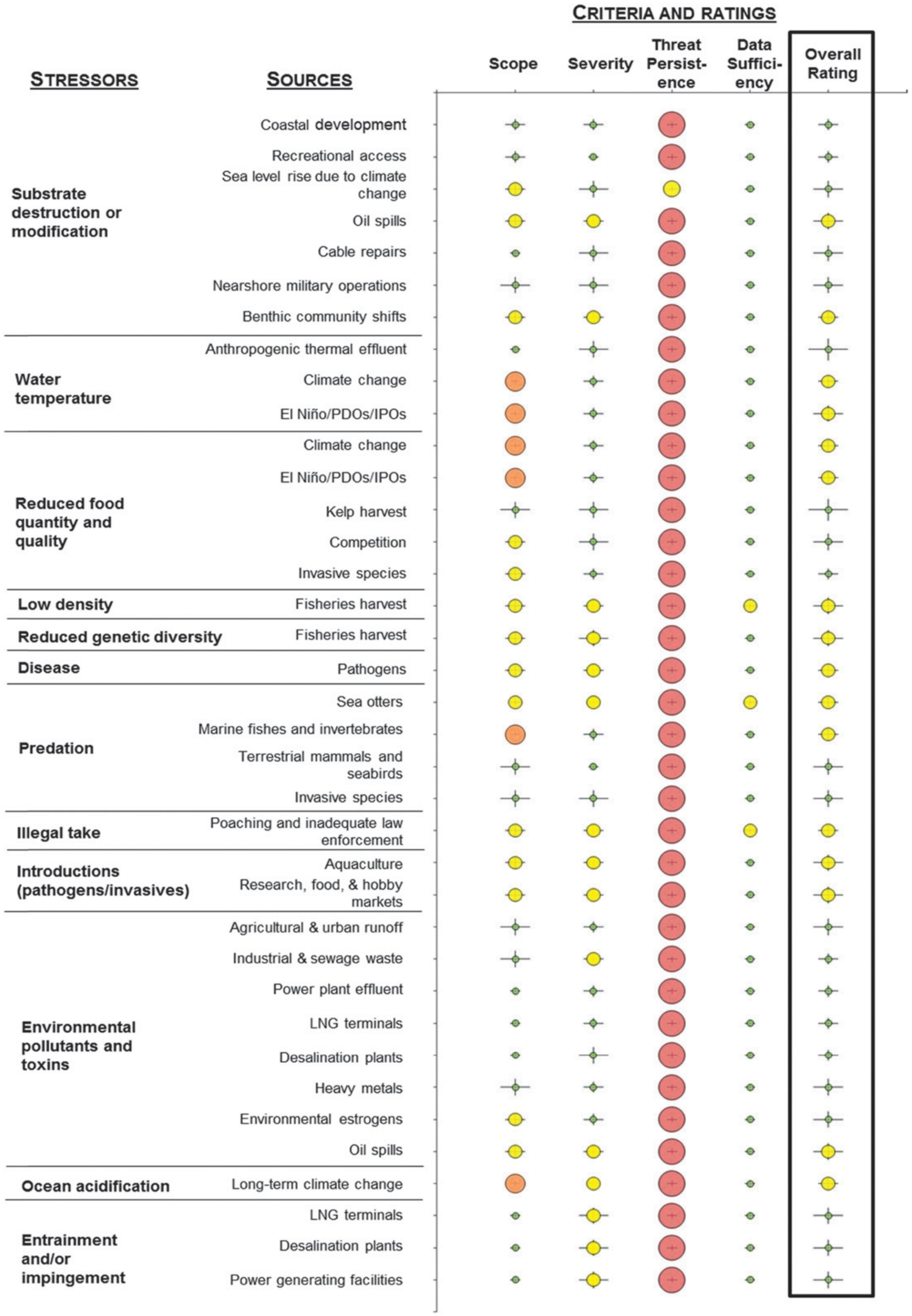

Figure 16. Threats assessment for the wild population of pinto abalone in the United States, Mexico, and Canada. The scope and severity of the stressor and the data sufficiency were rated as VH = very high, H = high, M = medium, or L = low. Threat persistence refers to the relative time frame(s) over which the threats were/are/will occur: H = historical, C = current, and F = future. The overall threat level for each source was rated as VH = very high, ◯; H = high, ◯; M = medium, ○; or L = low, ∘; based on the scores for scope, severity, threat persistence, and data sufficiency. The CV was calculated for the overall source rating and rated as: VH = very high, ┼; H = high, ┼; M = medium, +; or L = low, +.

Medium (M): CV value >0.5–1
Low (L): CV value = 0–0.5

Threats Assessment Results

Risks Related to Overutilization

The current low densities of pinto abalone populations were identified as the threat of greatest concern for the species. In a variety of locations throughout the pinto abalone range, local densities are estimated to be below the critical threshold densities identified for successful spawning and recruitment in other abalone species (0.15–0.34/m^2; Babcock & Keesing 1999, Neuman et al. 2010). Fisheries harvest of pinto abalone for commercial and recreational purposes (i.e., before the fishery closures) has contributed to this predicament. Harvest of pinto abalone is currently prohibited throughout the coast except in Alaska (i.e., for personal use and subsistence harvest) and Mexico. Data on harvest levels and the impacts on pinto abalone are not available for Alaska and Mexico, but the best available information indicates that these fisheries are not contributing substantially to overutilization of the species (ADF&G comments to NMFS on January 17, 2014). In Mexico, green and pink abalone are the focus of the abalone fishery, with other abalone species (including pinto abalone) making up only 1% of the abalone fishery (Boch et al. 2014). In Alaska, the daily limits for personal use and subsistence harvest were reduced in 2012 from 50 to 5 abalone per day. The data to assess how this harvest level would affect pinto abalone populations in Alaska does not exist; however, ADF&G believes that personal use and subsistence harvest of pinto abalone is currently low (ADF&G comments to NMFS on January 17, 2014). The average subsistence harvest of pinto abalone ranged from 350–382 abalone per household in 1972 but decreased to 3–9 abalone per household in 1997 (Bowers et al. 2011). In recent interviews, local residents have indicated to ADF&G that they are not participating in the personal use fishery due to the lack of abalone (Bowers et al. 2011). Based on this information, it is likely that personal use and subsistence harvest of pinto abalone in Alaska is low. Monitoring of harvest levels and pinto abalone populations is needed to obtain a better understanding of the impacts of these fisheries in Alaska and Mexico.

The effects of past fisheries harvest on local densities still persist today throughout the species' range. Past harvest levels, particularly in commercial fisheries in Alaska and British Columbia, were not sustainable and reduced densities to very low or nonexistent levels. Some populations (e.g., at the SJA in Washington) appear to be experiencing recruitment failure. In these cases, pinto abalone densities may be too low for successful spawning and recruitment. Evidence of recent recruitment exists in several areas throughout the species' range (British Columbia, California, Mexico), indicating that densities at those locations remain high enough to support reproduction and recruitment. In addition, Seamone and Boulding (2011) have demonstrated that successful reproduction and recruitment can occur despite very low densities because of the aggregative behavior of pinto abalone during the spawning season. These observations show that there is much more to learn about the species' population dynamics and the factors influencing successful reproduction and recruitment. For example, mean adult densities may not be an appropriate metric for predicting reproductive and recruitment success because they may not adequately represent the patchy distribution of abalone within an area. Fine-scale spatial distribution patterns (e.g., aggregations) may be more important for reproductive and recruitment success than the overall density of adults in an area.

Reduced genetic diversity is a potential risk associated with low densities. Withler et al. (2001) provide the only published assessment of population structure in pinto abalone and found high levels of genetic variation in pinto abalone populations sampled at 18 sites throughout coastal British Columbia and at one site in Sitka Sound, AK. Unfortunately, research on populations throughout the remainder of the species' range has not been conducted, and thus the Withler et al. (2001) study represents the best available information. Based on this, a moderate degree of concern was expressed, but most co-authors felt that the species' genetic diversity likely remains high.

Risks Related to Disease and Predation

Disease has been identified as a major threat to abalone species worldwide, with four significant abalone diseases emerging over the past several decades (WS, ganglioneuritis, vibriosis, and shell deformities). Pinto abalone are likely susceptible to all of these diseases and have been confirmed to be highly susceptible to WS, a disease that has resulted in significant declines in black abalone populations throughout southern California. No infectious diseases affecting wild pinto abalone have been reported in Alaska, Washington, or California, but two pinto abalone pathogens have been reported in British Columbia. To date, no outbreaks have been observed in wild populations and there is no evidence indicating that disease has been a major source of mortality in the recent past or currently. Multiple sources and pathways exist for pathogens or invasive species to be introduced into wild pinto abalone populations, including aquaculture facilities and the movement of abalone (e.g., import, transfer) for aquaculture, research, and food/hobby markets. Great care is needed to closely monitor and manage these sources and pathways, to protect wild populations from potentially devastating pathogens and invasive species.

Abalone face nonanthropogenic predatory pressure from a number of consumer species such as gastropods, octopuses, lobsters, sea stars, fishes, and sea otters (Ault 1985, Estes & VanBlaricom 1985, Shepherd & Breen 1992, Hofmeister et al. 2018). In the past, pinto abalone populations may have been better able to absorb losses due to predation without compromising viability. Specifically, predation by sea otters has been raised as a potentially significant factor in the continued decline and/or lack of recovery of pinto abalone populations in areas where the two species overlap. Sea otters were hunted to near extinction in the mid-1700s to 1800s but have begun to recover in recent decades with protection from the North Pacific Fur Seal Convention of 1911, the Marine Mammal Protection Act, and reintroductions in Southeast Alaska, British Columbia, and Washington in the late 1960s. Sea otter populations in these areas have been expanding in both abundance and distribution in recent years and are likely to continue to expand as the populations grow. Without a better understanding of how expansion and/or contraction rates of sea otter and pinto abalone populations will proceed and a better understanding of the predation pressure that sea otters exert on pinto abalone, it is very difficult to predict the long-term, population-level impacts that sea otters may have on fragile or recovering pinto abalone populations. The best available information at this time supports the view that sea otters and abalone can sustainably coexist, though abalone populations are not likely to reach the high abundances achieved in the recent past when sea otters were locally extirpated.

Risks Related to Climate Change

Ocean acidification is a concern particularly for early life stages of pinto abalone because of the potential for reduced larval survival and shell growth, as well as increased shell abnormalities. The impacts of ocean acidification can be patchy in space and time and may develop slowly. Some studies have shown deleterious effects of ocean acidification on abalone and other shell-forming species. Data availability is low, especially regarding how ocean acidification may affect the species throughout its range, given variability in local conditions throughout the coast, natural variation in ocean pH, species adaptability, and projections of future carbon dioxide emissions. Ocean acidification could also affect settlement habitat by affecting the growth of crustose coralline algae but the effects to pinto abalone are unclear. For example, McCoy (2013) and McCoy and Ragazzola (2014) found morphological changes (e.g., reduced thickness or density) in crustose coralline algal species in response to ocean acidification, with responses varying by species. Johnson et al. (2014) found that crustose coralline algal species exposed to varying carbon dioxide levels may be acclimatized to ocean acidification, with species-specific variation in the responses. North Pacific waters, including the California Current Ecosystem, have relatively low seawater pH values due to a variety of natural oceanographic processes (Feely et al. 2004, Feely et al. 2008, Feely et al. 2009, Hauri et al. 2009), and this may make crustose coralline algal species within the range of pinto abalone better able to adapt to the effects of ocean acidification. At this time it is unclear how ocean acidification may affect the chemical cues that are believed to attract pinto abalone to settle on crustose coralline algae.

Other climate phenomena that drive changes in long-term climate indices such as the El Niño/Southern Oscillation (ENSOs), Pacific Decadal Oscillations (PDOs), and the Interdecadal Pacific Oscillation (IPOs), as well as sea level rise, have the potential to cause water temperature increases and reduced food quantity and quality, but the certainty in how these factors will affect pinto abalone is low. For example, increased water temperatures associated with climate change may be widespread throughout the U.S. West Coast, though the latest climate report suggests that water temperature impacts will be least felt in the Pacific Northwest (Mote et al. 2014). Increased water temperatures could affect the health and range of pinto abalone, particularly at the northern and southern extremes of the species' range. Pinto abalone have a wide temperature tolerance and may be able to adapt to changing temperatures over time, such as by seeking depth refuges. It is also not clear how climate change drivers behind climate indices such as ENSO, PDO, and IPO may affect food quantity and quality for pinto abalone. Sea level rise may result in loss of suitable habitat in a preferred depth range because of increased erosion, turbidity, and siltation; however, the effects on pinto abalone are uncertain because they typically occupy subtidal habitats throughout much of their range and this would not change under rising sea level conditions.

Risks Related to Illegal Activities

Poaching has been a source of mortality for pinto abalone throughout their range since the establishment of harvesting regulations by the States and Canada. The problem of poaching clearly persists in some regions along the coast, particularly in remote parts of British Columbia. Existing regulatory mechanisms, and outreach and education programs, have effectively reduced the risks posed by illegal take in British Columbia in the past 5 y, as indicated by increases in local densities at survey sites. In other regions along the coast, poaching is recognized as a historical and future risk, but specific information on current levels of poaching is lacking. Although regulatory measures have been established, continued efforts to enforce the regulations and monitor their effectiveness are needed to protect the species from this threat.

Other Risks

Oil spill and response activities were also identified as a concern for pinto abalone, for both the potential effects on habitat and on abalone themselves. These effects would be of particular concern where the species occurs in intertidal and shallower waters (e.g., Alaska and British Columbia). The threat of an oil spill is greater in areas with higher ship traffic and human development. If a spill were to occur, acute effects could be very damaging in the localized area of the spill. There is little information available on the effects of oil spills on subtidal habitats where pinto abalone tend to occur throughout most of their range, as well as little information available on the effects of oil on abalone. Coastal development, recreational access, cable repairs, nearshore military operations, and benthic community shifts, may have impacts on pinto abalone and their habitat, but if they occur infrequently and have a narrow geographic scope then the overall risk to pinto abalone throughout its range is likely low.

Environmental pollutants and toxins are likely present in areas where pinto abalone have occurred and still do occur, but evidence suggesting causal and/or indirect negative effects on pinto abalone due to exposure to pollutants or toxins is lacking. In addition, very little is known regarding entrainment and/or impingement risks posed by coastal facilities. Direct effects would be localized and focused on larval stages. Despite uncertainties due to lack of data, the overall risk that environmental pollutants and toxins and entrainment/impingement pose on the species throughout its range is probably low given their limited geographic scope.

Summary

Overall, most of the stressors and sources posed low threat levels to pinto abalone and none were believed to pose high or very high threats to pinto abalone. Several stressors and sources were believed to pose moderate levels of risk to pinto abalone. Among these, low densities as a result of past fisheries harvest of pinto abalone, the potential threats by ocean acidification, and illegal take due to poaching and inadequate law enforcement, posed the highest risks. The potential for reduced genetic diversity as a consequence of low population densities and abundances and the potential for predation (particularly by sea otters) to further reduce local densities were also identified as threats of greater concern relative to other threats. Finally, oil spills and disease outbreaks (through the spread of pathogens) were highlighted as highly uncertain risks to pinto abalone that need to be addressed through careful planning, monitoring, and management.

Demographic Risk Assessment Methods

To evaluate demographic risks to the species, the collective condition of individual populations at the ''species'' level were analyzed according to four criteria: abundance, growth rate/productivity, spatial structure/connectivity, and diversity. These four general viability criteria, reviewed in McElhany et al. (2000), reflect concepts that are well founded in conservation biology, are generally applicable to a wide variety of species, and describe demographic risks that individually and collectively provide strong indicators of extinction risk. In the following paragraphs, these demographic risk criteria as they relate to a species' extinction risk are summarized.

Evaluating the extinction risk of a species includes considering the available information concerning the abundance, growth rate/productivity, spatial structure/connectivity, and diversity of a species and assessing whether demographic risks are such that it is nearing extinction or likely to become so in the foreseeable future. A species at very low levels of abundance and with few populations will be less tolerant to environmental variation, catastrophic events, genetic processes, demographic stochasticity, ecological interactions, and other processes (e.g., Gilpin & Soule 1986, Meffe & Carroll 1994, Caughley & Gunn 1996). A rate of productivity that is unstable or declining over a long period of time may reflect a variety of causes but indicates poor resiliency to future environmental variability or change (e.g., Lande 1993, Foley 1997, Middleton & Nisbet 1997). For species at low levels of abundance, in particular, declining or highly variable productivity confers a high level of extinction risk. A species that is not widely distributed across a variety of well-connected habitats will have a diminished capacity for recolonizing locally extirpated populations and is at increased risk of extinction due to environmental perturbations and catastrophic events (Schlosser & Angermeier 1995, Hanski & Gilpin 1997, Tilman & Lehman 1997, Cooper & Mangel 1999). A species that has lost locally adapted genetic and life history diversity may lack the raw resources necessary to endure short- and long-term environmental changes (e.g., Groot & Margolis 1991, Wood 1995).

The demographic risk criteria described previously were evaluated based on the present species' status in the context of historical information, if available. Demographic extinction risk was evaluated by assessing the likelihood of a number of questions related to the four-risk criteria (abundance, growth rate/productivity, spatial structure/connectivity, and diversity). These questions, listed below, were taken from the document ''Guidance on Responding to Petitions and Conducting Status Reviews under the ESA'' released by NMFS Protected Resources on May 24, 2013.

Abundance Questions

1. Is the species' abundance so low that it is at risk of extinction due to environmental variation or anthropogenic perturbations (of the patterns and magnitudes observed in the past and expected in the future)?
2. Is the species' abundance so low, or variability in abundance so high, that it is at risk of extinction due to depensatory processes?
3. Is the species' abundance so low that its genetic diversity is at risk due to inbreeding depression, loss of genetic variants, or fixation of deleterious mutations?
4. Is a species' abundance so low that it is at risk because of demographic stochasticity?

Population Growth Rate Questions

1. Is a species' average population growth rate below replacement such that it is at risk of satisfying the abundance conditions described previously?
2. Is the species' average population growth rate below replacement such that it is unable to exploit requisite habitats/niches/etc. or at risk because of depensatory processes during any life history stage?
3. Does the species exhibit trends or shifts in demographic or reproductive traits that portend declines in per capita growth rate which pose risk of satisfying any of the preceding conditions?

Spatial Structure Questions

1. Are habitat patches being destroyed faster than they are naturally created such that the species is at risk of extinction due to environmental and anthropogenic perturbations or catastrophic events?
2. Are natural rates of dispersal among populations, metapopulations, or habitat patches so low that the species is at risk of extinction due to insufficient genetic exchange among populations, or an inability to find or exploit available resource patches?
3. Is the species at risk of extinction due to the loss of critical source populations, subpopulations, or habitat patches?

Diversity Questions

1. Is the species at risk of extinction due to a substantial change or loss of variation in life history traits, population demography, morphology, behavior, or genetic characteristics?
2. Is the species at risk of extinction because natural processes of dispersal, migration, and/or gene flow among populations have been significantly altered?
3. Is the species at risk of extinction because natural processes that cause ecological variation have been significantly altered?

After reviewing all relevant biological information for the species, each author assessed the questions (aforementioned) using a voting process that is described in more detail in NMFS (2014).

Demographic Risk Assessment Results

Depensatory processes due to low and/or highly variable abundance or low population growth rate were a concern for pinto abalone in a number of locations (e.g., San Juan Islands and Alaska). Pinto abalone abundance and population growth have declined throughout the species' range, and, although there is some indication that recent recruitment has occurred in localized areas (e.g., Alaska, British Columbia, Mexico, and San Diego, Los Angeles, and Mendocino Counties, CA), the rate of population growth is unknown. Concern was expressed that population growth may not be occurring at a pace or extent sufficient to buffer against possible further declines due to processes happening over longer (e.g., PDO, IPO, climate change, and ocean acidification over decades; ENSO events over years) and/or uncertain time scales (e.g., cumulative oil spill impacts, poaching events, or harvest impacts).

Habitat destruction and loss of variation in life history traits, population demography, morphology, behavior, or genetic characteristics were of lower concern for pinto abalone, leading to the conclusion that spatial structure and diversity posed lower relative risk to the species throughout its range compared with abundance and population growth. The prevailing justification here was that other related species of abalone that experienced sharp declines in abundance (e.g., red, pink, black, and green abalone), have made remarkable recoveries in multiple locations over a period of roughly 2 decades (Richards & Whitaker 2012, VanBlaricom unpublished data, Vantuna Research Group unpublished data). Still, within the spatial structure category, there was concern, but high uncertainty, about whether pinto abalone are at risk due to the loss of critical source populations or subpopulations in many areas as a result of intense past fishing pressure.

Overall, low (i.e., for spatial structure and diversity) to moderate levels (i.e., for abundance and population growth) of concern and a high degree of uncertainty existed for most questions and demographic categories. Although this process helps to integrate and summarize a large amount of diverse information, there is no simple way to translate the results of this demographic risk assessment directly into a determination of overall extinction risk. This demographic risk assessment did not take into consideration the effects of past, present, and future threats on the persistence of the species into the foreseeable future. Thus, an additional assessment of overall extinction risk was undertaken to incorporate the results of both the threats assessment and demographic risk assessment.

OVERALL RISK DETERMINATION

The overall risk assessment considers demographic risks together with threats to evaluate the level of extinction risk faced by the species now and in the foreseeable future. Because data are not available to quantitatively assess the species' extinction risk (e.g., through development of a population viability model), an approach similar to what has been carried out in previous abalone status reviews was adopted, using a voting process to organize and summarize the professional judgment of the coauthors regarding the overall level of extinction risk to the species.

For the purpose of this extinction risk analysis, the term ''foreseeable future'' was defined as the timeframe over which threats can be predicted reliably and over which their impacts to the biological status of the species may be observed. Life history of pinto abalone and the availability of data regarding threats to the species were considered, leading to two definitions of the foreseeable future:

30 y: A time frame of 30 y represents approximately three generations for pinto abalone (McDougall et al. 2006, COSEWIC 2009). This time frame is consistent with what was used to define the foreseeable future in the black abalone status review report (NMFS 2009) and represents a reasonable time frame over which threats can be predicted reliably and impacts to the species' status would be observable.

100 y: Although a longer time frame introduces more uncertainty, a time frame greater than 30 y was deemed appropriate to adequately consider the effects of longer-term threats, such as climate change and ocean acidification. A foreseeable future of 100 y was selected as a reasonable time frame over which some information exists to predict impacts of longer-term threats. In addition, this time frame was used to examine the risk of multiple coral species that are threatened by climate change and ocean acidification (Brainard et al. 2011).

The assessment over a foreseeable future of 30 y represents the overall level of extinction risk faced by pinto abalone given the species' current status and threats that can be predicted over the next 30 y. This assessment recognizes that predictions regarding longer-term threats are highly uncertain and can change within this time frame, as can the quality of information on the species and its environment. The assessment over a foreseeable future of 100 y represents the overall level of extinction risk faced by the species in light of current predictions regarding long-term threats (e.g., climate change, ocean acidification) and expectations regarding oceanographic regime shifts as indicated by ocean indices (e.g., ENSOs, PDOs/IPOs).

Overall Risk Assessment Methods

The following five levels of extinction risk, defined below, were used to assess the overall extinction risk to the species now and in the foreseeable future (defined as 30 and 100 y):

No or very low risk: It is unlikely that this species is at risk of extinction due to projected threats or trends in abundance, productivity, spatial structure, or diversity.

Low risk: It is unlikely that this species is at risk of extinction due to trends in abundance, productivity, spatial structure, or diversity; however, current threats (or projected threats) may (or will) alter those trends but not yet by enough to cause the species to be influenced by stochastic or depensatory processes.

Moderate risk: The species exhibits a trajectory indicating that it is approaching a level of abundance, productivity, spatial structure, and/or diversity that places its current or future persistence in question. A species may be at moderate risk of extinction due to declining trends in abundance, productivity, spatial structure, or diversity and current or projected threats that inhibit the reversal of these trends.

High risk: The species is at or near a level of abundance, productivity, spatial structure, and/or diversity that places its current or future persistence in question. Similarly, it faces clear and present threats that are likely to create such demographic risks.

Very high risk: The species is strongly influenced by stochastic or depensatory processes, facing current threats exacerbating the demographic risks and indicating imminent extinction.

To allow individuals to express their level of uncertainty in assessing the overall level of extinction risk facing the species, the ''likelihood point'' method, often referred to as the FEMAT method was adopted because it is a variation of a method used by scientific teams evaluating options under the Forest Plan (Forest Ecosystem Management: An Ecological, Economic, and Social Assessment Report of the Forest Ecosystem Management Assessment Team, or FEMAT) (FEMAT 1993). This approach has been used in previous status reviews (e.g., Pacific salmon, rockfish in Puget Sound, Pacific herring, black abalone, and scalloped hammerhead sharks) to structure thinking and express levels of uncertainty in assigning risk categories. For this approach, each coauthor distributed 10 ''likelihood points'' among the five levels of risk. The scores were then tallied and summarized.

Overall Risk Assessment Results

Over both time frames, likelihood points were distributed across all five extinction risk categories, with most likelihood points placed in the Low risk and Moderate risk categories and very few (1–2) points placed in the Very High risk category (Fig. 17). When considering a foreseeable future of 100 y, the distribution of likelihood points shifted from the No/Very Low and Low risk categories toward the Moderate and High risk categories, indicating greater concern regarding demographic risks and threats over the 100-y time frame compared with the 30-y time frame (Fig. 17).

OVERALL RISK CONCLUSION

Overall, pinto abalone have a Low to Moderate level of extinction risk now and in the foreseeable future (over both the 30- and 100-y time horizons). There is a high level of uncertainty regarding demographic factors, in particular regarding abundance and productivity levels. The main concerns highlighted by the coauthors include declines in abundance and uncertainty regarding whether current abundance and productivity levels are sufficient to support the persistence and recovery of the species in the face of continuing and potential future threats. Long-term declines have been observed in surveyed areas throughout the species range despite the protective measures put in place throughout the species range; in particular, harvest prohibitions in all areas except for Alaska (where subsistence and personal use fisheries still exist) and Mexico. There is concern that these declines may be putting the populations at the SJA at risk, because the populations appear to be experiencing recruitment failure. Throughout the rest of the species' range, densities remain low but the species persists and/or recent recruitment events have been observed and have even resulted in increased densities (of mature and all sizes of pinto abalone) at several index sites in British Columbia. Observed recruitment events indicate that demographic characteristics are sufficient to support reproduction in locations throughout the species range, but productivity is variable and occurring at undetermined rates. Observations suggest that abalone recruitment and populations, in general, are both temporally and spatially episodic. One of the main data gaps is the lack of historical data on the status of the species before fisheries harvest and before the removal of sea otters throughout most of the coast. Lacking this baseline for comparison further increases the uncertainty regarding how to interpret the limited demographic data available for the species, and points to the need for improved monitoring of pinto abalone populations throughout its range to adequately assess the species' status.

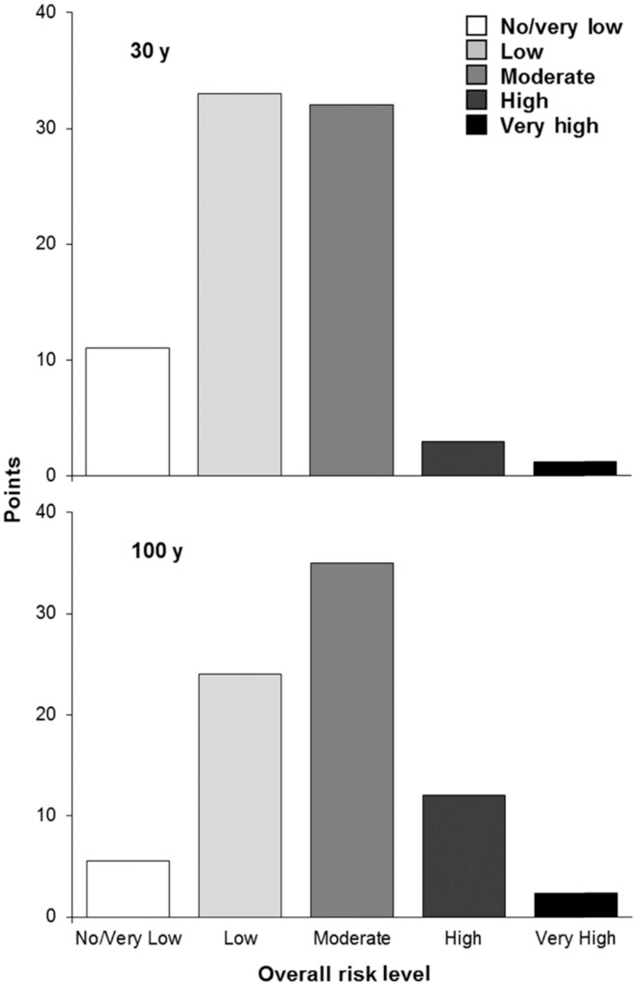

Figure 17. Overall level of extinction risk of pinto abalone based on qualitative risk assessments (threats and demographic). Two time frames (A) 30 y and (B) 100 y were assessed. The bars represent the total number of likelihood points that each voting author placed in each of the five extinction risk levels.

The main reason for the increase in likelihood points for the Moderate risk category versus the Low risk category when considering a foreseeable future of 100 y was the general perception by the coauthors that the species is likely to face more challenging conditions over the longer time frame, given the currently available predictions regarding climate change impacts, ocean acidification, and increasing sea otter populations. It was also recognized that there is more uncertainty associated with understanding and predicting these threats and their effects on the species over the longer time frame. Additional sources of uncertainty include: the lack of information regarding how naturally occurring events may affect the species into the future (e.g., IPOs, predation); the unpredictability of some threats (e.g., oil spills, climate change impacts); and the potential for pinto abalone to adapt to changing climate and conditions, as well as to recover from low abundances, which has been observed for other abalone species.

Assessing both the threats and demographic trends of pinto abalone across the species' range and over two time frames helped the authors consider the best available information for the species using different frameworks and led to greater confidence by decision makers to maintain pinto abalone as a NOAA species of concern (70 FR 77998). A dual approach to assessing risk is particularly important for rare species that lack baseline fishery-independent information. This approach also helped to highlight particular activities and particular areas where the demographic trends are of greater concern relative to others. Resource managers and researchers can use the information presented here to help strategically guide conservation actions and research during times when resources and funding are limited. Limitations in interpreting the future status of and prioritizing conservation activities for this and other NOAA species of concern could be overcome by conducting consistent and long-term demographic monitoring at key index sites throughout the species' range.

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