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Experimental rearing methods of pinto abalone (*Haliotis kamtschatkana*)
and their effect on outplant survival in Washington State

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Abstract

Experimental rearing methods of pinto abalone (*Haliotis kamtschatkana*) and their effect on outplant survival in Washington State

Bethany C. Stevick

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Associate Professor Carolyn S. Friedman
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Pinto abalone, *Haliotis kamtschatkana*, have been protected from fishing in Canada and Washington State since the 1990s, but populations continue to decline and exhibit recruitment failure. This species is a good candidate for a conservation aquaculture program and may benefit from enhancement efforts, such as outplants of hatchery juveniles. It is important to consider that hatchery conditions often differ from those in the natural environment and that the behavior of cultured abalone may reduce survival in the wild. This study investigates if modifying the rearing environment in a way that more closely resembles the natural environment can affect outplant success of pinto abalone. We reared juvenile pinto abalone using conventional and habitat-enriched methods for 7 months and found that survival proportion was higher in habitat-enriched tanks (analysis of deviance, $F = 21.87$, $df = 1,6$, $P < 0.01$) and mean shell length was larger in conventional tanks at the end of the experiment (Student's t -test, $P < 0.05$). In September 2009 we outplanted these differentially reared abalone to three discrete sites in the Strait of Juan de Fuca, Washington and conducted SCUBA visual surveys using noninvasive

methods at 24 hours, 2 months, 7 months and 1 year post-release. In addition, an invasive survey was conducted 1 year post-release after the noninvasive survey, during which time divers attempted to recover all surviving abalone by overturning rocks and boulders at each site. No difference in survival with respect to treatment or site for all surveys was detected (analysis of deviance, $P > 0.05$). At the one year, invasive survey, a total of 6.6% of all abalone outplanted survived. Mean initial shell length of the one year survivors was significantly greater than the mean initial shell length of all animals at time of outplant (t -test assuming equal variances, $t = 1.96$, $df = 742$, $P < 0.0001$). For abalone conservation programs wanting to use outplanting as a conservation tool, we recommend implementing low density, conventional methods in the hatchery. Since size influenced outplant survival, producing healthy, larger juveniles from diverse family lines should be a focus.

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DEDICATION

To Mom, Dad and Frank.

Abalone Conservation Literature Review

Abalone are marine gastropod mollusks belonging to the family Haliotidae. There are 56 recognized species and 10 subspecies, with a worldwide distribution (Geiger 1999). They are found along the coasts of every continent with the exception of the Atlantic coast of South America, the Caribbean, and the East Coast of North America. Abalone are dioecious, broadcast spawners that aggregate during spawning (Breen & Adkins 1980). This behavior is thought to increase the chance of fertilization and ultimately increase reproductive success in many species (Sloan & Breen 1988; McShane 1995; Babcock & Keesing 1999). The lecithotrophic, planktonic phase includes a trochophore and veliger larva, is relatively short and is temperature dependent (Sloan & Breen 1988; Bouma 2007). Settlement is triggered by a chemical (gamma-aminobutyric acid or GABA) produced by *Lithothamnion* (Morse et al. 1979) and occurs preferentially in rocky subtidal areas between near-shore kelp forests and deeper habitats with crustose corraline algae (Sasaki & Shepherd 2001). Abalone have been traditional food for native cultures, have been the target species for fisheries and became an aquaculture species in the 1950s. In addition, abalone shells are used for jewelry and decoration.

Over recent decades, worldwide abalone stocks have declined due to a variety of factors including poaching (Daniels & Floren 1998; Campbell 2000; Tarr 2000), disease (Friedman et al. 1997; Bower 2000; VanBlaricom et al. 2009) and poor fishery management (Karpov et al. 2000). As a result of over-exploitation, white abalone, *Haliotis sorenseni*, populations dramatically declined in California (Hobday et al. 2000) and were the first marine

invertebrate to be protected by the U.S. Endangered Species Act of 1973 (ESA; 16 U.S.C. 1531 et seq.) as amended in 2005. More recently, populations of another California species, the black abalone, *H. cracherodii*, have declined due to fishing pressure and disease (Altstatt et al. 1996; Friedman et al. 2000; VanBlaricom et al. 2009; Neuman et al. 2010) and was added to the List of Endangered Species (per ESA) in 2009. In addition, three other declining NE Pacific species are US Federal Species of Concern, including pink (*H. corrugata*), green (*H. fulgens*) and the pinto or northern (*H. kamtschatkana*) abalone. Due to these declining trends in populations, many abalone stocks are closed to fishing and have become a focus of conservation efforts.

Abalone have a natural history that facilitates overfishing, which has been a problem in every abalone producing country worldwide (Rothaus et al. 2008). In general, abalone reside in shallow water in relatively predictable and easy to access locations, only move on a scale of tens of meters, have irregular recruitment, are long lived and have slow growth rates (Breen 1986; Hobday et al. 2000). Serial depletion of California abalone species occurred and all commercial harvesting of abalone was closed in California in 1997; however a recreational free-diving fishery exists for red abalone, *Haliotis rufescens*, in Northern California. The British Columbia commercial fishery for pinto abalone, *Haliotis kamtschatkana*, peaked in the 1970's at over 400 metric tons per year (Sloan & Breen 1988; Campbell 2000) and was closed in 1990 due to stock declines and conservation concerns (Jamieson 2001). After this closure, stock assessment surveys showed no signs of natural stock rebuilding and some populations further declined (Campbell 2000). A commercial fishery for pinto abalone never existed in Washington State; however a recreational fishery existed until 1994 when a

moratorium prohibited abalone take. Like Washington, Alaska currently does not have a commercial fishery for pinto abalone, but recreational, snorkel-only harvest is allowed.

Alaska had a commercial pinto abalone fishery for 25 years, with a peak annual catch of 172 metric tons during the 1979-1980 season. Abundance exhibited a downward trend in the late 1980s, and the fishery was closed in 1995 (Woodby et al. 2000). Australia has experienced declining populations in abalone as well. Greenlip abalone, *Haliotis laevis*, populations have experienced slow declines (Shepherd & Rodda 2001) and two populations collapsed during the 1990s after 30 years of fishing (Shepherd et al. 2001).

The pinto or northern abalone is the northernmost abalone species in the eastern Pacific and is the predominant abalone found in Washington and Alaska, USA and in British Columbia, Canada. Pinto abalone are generally patchily distributed along exposed and semi-exposed coasts at depths of 0-100 m, ranging from Sitka Island, Alaska to Baja California (Sloan & Breen 1988). They have been a target species for recreational and commercial fisheries and, in Canada, for subsistence harvest by First Nations groups. Generally, populations are in sharp decline or small (Rothaus et al. 2008), a phenomenon that has been attributed to excessive harvest and poor recruitment (Campbell 2000). Insufficient recruitment is most likely a result of low adult densities that decrease reproductive success due to low fertilization (Allee 1949). Washington state populations of pinto abalone have been monitored by the Washington Department of Fish and Wildlife (WDFW) since 1992. This ongoing study includes ten index sites in the San Juan Archipelago. During the survey period of 1992-2006 abundance decreased by 77% with a mean standard length increase of 10.4 mm; densities are thought to be below the threshold needed for successful fertilization

(Rothaus et al. 2008). These results indicate consistently low recruitment rates, a finding confirmed by separate juvenile abundance surveys (Bouma 2007). In British Columbia Canada, Fisheries and Oceans Canada (DFO) conducted surveys at index sites and found that abundances declined by more than 75% from 1978 to 1984 and remain at low levels (Breen & Adkins 1979, 1981; Winther et al. 1995; Campbell 2000). Furthermore, Zhang et al. (2007) used growth models to study the stock-recruitment relationships for British Columbia populations and reported that these populations are unproductive. Recognition of persistent population declines was followed by the listing of pinto abalone as a “threatened” species in Canada in April 1999 by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) and more recently as “endangered” in Canada in 2009, as a NOAA “species of concern” in Washington in 2004, and as “endangered” on the IUCN Red List in 2006.

Fisheries closures were used as the first step towards recovery efforts, yet populations were still in decline after the closures noted above. Recovery efforts for declining abalone populations worldwide have varied in strategy, but focus on a similar goal: to rebuild or enhance natural populations. Besides local fishery closures, rebuilding strategies have included transplantation of wild juvenile and adult abalone (Shepherd 1986; Henderson et al. 1988; Emmett & Jamieson 1989; Tegner 1992, 1993, 2000), larval seeding (Tong et al. 1987; Shepherd et al. 2000), marine reserves (Edgar & Barrett 1999; Wallace 1999; Davis 2000; Rogers-Bennett et al. 2002; Micheli et al. 2008), and outplants of hatchery reared juvenile abalone (Tegner & Butler 1989; McCormick et al. 1994; Seki & Taniguchi 2000; Shepherd et al. 2000).

Many transplants of adult wild abalone to sites previously known to be productive for abalone have been attempted in order to establish breeding populations and increase abundance. Since fertilization success depends on adult density (Babcock & Keesing 1999), it is sensible that aggregating wild brood stock would increase abundance in the long term. Abalone are broadcast spawners, and sexually mature adults display strong aggregative behavior during the spawning season (Shepherd 1986). In 1983, California Department of Fish and Game (CDFG) transplanted 517 adult pink abalone from San Clemente Island to Santa Catalina Island with the goal of creating a concentrated spawning stock (Henderson et al. 1988). Only 5% and 18% of the transplanted abalone were recovered as live and dead, respectively; abalone that were unaccounted for (77%) were attributed to illegal harvest. Emmett and Jamieson (1989) performed transplants of large (50-100 mm) pinto abalone from exposed sites (where abalone tend to exhibit slower growth) to two sites that had sheltered, productive abalone habitat. Persistence of the transplanted abalone was high (39% and 72%) after nine months at the two sites. In addition growth of the transplanted abalone was enhanced in the new environments. Evidence of local recruitment was observed upon transplantation of reproductively mature green abalone, *H. fulgens*, in California until poaching of the broodstock occurred (Tegner 1992, 1993, 2000). Campbell et al. (2003) found that pinto abalone from exposed “surf” areas mature at a smaller size than those from sheltered areas and suggested future studies to investigate transplanting “surf” abalone to more productive regions as a means of pinto abalone restoration.

Larval seeding (the release of hatchery abalone larvae onto wild habitat) is an enhancement strategy that shows promise as a restoration method. Although survival has generally been

low, this method avoids the high cost of juvenile abalone production in the hatchery. Tong et al. (1987) released 13 day old larvae of paua, *Haliotis iris*, to an exposed rocky shore in New Zealand and observed higher densities of 2 mm juveniles at the experimental sites relative to control sites. In South Australia, Schiel (1992) used enclosure tents to release larval greenlip abalone, *Haliotis laevigata*, onto natural reefs and found that only about 10% of the larvae settled. In another study using greenlip abalone, survival rates of post-larvae 49 days after release was about 0.5% and density-dependent mortality was observed (Preece et al. 1997). In 1994, blacklip, *Haliotis rubra*, and greenlip abalone larvae were released to sites in South Australia, with survival rates 6-9 days after settlement that varied between 0.02 and 7.8% (Shepherd et al. 2000). Advantages of larval seeding over other enhancement strategies are low rearing costs and the avoidance of selection that can occur in the hatchery. More research is needed in this area to determine if it is an effective method in the long term.

The use of marine refugia as sources of replenishment may be applied to abalone fisheries and could potentially compensate for recruitment overfishing (Dugan & Davis 1993). Recruitment overfishing describes the circumstance when an adult population is fished intensively and decreases so much that the population no longer has the capacity to replenish itself. Increased abundances within refugia have been detected for abalone species in California (Tegner 1989) and Australia (Shepherd 1990), although recruitment was either not assessed or were not enhanced in these areas. In a large (7 km of coastline) marine reserve, the mean size of blacklip abalone was greater within the reserve as compared to mean size at outside reference sites (Edgar & Barrett 1999). In addition, densities of both smaller and larger abalone increased within the marine reserve during the study period. Wallace (1999)

used pinto abalone as an indicator of marine reserve effectiveness and found that abalone populations and reproductive output were greater in sites that had enforced fishing closures. Even though larval dispersal dynamics are not well understood for this species, Wallace maintains that marine reserve establishment can be an effective tool for abalone conservation. It has been suggested that aggregating wild adults on refugia reefs may be one of few options for producing new generations of white, pink (*Haliotis corrugata*) and black abalones (Davis 2000). Rogers-Bennett et al. (2002) compared white abalone abundances in California inside protected and fished areas and found larger abalone and higher abalone abundances in protected sites over time as compared to the fished sites, even though the protected sites initially had lower abundances. Similar results were found by Micheli et al. (2008) in a study that examined red and black abalone abundance and size between protected areas and public access sites in central California. Although no increasing or decreasing trends in red abalone abundance were found, higher abundances were found within the protected areas. In addition, larger sizes of black abalone made up a greater proportion of the population in the reserve than in the public access sites. Combined, these data suggest abalone populations thrive in protected areas when fishery closures are maintained. However, it is not apparent that reserves are effective tools for restoring abalone to historical levels.

Outplants of hatchery reared abalone have been conducted worldwide (for review see (Tegner & Butler 1989; McCormick et al. 1994) and survival rates of the seeded abalone have varied greatly. The Japanese first pioneered outplanting hatchery seed to enhance their abalone fisheries. For almost 25 years beginning in 1950, over 4.5 million hatchery reared ezo abalone, *Haliotis discus hannai*, seed were released to Funaka Bay, Japan in order to create

a fishery in an area that previously did not have naturally occurring abalone populations (Saito 1979). After two years, survival was estimated to be 25-30%. In a later study, Saito (1984) reported recapture rates of 5-10% 2-3 years after planting hatchery reared ezo abalone with an average size of 24.8 mm. Seki and Taniguchi (2000) had similar success in a later study where 146,000 juvenile ezo abalone were outplanted during 1996-1998 to sites where wild abalone populations had been depleted by overfishing. They observed a survival rate of 26.7% at 2 years after outplant. It is difficult to compare Japanese recovery rates of seeding events to those of other countries, since the Japanese have continuous intensive fisheries for abalone predators such as octopuses, crustaceans and gastropods.

There have been several outplant attempts in California with a variety of abalone species. In order to enhance native populations of red abalone, Tegner and Butler (1985) placed over 600 hatchery raised red abalone at a size of 40-80 mm in a kelp forest in California and reported a recovery rate of about 1% with an average growth of 30 mm per year. Poor success of the outplant was attributed to intense predation pressure (Tegner & Butler 1985). McCormick et al. (1994) released green abalone (20-30 mm), *Halitotis fulgens*, from two different hatcheries, using a novel planting module. They observed 39% and 91% survival after 90 days for the two groups; shipping time from hatchery to outplant site affected survival. In Northern California, 50,000 juvenile red abalone (8 mm) were seeded at sites with and without red urchins, *Strongylocentrotus franciscanus*. After two years, recovery rates were less than 1% for all sites with higher rates at sites with sea urchins (Rogers-Bennett & Pearse 1998).

In addition to the activity in California, there have been many efforts to enhance declining abalone populations in the South Pacific region. In New Zealand, after one year, 9-32% of planted wild and hatchery juvenile paua survived and more wild juveniles survived than hatchery animals for all size classes (Schiel 1992). Larger scale studies by Schiel (1993) outplanted paua to eight different sites and observed survival rates ranging from 1.2% to 72.4%. He noted large variations in survival among sites and that those with the poorest survival rates had areas of sandy habitat prone to movement that could potentially increase abalone mortality. In 1994 and 1995, 7 month old blacklip abalone were outplanted to sites with 3 layer boulder habitat in Tasmania and exhibited a survival rate of 16.5% one year after outplant (Shepherd et al. 2000). This difference was attributed to handling stress and behavioral deficits in the presence of predators. In addition, juvenile greenlip abalone were seeded using artificial modules to seagrass habitat that had few predators and the mortality rate was lower than that calculated in the previous experiment (Shepherd et al. 2000).

As noted above, the results of restocking with hatchery raised individuals are usually poor or unsuccessful (Svåsand et al. 2000; Bell et al. 2006). Variation in the success of abalone outplants may be attributed to a variety of factors, including size, site (Schiel 1993; Shepherd et al. 2000), stress, and predation. During a pilot outplant of pinto abalone in Washington, USA, initial shell length was a good predictor of survival (D. Rothaus et al. unpublished data). An abalone outplanted at 25 mm had a 20% chance of survival, whereas a 41 mm abalone had a 75% chance of survival. In addition, Saito (1984) outplanted Ezo abalone (10-52 mm) and found that optimal survival occurred at initial shell lengths of 34-36 mm. In contrast to these studies however, size and survival were not correlated for outplanted red

abalone (40-80 mm) at Palos Verdes, California (Tegner & Butler 1985). Shifting rock, gravel and sand at sites affected outplanted juvenile *H. iris* survival by the burial of juvenile habitat (Sainsbury 1982; Schiel 1992, 1993). Concerning site selection, Shepherd et al. (2000) recommend choosing sites that exhibit preliminary success with seeded abalone and ones with historical abalone populations. It has been observed that when abalone are handled and become stressed, they produce mucous, which attracts predators (Tegner & Butler 1985). This observation and the potential for increased detection by predators are the impetus for initially protecting outplanted abalone with some kind of planting module that allows the abalone to acclimate and recover from the stress of transport. In addition to man, predators of abalone include sea otters, some species of seastars, octopus, crab and a variety of fish species fish (Mower & Shepherd 1988; Sloan & Breen 1988; Tegner & Butler 1989; Watson 2000). Predator control during outplant events may increase survival. An initial outplant study done in California showed lower mortality one month after predators were removed. A subsequent study using larger abalone (40-80 mm SL) and monthly predator control observed about 1% survival after one year. It was concluded that the predator control most likely resulted in fewer abalone were preyed upon, however such controls must continue frequently in order to increase survival in the long term (Tegner & Butler 1985, 1989).

Differences in behavior between wild and cultured abalone may also impact juvenile survivorship during an outplant event. During laboratory experiments using cultured and wild red abalone, differences in movement patterns were found with wild abalone moving and concealing themselves faster than the cultured abalone when placed on complex reef habitat (Schiel & Welden 1987). In addition, more cultured animals were consumed than wild

animals when exposed to different combinations of predators in the laboratory (Schiel & Welden 1987). Tegner and Butler (1989) found that habitat selection by cultured and wild green abalone differed in the laboratory. Cultured abalone changed habitat more often, were more dispersed among the habitat categories available in the laboratory, and were out in the open more often than the wild abalone.

It is clearly important to take steps necessary to minimize behavioral patterns of cultured juvenile abalone that are maladaptive following outplanting. A potential approach to such a goal is to alter the habitat in which animals are reared. With a goal of conservation, hatcheries should aim to simulate natural rearing conditions by providing complex habitat (cover, structure, substrate, etc.) to produce animals that are similar to their wild counterparts and that have more natural behaviors and higher survival (Flagg & Mahnken 2004). Using enriched rearing environments can affect behavior in some cultured fish species. Berejikian et al. (2000) reared juvenile steelhead, *Oncorhynchus mykiss*, in habitat-enriched tanks that featured in-water structures, underwater feeders and overhead cover and compared behaviors of fish raised in enriched tanks to those raised in more conventional tanks. They found that during laboratory experiments, steelhead juveniles reared in habitat-enriched tanks were socially dominant over same sized competitors reared in the conventional tanks. Experiments using Atlantic salmon, *Salmo salar*, parr reared in conventional hatchery conditions or enriched conditions (tanks contained plants, rocks, novel objects) found that foraging abilities were enhanced in fish that had been reared in the enriched environment and had been exposed to live prey (Brown et al. 2003). This study suggested that rearing fish under enriched-habitat conditions in addition to exposing them to live prey could increase post-

release survivorship of cultured animals. Furthermore, Lee and Berejikian (2008) made behavioral comparisons among steelhead fish reared in barren, structured and structurally variable environments and found that stable (not variable) structure increased subsequent exploratory behavior. These studies suggest that rearing fish in more natural settings may result in cultured animals that behave more similarly to their wild conspecifics, possibly facilitating the minimization of behaviors maladaptive in the context of survival rate in the wild.

That rearing habitat can affect behavior has now been demonstrated in pinto abalone. Straus and Friedman (2009) determined that habitat selection and predator avoidance behavior in pinto abalone differed between abalone reared in conventional tanks and those reared in habitat enriched tanks. However, it is unclear if observed behavioral differences translate to differential survival of abalone once they are outplanted into the wild. In order to increase success of outplants for the purpose of restoration of pinto abalone, maladaptive behavioral deviations between cultured and wild individuals must be minimized. Methods for rearing pinto abalone in a more natural setting must be developed further, and studies to determine if the new culture methods result in higher survivorship of outplanted individuals is crucial for successful abalone restoration via supplementation.

**Experimental rearing methods of pinto abalone (*Haliotis kamtschatkana*)
and their effect on outplant survival in Washington State**

Introduction

Over recent decades, worldwide abalone stocks have declined due to a variety of factors including poaching (Campbell 2000; Tarr 2000), disease (Friedman et al. 1997; Bower 2000; VanBlaricom et al. 2009; Neuman et al. 2010) and poor fishery management (Karpov et al. 2000). Declines have been observed in all eight abalone species that inhabit the northeast Pacific ocean, with many of them now protected and closed to fishing. For example, white abalone (*Haliotis sorenseni*) populations dramatically declined in California as a result of over-exploitation (Hobday et al. 2000) and were the first marine invertebrate protected by the U.S. Endangered Species Act of 1973 (ESA; 16 U.S.C. 1531 et seq.) as amended in 2005. More recently, populations of another California abalone species, the black abalone (*H. cracherodii*) have declined due to fishing pressure and disease (Altstatt et al. 1996; Friedman et al. 2000; VanBlaricom et al. 2009; Neuman et al. 2010) and was added to the List of Endangered Species (per ESA) in 2009. In addition, three other declining NE Pacific species are US Federal Species of Concern, including pink (*H. corrugata*), green (*H. fulgens*) and the pinto or northern (*H. kamtschatkana*) abalone. With increasing concern surrounding the status of abalone, many stocks have become a focus of conservation efforts.

Pinto abalone population declines have been attributed to excessive harvest and recruitment failure (Bouma 2007; Rothaus et al. 2008), possibly due to Allee effects, which are a phenomenon for small populations where reproduction and survival rates of individuals increase with increasing population density (Allee 1949). All fisheries for pinto abalone in the northeast Pacific are currently closed, except for a recreational, snorkel-only fishery in Alaska. Abundances in Washington state, USA during a survey period of 1992-2006 decreased by 77% with a mean standard length increase of 10.4 mm and densities thought to be below the threshold needed for successful fertilization (Rothaus et al. 2008). These results indicate consistently low recruitment rates, a finding confirmed by separate juvenile abundance surveys (Bouma 2007).

Fisheries closures were used as the first step towards recovery efforts for pinto abalone, yet populations were still in decline after these closures. Recognition of persistent declines was followed by the listing of pinto abalone as a NOAA “species of concern” in Washington State in 2004 and as “endangered” on the IUCN Red List in 2006 and in Canada in 2009. Recovery efforts for declining abalone populations worldwide have varied in strategy, but focus on a similar goal: to rebuild or enhance natural populations. Besides local fishery closures, rebuilding strategies have included transplantation of wild adults (Tegner 2000) or hatchery-reared juveniles (Seki & Taniguchi 2000; Shepherd et al. 2000; Tegner 2000), larval seeding (Shepherd et al. 2000; Tegner 2000), and marine reserves (Micheli et al. 2008; Davis 2000). In fact, outplants of hatchery reared abalone have been conducted worldwide (for review see (McCormick et al. 1994; Tegner & Butler 1989); however, survival rates of the seeded abalone have varied greatly (Tegner & Butler 1985; Schiel 1992; Davis 1995;

Rogers-Bennett & Pearse 1998; Dixon et al. 2006; Goodsell et al. 2006) after one year post-outplant. Recovery efforts that focus on outplanting seed or juvenile abalone to natural reef sites depend on aquaculture operations for the production of juveniles.

Restoration aquaculture is a tool that can be employed to supplement wild stocks and has been used for a variety of fish (Pikitch et al. 2005; Berejikian et al. 2000; Weber & Fausch 2005) and invertebrate (Wilbur et al. 2005; Carlsson et al. 2008; Doall et al. 2008) species. The results of restocking with hatchery raised individuals are usually poor or unsuccessful (Svåsand et al. 2000; Bell et al. 2006). Variation in success of abalone outplants may be contributed to a variety of factors, including size (Rothaus et al. unpublished data), juvenile shell color (McCormick et al. 1994), site habitat (Schiel 1993; Shepherd et al. 2000), stress caused by transport and handling (Schiel 1993; McCormick et al. 1994; Sweijd et al. 1998) and predation (McCormick et al. 1994). Differences in behavior between wild and cultured abalone may also impact juvenile survivorship during an outplant event. During laboratory experiments using cultured and wild abalone, differences in movement patterns (Schiel & Welden 1987), predation rates (Schiel & Welden 1987), and habitat selection (Tegner & Butler 1989) were found. It is clearly important to take steps necessary to minimize behavioral patterns of cultured juvenile abalone that are maladaptive following outplanting. A potential approach to such a goal is to alter the habitat in which animals are reared. Using enriched rearing environments (e.g. tanks that feature in-water structures, overhead cover, plants, rocks, or novel objects) can affect behavior in some cultured fish species (Berejikian et al. 2000; Brown et al. 2003; Lee & Berejikian 2008). These studies suggest that rearing fish in more natural settings may result in cultured animals that behave more similarly to

their wild conspecifics, possibly facilitating the minimization of behaviors maladaptive in the context of survival rate in the wild.

That rearing habitat can affect behavior has now been demonstrated in pinto abalone. Straus and Friedman (2009) determined that habitat selection and predator avoidance behavior in pinto abalone differed between abalone reared in conventional tanks and those reared in habitat enriched tanks. However, it is unclear if these behavioral differences translate to differences in survival of abalone once they are outplanted into the wild. In order to increase success of outplants for the purpose of restoration of pinto abalone, and maladaptive behavioral deviations between cultured and wild individuals must be minimized. In an effort to determine if rearing method affects abalone outplant survivorship, we reared pinto abalone in conventional and habitat enriched tanks (using methods similar to Straus & Friedman 2009)), prior to outplanting the animals onto natural, rocky reef sites. We hypothesized that the habitat enriched treatment would significantly influence outplant survivorship.

Methods

Husbandry

Adult pinto abalone broodstock, which were collected from Washington waters, were induced to spawn in fall 2007 at the National Oceanic and Atmospheric Administration (NOAA) field station in Mukilteo, Washington, USA. Nine families from single parent crosses were created. Two year old juveniles from these families were used for this

experiment. Animals were tagged with a two digit, colored bee-tag (plastic, 2 mm diameter applied to shell with cyanoacrylate) and were number coded to indicate family line (n = 3,200, shell length (mean \pm standard error) = 11.6 mm \pm 0.09 mm, weight (mean \pm standard error) = 0.25 g \pm 0.01 g). The newly tagged animals remained in their natal tanks for one and a half months. On January 31, 2009, animals (n = 2,664) were evenly distributed among eight experimental tanks (four conventional and four habitat-enriched). It was not possible to use all 3,200 tagged animals due to post-tagging mortalities and because we distributed the survivors so that each family made up the same proportion of the abalone population in each tank.

Eight circular 220 L fiberglass tanks were used in this study. Half of the tanks were used in each culture treatment: control (conventional) and experimental (habitat enriched). The conventional treatment included four tanks containing three half rounds of 6" diameter polyvinyl chloride (PVC) on the bottom and three hanging from the tank sides to provide shelter, increase surface area and provide easier access to food. This configuration is similar to conventional abalone aquaculture techniques used for pinto abalone and currently is used at the NOAA Mukilteo facility. In contrast, habitat was provided in the experimental (habitat enriched) tanks by the addition of rock and cobble encrusted with coralline algae collected from Freshwater Bay, Strait of Juan de Fuca, Washington, USA rocky reef habitats. These rocks and cobbles were submerged in freshwater for 10-15 minutes before being placed in the habitat-enriched tanks in order to remove potential predators. In both treatments, available substrate provided abalone habitat as well as an equivalent amount of surface area on which diatoms and benthic microalgal food for the abalone can grow. In addition, one red

sea urchin (*Strongylocentrotus franciscanus*) and two green sea urchins (*Strongylocentrotus droebachiensis*) collected from the San Juan Islands, Washington, USA were added to each habitat-enriched tank for the purpose of increasing potential nursery sites for the juvenile abalone. Sea urchins are common in wild habitat where abalone are found in Washington state, and the presence of urchins during rearing may provide important accommodation and familiarization to conditions in wild habitat. In addition, sea urchins are well adept at capturing drift algae, which may aid in the abalone's ability to find and consume food.

All eight tanks were supplied with filtered (25 μ m) flow-through seawater pumped from Puget Sound and natural and artificial lighting during daytime hours (lights were on a 12:12 cycle). Kelps (*Nereocystis* and *Laminaria*) and cultured algae (*Palmaria mollis* and *Chondracanthus exasperatus*) were supplied to all tanks as feed. All observed mortalities were recorded and measured to estimate survivorship and a subsample of live animals from each tank were measured every 1-2 months to track growth.

After seven months, each remaining animal was measured and tagged with an additional bee-tag to indicate treatment (conventional or habitat-enriched). A subsample of the animals (n = 60) were sacrificed for pathology screening using histology and molecular techniques following protocols established by the World Animal Health Organization (International Office of Epizootics 2006). The sub-sample was certified 100% free of identifiable pathogens. Abalone (n = 713, shell length (mean \pm standard error) = 24.6 mm \pm 0.3 mm) were sorted into three groups destined for different outplant sites. These groups were

assembled such that mean size and family and treatment proportions were statistically the same across the groups.

Outplant

Outplant sites were selected by using a combination of previously established experimental outplant sites in Freshwater Bay, Strait of Juan de Fuca, Washington, USA (used in a pilot outplant study) and sites that Washington Department of Fish and Wildlife (WDFW) maintains for long term ecological studies in the same area. Three sites (designated as West, Central and East sites) were chosen that were 25-60 m² in size, contained suitable stable rocky habitat and food resources, were spatially isolated from other rocky reef habitats to avoid immigration/emigration and had none or few existing abalone. Prior to the outplant event these sites were surveyed by SCUBA divers to ensure all criteria were met. Divers delineated these sites by marking each corner with a concrete pier block that had a PVC mast attached as well as orange flagging tape. A pre-outplant survey at each site was conducted to search for juvenile and adult abalone. One adult abalone was found at the East site (shell length (SL) = 124 mm) and two were found at the Central site (SL = 82.4 mm and 127.5 mm).

The day prior to outplant, juveniles were put into ‘abalone introduction modules’ (AIMs) at the NOAA Mukilteo facility. An AIM is a length (~12-18”) of 6” PVC capped at both ends with a double layer of Nynetex mesh secured by cable ties. It has been shown that such modules can protect newly outplanted abalone by providing an acclimation period to the

novel habitat before the abalone are released and exposed to potential predators (Goodsell et al. 2006). The AIMs with abalone inside were transported from the NOAA Mukilteo facility to a Port Angeles, Washington marina in a fish tote filled with aerated seawater. The AIMs were then transferred to coolers filled with seawater onboard the diving vessel for transport to the outplant site.

Divers placed the AIMs at the corresponding three sites and revisited the sites 24 hours later to release the abalone by removing the Nynetex mesh. Divers conducted noninvasive surveys at 24 hours, two months, seven months and one year post release to record tag information for every abalone observed at each site. The sites were divided into 1-2 m wide lanes using lead lines, and a two-person dive team was responsible for intensively surveying the site one lane at a time. One year post-outplant, dive teams also conducted an invasive survey where rocks and cobbles were carefully moved and turned over in an attempt to find all abalone at each site. Tag information and shell length of each animal were recorded and the animals were removed from the sites and placed in aquaria in Port Gamble, Washington.

Data analysis

Specific growth rates (SGR) were calculated according to Hopkins (Hopkins 1992) as:

$$SGR(\%/d) = \{[\ln(L_f/L_i)]/t\} * 100$$

where:

L_f = final shell length (mm),

L_i = initial shell length (mm),

t = time elapsed between both measurements (days).

Relative growth (RG) was calculated as:

$$RG = \ln(L_f/L_i)$$

where:

L_f = final shell length (mm),

L_i = initial shell length (mm).

Student's t -tests were used to detect differences in SGR between treatments during the rearing period and differences in RG between treatments during the outplant period.

Survivorship was calculated as:

$$S = n_t/n_0$$

where:

n_t = number of abalone at time t ,

n_0 = number of abalone at start of experimental period

Differences in survivorship between treatments during the rearing and outplant periods were determined by using an analysis of deviance on proportion data with quasibinomial errors (Crawley 2007). A quasibinomial error structure was chosen due to overdispersion.

Show factor was calculated as:

$$SF = n_n/n_i$$

where:

n_n = number of abalone sighted using noninvasive methods,

n_i = number of abalone sighted using invasive methods.

The one year survey show factor was compared to that for a previous pilot outplant in the same bay in 2007 using a X^2 test for equality of proportions with continuity correction.

A generalized linear model (glm) was used to fit outplant survival data (family = binomial, link = logit). Factors that were used to determine if they were good predictors of survival were site, treatment and initial size.

Results

Rearing period

Overall survival (mean \pm 95% confidence interval (CI)) was significantly greater in the habitat enriched tanks than in the conventional tanks during the rearing period (60.36% \pm 5.31% vs. 36.94% \pm 14.55%, respectively; analysis of deviance, $F = 21.87$, $df = 1,6$, $P < 0.01$; Fig. 1). Mean shell length differed significantly between rearing treatments at each measured timepoint (days 33, 69, 95, 160 and 200 of the rearing experiment) (Student's t -test, $P < 0.05$; Fig. 2). At day 200, shell lengths (mean \pm 95% CI) of abalone reared under conventional conditions were larger than those reared under habitat enriched conditions (25.5 mm \pm 0.96 mm vs. 21.7 mm \pm 0.72 mm, respectively). Mean specific growth rates (SGR) were not significantly different between treatments except at day 200 (Student's t -test, $df = 343$, $t = 4.80$, $P < 0.00001$; Table 1). At day 200, the SGR (mean \pm 95% CI) of conventionally reared abalone was greater than the mean SGR of abalone reared under habitat enriched conditions (0.13 % day⁻¹ \pm 0.03 % day⁻¹ vs. 0.03 % day⁻¹ \pm 0.03% day⁻¹, respectively).

Outplant period

After the one year invasive survey for all three sites combined, 12.6% of the total number of abalone outplanted were recovered as live and dead; 6.6% were recovered as alive and 6.1% were recovered as dead. The total proportion of live abalone found at all three sites combined were 0.35, 0.04, 0.03 and 0.02 for the 1 day, 2 months, 7 months and 1 year noninvasive, post outplant SCUBA surveys, respectively, and 0.07 for the 1 year invasive survey. No differences in the mean proportion of live abalone sighted between conventional and habitat

enriched treatments were observed during any of the four noninvasive, post outplant surveys (analysis of deviance, $P > 0.05$; Fig. 3) nor during the invasive, post outplant survey (analysis of deviance, $P > 0.05$).

Mean initial shell length of the one year survivors was significantly greater than the mean initial shell length of all animals at time of outplant (t -test assuming equal variances, $t = 1.96$, $df = 742$, $P < 0.0001$) and of recovered mortalities (t -test assuming equal variances, $t = 1.96$, $df = 67$, $P = 0.03$). Mean initial shell length of recovered mortalities was not significantly different than the mean initial shell length of all animals at time of outplant (t -test assuming equal variances, $t = 1.96$, $df = 745$, $P = 0.07$).

Average total growth of survivors at the one year survey was 21.4 mm (range: 19.0-24.2, 95% bias corrected CI). Recovered mortalities had an average total growth of 2.6 mm (range: 1.4-4.2 95% bias corrected CI), with 53% showing less than 1 mm growth and 19% showing greater than 5 mm growth. Mean relative growth of one year survivors was significantly greater than that for recovered mortalities, 0.52 (bootstrap 95% CI: 0.47-0.58) vs. 0.07 (bootstrap 95% CI: 0.04-0.10), respectively (t -test assuming equal variances, $t = 1.99$, $df = 62$, $P < 0.0001$; Fig. 7). No differences in relative growth between treatments were observed for one year survivors or for recovered mortalities (t -test assuming equal variances, $P > 0.05$; Fig. 8).

The show factor for the one year survey was 31%, which is statistically similar to the show factor of 32% for a previous 2007 pilot outplant (X^2 test for equality of proportions with continuity correction, $X^2 = 0$, $df = 1$, $P > 0.05$).

Treatment and site were not good predictors of survival, however initial shell length was a good predictor of survival (glm, family = binomial, link = logit, $X^2 = 20.03$, $df = 1$, $P < 0.001$). According to the model, an abalone that is outplanted at 20 mm has a predicted survival proportion of 0.026, at 30 mm it is 0.056, at 40 mm it is 0.12 and at 50 mm it is 0.23. This model is conservative in its predicted survival proportion since we overreported mortality in our dataset used to create this model. Of the 714 animals planted, 47 survived but only 33 of these had readable family tags. As we were unable to discern which of the 681 unrecovered abalone were the 14 animals with unreadable tags these animals were recorded as mortalities instead of survivors.

Discussion

This is the first study (to the best of our knowledge) to test the influence of enriched rearing methods on outplant survival for any aquatic species, fish or invertebrate. Given that it has been demonstrated that an enriched rearing environment can affect behavior of animals compared to wild conspecifics, it was imperative to test if these rearing methods could influence survival in the wild. The results of our study indicate that the rearing methods we employed had no differential effect on outplant survival for abalone, which is in contrast to our initial hypothesis.

We did find that initial size at time of outplant was a strong predictor of survival after one year. This was also found during a 2007 pilot study using pinto abalone at similar study sites in Freshwater Bay (D. Rothaus et al. unpublished data). De Waal and Cook (2001) reported that initial size was important to survival for outplanted *H. midae*. In addition, Saito (Saito 1984) outplanted Ezo abalone (10-52 mm) and found that optimal survival occurred at initial shell lengths of 34-36 mm. In contrast to these studies however, size and survival were not correlated for outplanted red abalone (40-80 mm) at Palos Verdes, California (Tegner & Butler 1985) suggesting that optimal outplant size may vary among species and locations and must be empirically assessed.

While our reported survival rate of 6.6% is definitely within the range of those previously reported for abalone outplants, it is not at the higher end of the observed range. Dixon et al. (2006) had 1-57% survival 9 months post outplant of seeded greenlip abalone at eight different sites, with the lowest survival sites having a high presence of predators. They also found during a pilot study that complex, two layer boulder habitats increased survival compared to one layer habitats. In another study, survival was as high as 82.4% for some sites seeded with *H. iris*; lowest survival rates occurred at sites that had sand movement, which can bury juvenile abalone habitat (Schiel 1993). Multi-tiered concrete blocks were successfully used as artificial habitat for outplanted *H. rufescens* which experienced 32% survival after one year (Davis 1995). While our sites did contain complex rocky habitat suitable for abalone, much of the available habitat was not multi-layered and some sites had large areas of sand between patches of rocky habitat. Thus site characteristics may have

contributed to the observed overall survival. However, we chose our sites in order to contain our outplanted abalone onsite, allowing for better estimation of total survival and ultimately, the influence of rearing method.

Many of the mortalities (empty shells) we recovered showed signs of crab predation, even during the later surveys. While 53% of the recovered mortalities showed little to no growth (< 1 mm), 19% showed greater than 5 mm of growth. Combined, these observations indicate that while a majority of recovered mortalities occurred shortly after introduction, mortality continued throughout the survey period. Only 6.1% of abalone outplanted were recovered as mortalities, which is not a good indication of total mortality that we assume to be 93.4%.

Since our sites were isolated rocky reefs surrounded by sand and we invasively surveyed at the one year mark, we are confident that we recovered nearly all of the survivors. We do not believe that abalone moved offsite, and can therefore assume that those not found during the surveys did not survive. Mortalities not accounted for could have been ingested by predators (fish) or the empty shells could have been displaced offsite by water movement.

Another interesting observation revealed in this study was the calculation of an abalone survey show factor. As previously reported, our noninvasive surveys found 31% of the number of abalone found during the subsequent invasive surveys. This show factor is statistically the same as the 32% show factor calculated for the 2007 pilot study (D. Rothaus et al. unpublished data). It is encouraging that these two show factors are nearly identical and that they might be used for future outplant surveys. Noninvasive surveys, which take less

time to conduct, pose less risk of injury to outplant survivors, and do not disturb site habitat. Additional assessment of show factor at a variety of sites is thus warranted.

We report differences in hatchery survival by rearing method, which was an unexpected result since Straus and Friedman (2009) reported no differences in survival between conventional and enriched tanks. We used slightly different conventional methods (half pipes of PVC instead of fiberglass wavy plates) and had more individuals per tank than used by Straus and Friedman (2009). Even though Straus and Friedman (2009) reported no difference in survival between rearing methods, the trend toward higher survival in enriched tanks started to appear towards the end of their experiment. Despite the fact that we had significantly higher survival in the enriched tanks, we outplanted the same number of abalone from each treatment, and the two groups had the same mean size. The difference in hatchery survival between treatment should not have affected the results of our outplant study.

We were not able to determine how family influenced survival. Family information was lost due to tag damage during both the rearing and outplant periods. We did notice that family seemed to influence survival in the hatchery, but since so many individuals had unknown family lines, we do not report this conclusively (see Appendix). Many individuals had unknown family lines at the end of the one year outplant study. It is well understood that genetics plays a strong role in growth and survival, and analysis of differential survival among families could have enhanced our understanding of the influence of genetics on outplant survival.

For abalone conservation programs wanting to use outplanting as a tool, we recommend implementing low density, conventional methods in the hatchery. While the conventional rearing method resulted in lower survival, it produced larger individuals on average and is an easier, less time intensive culture method. Since size was so influential for outplant survival in this study as well as previous ones, producing healthy, larger juveniles from diverse family lines should be a focus. We have demonstrated that outplanting juvenile pinto abalone using introduction modules with a 24 hour acclimatization period works: nearly 7% of planted animals survived after one year and exhibited growth. However, we need to continue to work towards increasing survival. A further examination of enriched methods such as the addition of predator exposure prior to outplant may help achieve the goal of increased success, and ultimately increased local abalone populations.

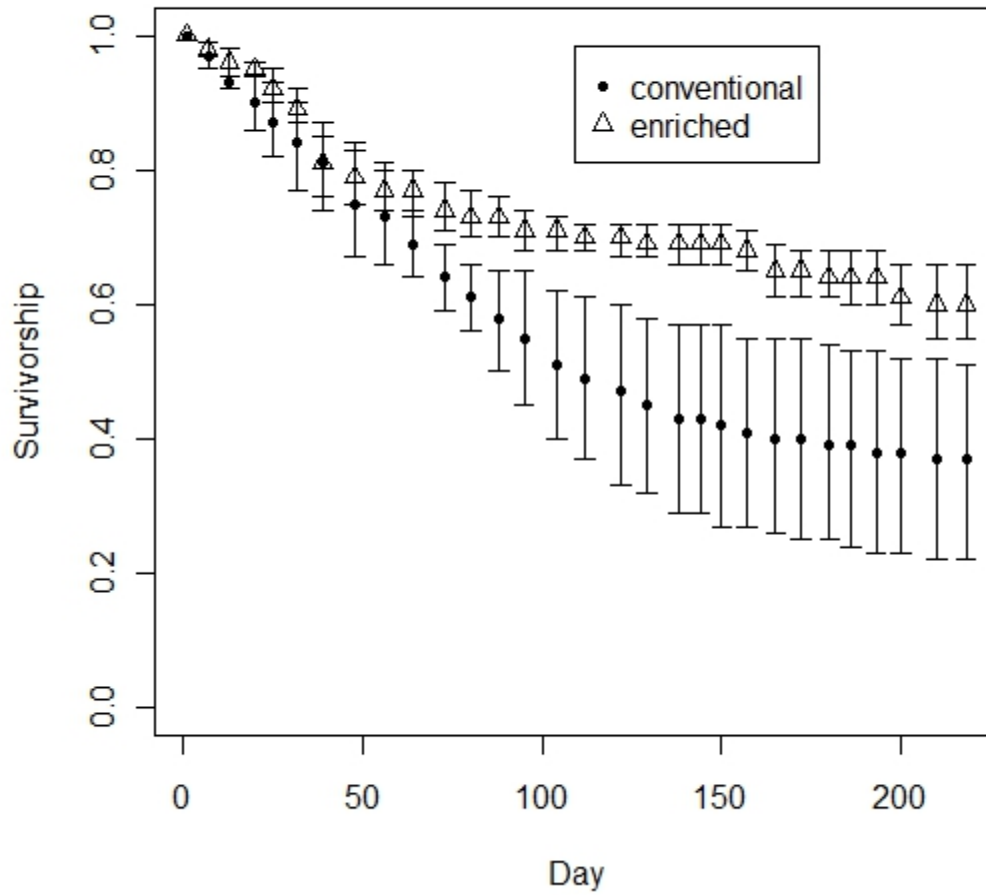


Figure 1. Survivorship (mean \pm 95% confidence interval) of juvenile pinto abalone (*Haliotis kamtschatkana*) reared in conventional and habitat enriched tanks over 218 days. There was a significant difference in overall survival between treatments (analysis of deviance, $F = 21.87$, $df = 1,6$, $P < 0.01$).

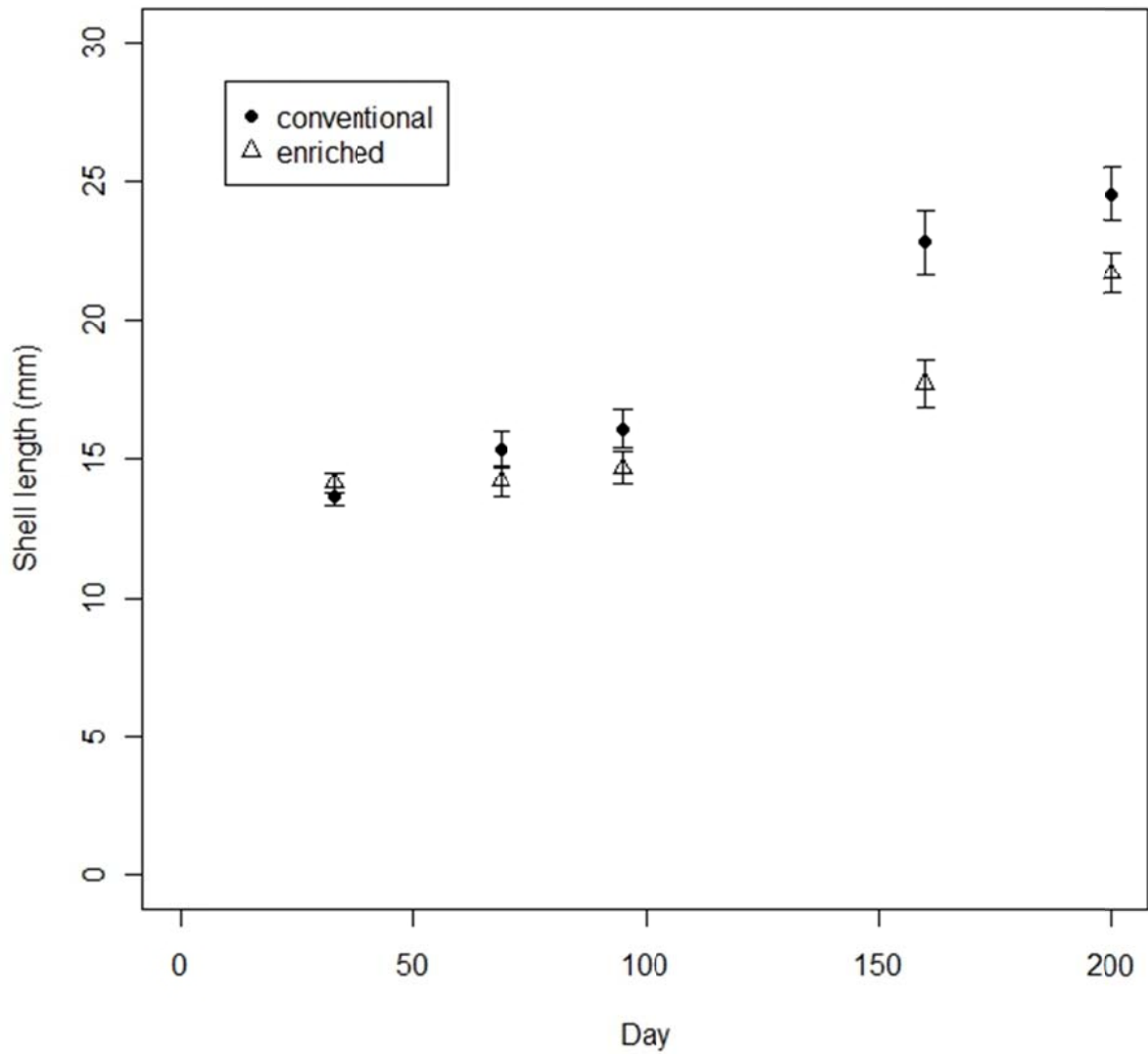


Figure 2. Shell length (mean \pm 95% confidence interval) of juvenile pinto abalone (*Haliotis kamtschatkana*) reared in conventional and habitat enriched tanks over 218 days. Differences in mean shell length between treatments were observed for all 5 measured timepoints (Student's *t*-test, $P < 0.05$).

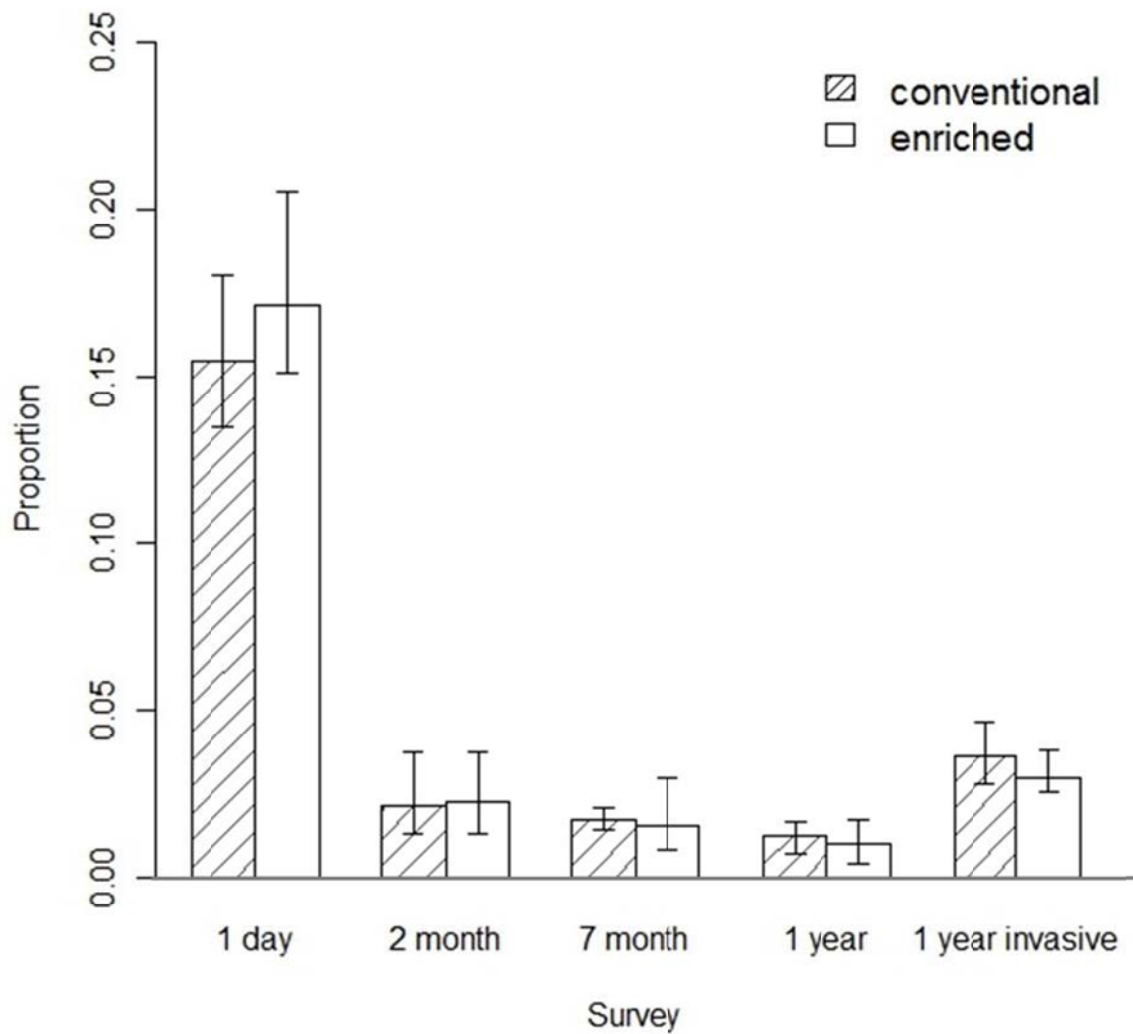


Figure 3. Mean proportion (\pm 95% bias corrected CI) of juvenile pinto abalone (*Haliotis kamtschatkana*) observed at our sites over a one year period using non-invasive post outplant SCUBA surveys. Abalone rearing methods prior to outplant are designated by cross hatched bars for conventional rearing and open bars for habitat enriched rearing. No differences in mean proportion between treatments were observed for all 4 surveys (analysis of deviance, $P > 0.05$).

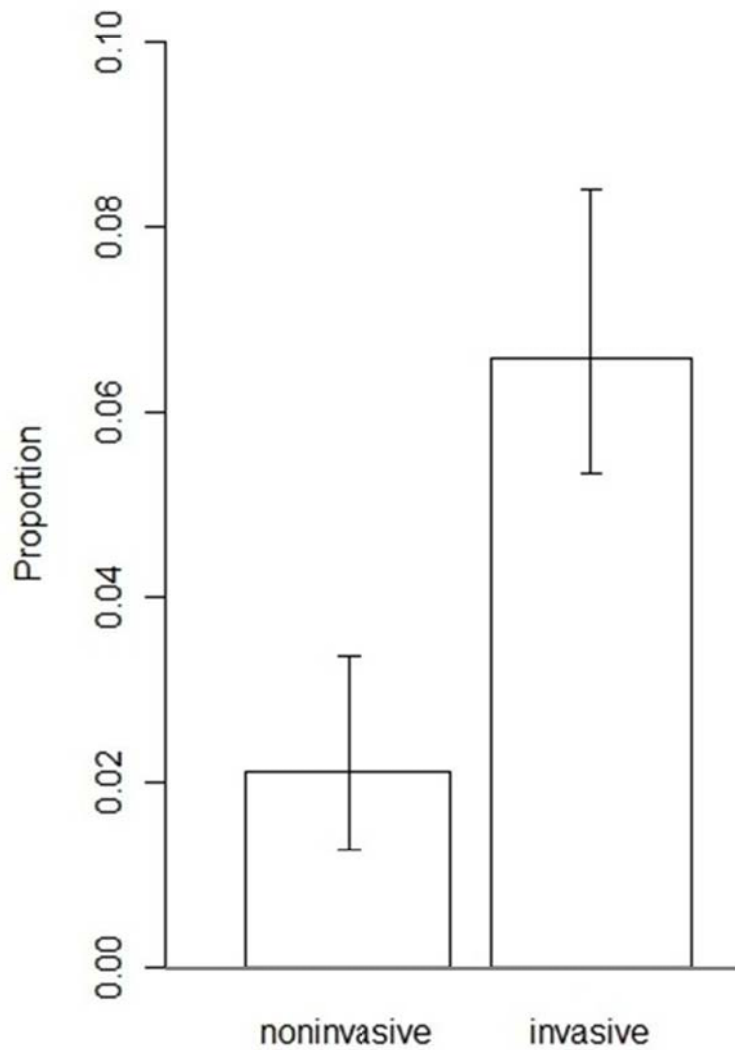


Figure 4. Mean proportion (\pm 95% bias corrected CI) of juvenile pinto abalone (*Haliotis kamtschatkana*) sighted during invasive one year post outplant SCUBA surveys at three sites was significantly greater than that using noninvasive methods (analysis of deviance, $F = 11.42$, $df = 1, 4$, $P = 0.028$).

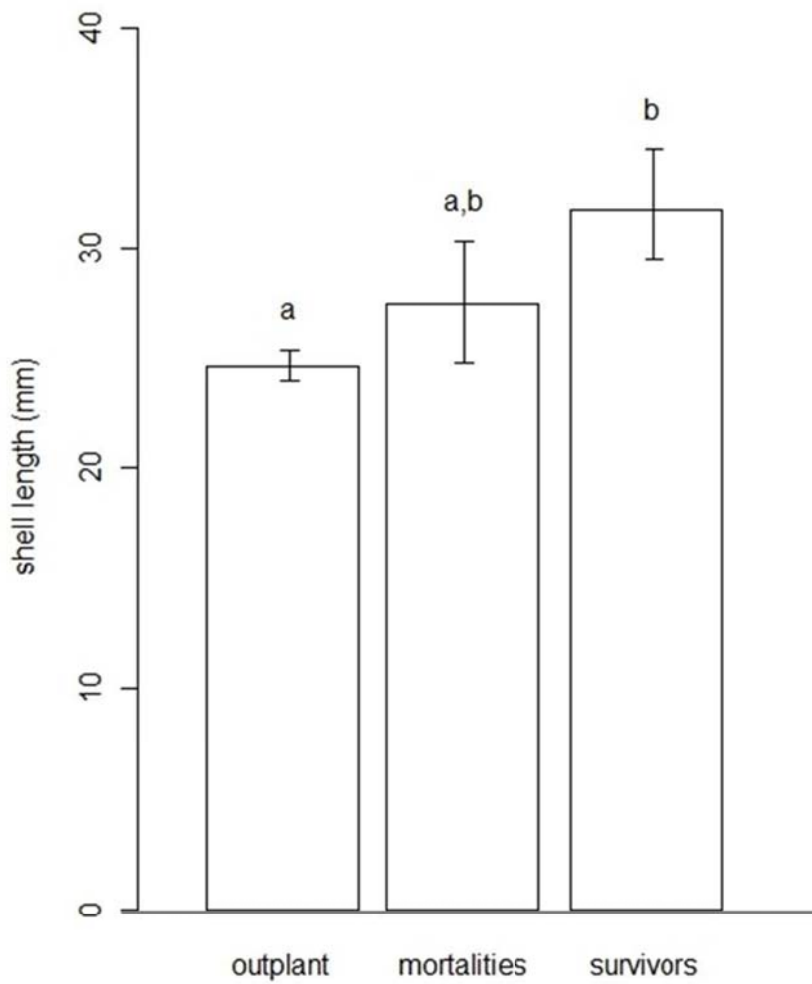


Figure 5. Mean initial shell length (\pm 95% bias corrected CI) of juvenile pinto abalone (*Haliotis kamtschatkana*) at outplant for all individuals (n=711), all recovered mortalities (n=36) and all recovered survivors (n=33) with known initial shell lengths. Bars with the same letter superscript indicates no significant difference in mean shell length ($\alpha = 0.05$).

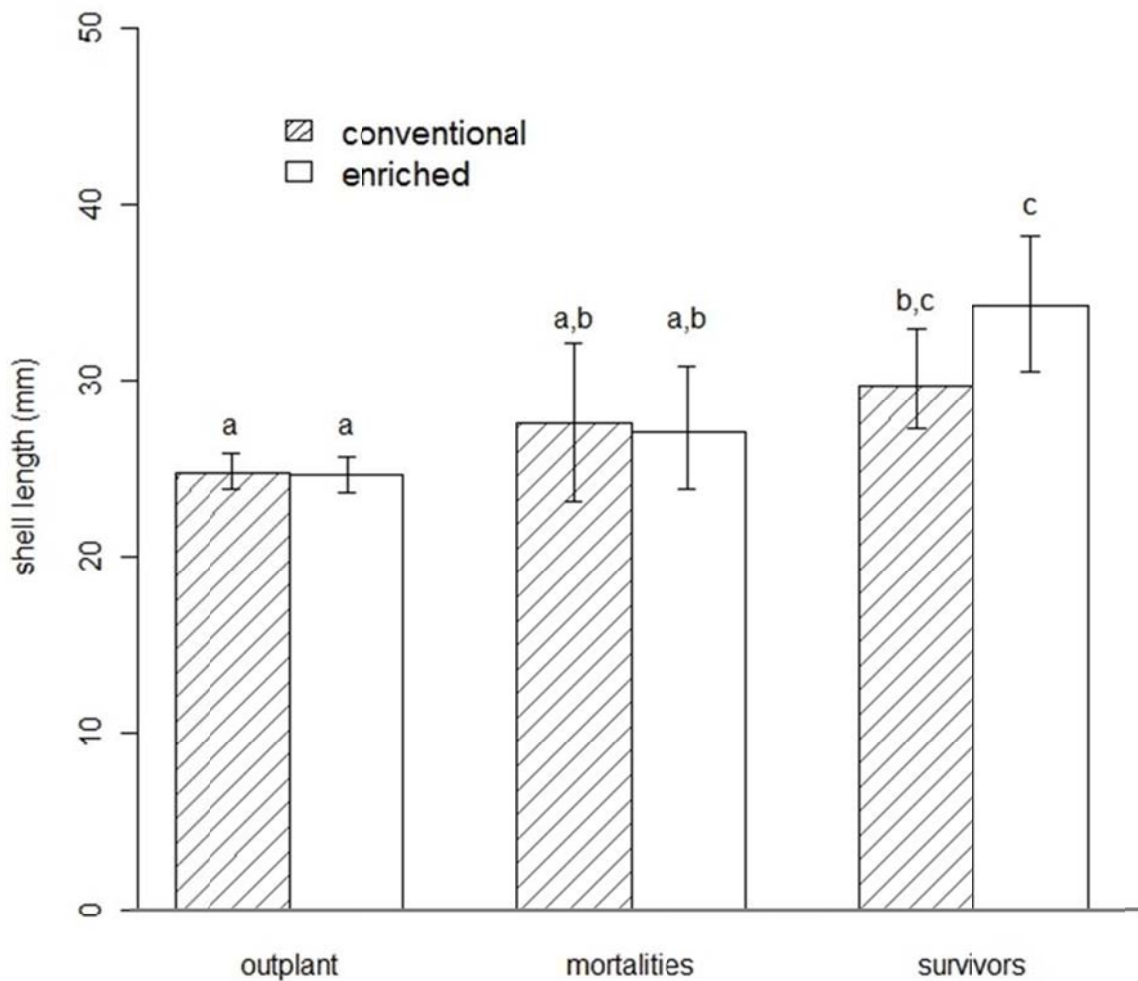


Figure 6. Mean initial shell length (\pm 95% bias corrected CI) of juvenile pinto abalone (*Haliotis kamtschatkana*) at outplant for all individuals (n=351,360 for conventional and enriched, respectively), all recovered mortalities (n=13,23) and all recovered survivors (n=18,15) with known initial shell lengths. Abalone were reared in conventional (cross hatched bars) and habitat enriched (open bars) tanks for 7 months prior to time of outplant. Bars with the same letter superscript indicates no significant difference in mean shell length ($\alpha = 0.05$).

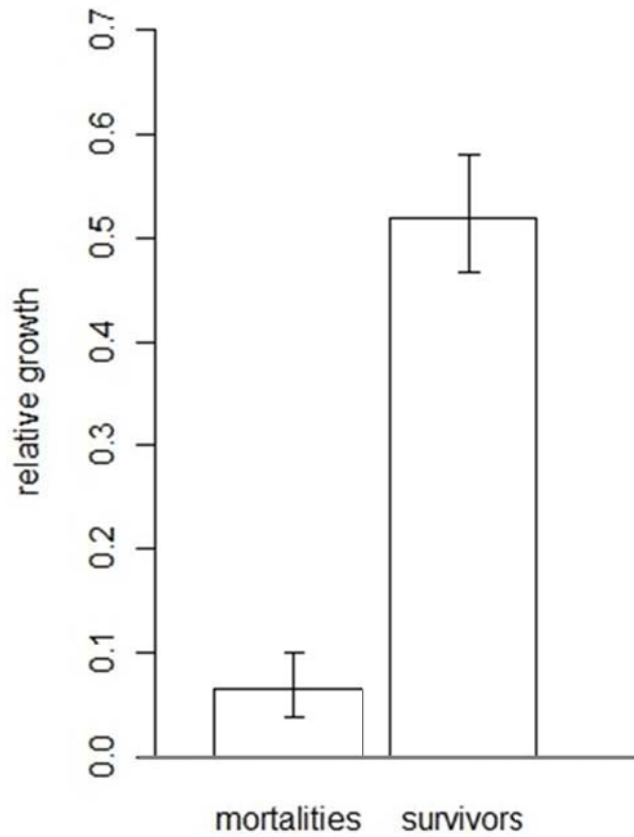


Figure 7. Mean relative growth ($\ln(\text{final SL}/\text{initial SL})$) (\pm 95% bias corrected CI) of juvenile pinto abalone (*Haliotis kamtschatkana*) for all recovered mortalities (n=32) and all recovered survivors (n=32) with known initial and final shell lengths.

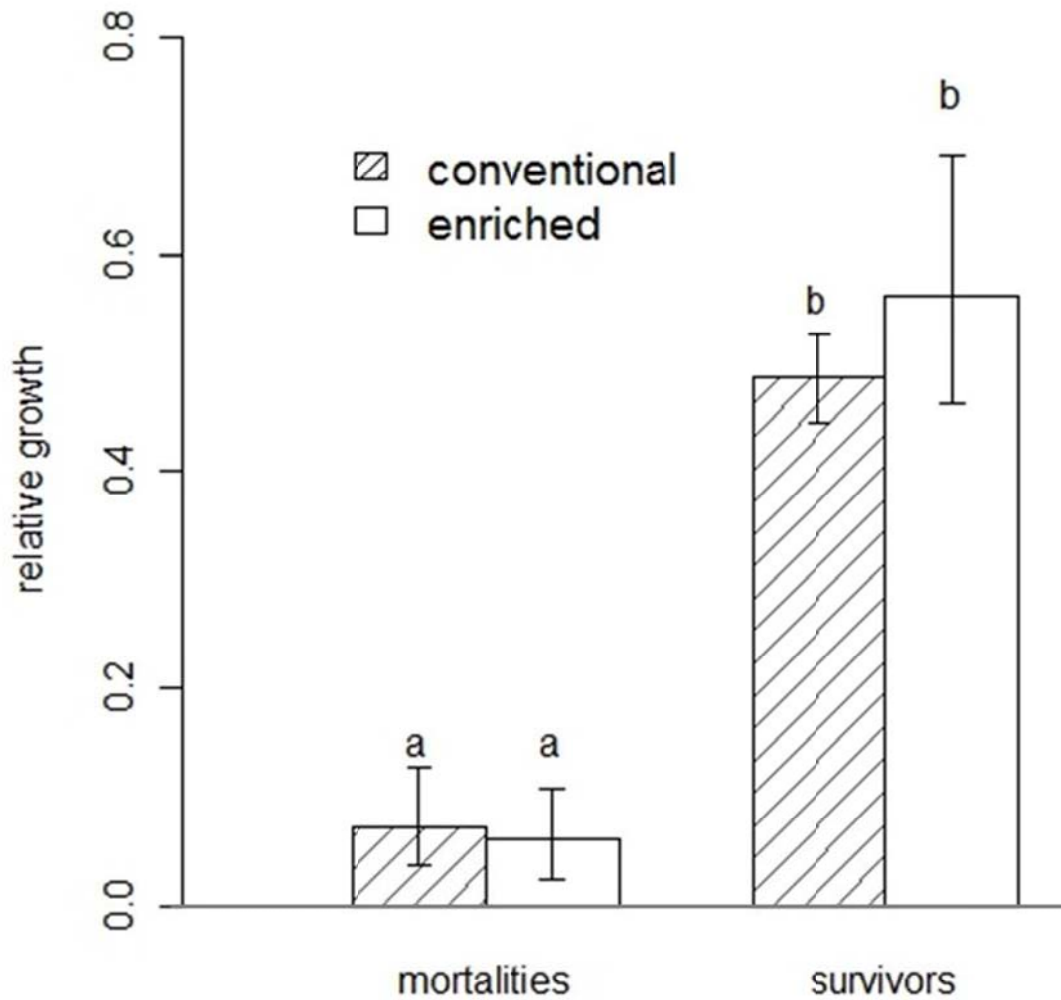


Figure 8. Mean relative growth ($\ln(\text{final SL}/\text{initial SL})$) (\pm 95% bias corrected CI) of juvenile pinto abalone (*Haliotis kamtschatkana*) for all recovered mortalities ($n=12,20$ for conventional and enriched, respectively) and all recovered survivors ($n=18,14$) with known initial and final shell lengths. Abalone were reared in conventional (cross hatched bars) and habitat enriched (open bars) tanks for 7 months prior to time of outplant. Bars with the same letter superscript indicates no significant difference in mean relative growth ($\alpha = 0.05$).

Table 1. Mean specific growth rate (SGR, % day⁻¹ ± 95% confidence interval) of juvenile pinto abalone (*Haliotis kamtschatkana*) reared in conventional or habitat enriched tanks in the hatchery for 218 days. Treatments with the same superscript indicates no significant difference in mean SGR ($\alpha = 0.05$).

Rearing treatment	Day			
	69	95	160	200
conventional	0.13 ^a ± 0.02	0.12 ^a ± 0.05	0.24 ^a ± 0.03	0.13 ^a ± 0.03
habitat enriched	0.11 ^a ± 0.03	0.07 ^a ± 0.07	0.24 ^a ± 0.04	0.03 ^b ± 0.03

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Appendix: Survival by family during the rearing and outplant periods

At the end of the 7 month rearing experiment, relative family survival proportion (the number of abalone of one family that survived after 7 months divided by the total number of abalone at the beginning of experiment) were equal between conventional and enriched habitat tanks with the exception of families 4, 6 and 8 (t-test assuming equal variances, $df = 6$, $P < 0.05$; Figure 9). Families 4 and 6 had a higher survival proportion in the enriched habitat tanks, whereas family 8 experienced a higher survival proportion in the conventional tanks. It is interesting to note that families 4 and 6 are half siblings by the father, however family 7 is also a half sibling by the father but survival proportion was the same between treatments. With respect to total survival proportion (the number of abalone of one family that survived after 7 months divided by the number of abalone of that family at the beginning of the experiment), families 1, 2, 3, 4 and 6 exhibited a significantly higher survival proportion in habitat enriched tanks than in conventional tanks (t-test assuming equal variances, $df = 6$, $P < 0.05$; Figure 10). There was no significant difference in total survival proportion between treatments for families 5, 7, 8 and 9 (t-test assuming equal variances, $df = 6$, $P > 0.05$; Figure 10).

These results indicate that in general, family did not influence relative survival proportion by treatment. However genetics may have influenced the chances of survival for a few families, but not necessarily in the same way. While interesting, these findings probably will not affect our current hatchery and outplant strategies. It is not our goal to select for certain families or traits in the hatchery and then outplant these individuals. High survival in the hatchery does not infer high survival in the wild. Traits selected for survival in a hatchery environment may

be less adaptive for challenges faced upon release into the natural environment (Blaxter 1976). Animals reared in a hatchery may not be able to perform basic survival strategies such as eating and avoiding predation needed in the wild (Olla et al. 1995). Our strategy is to create as many diverse family lines as possible with the current broodstock and subsequently outplant these genetically diverse groups.

It is unclear whether family influenced one year post outplant survival. While there were no differences in family proportion at time of outplant compared to the one year survivors (X^2 test for equality of proportions with continuity correction, $df = 1$, $P > 0.05$; Table 2), it is important to note that 24 of the 47 survivors came from unknown families due to tag loss or damage. This fact may have influenced the observed family proportions at the end of the one year outplant. Four of the families (families 5, 6, 7 and 9) had zero survivors at the end of the outplant period, and family 8 had the most number of survivors at 8 individuals (Table 2).

It would be interesting to investigate outplant survival by family in future studies and this could be accomplished more effectively through the use of different tagging methods.

Although very few bee tags were lost during this study, many were worn down so that colors and numbers of the tags became unreadable. One way to avoid this from occurring might be to add a drop of glue to the top of the tag as a protective coat or to use a different type of tagging method altogether. Some researchers have had high success using spring tags where the tag is attached to the margin of the shell and over time the tag becomes imbedded within the shell. These tags may not work with smaller juveniles however, so tagging must be delayed in the hatchery until the abalone are at a big enough size. If future outplant studies

use tagging methods that enable retention of family information, the survival proportion by family can be estimated. This might be good information to know when assessing genetic diversity of the surviving outplant population on a site. Even if diverse groups are initially outplanted, the survivors might exhibit a different family makeup by proportion.

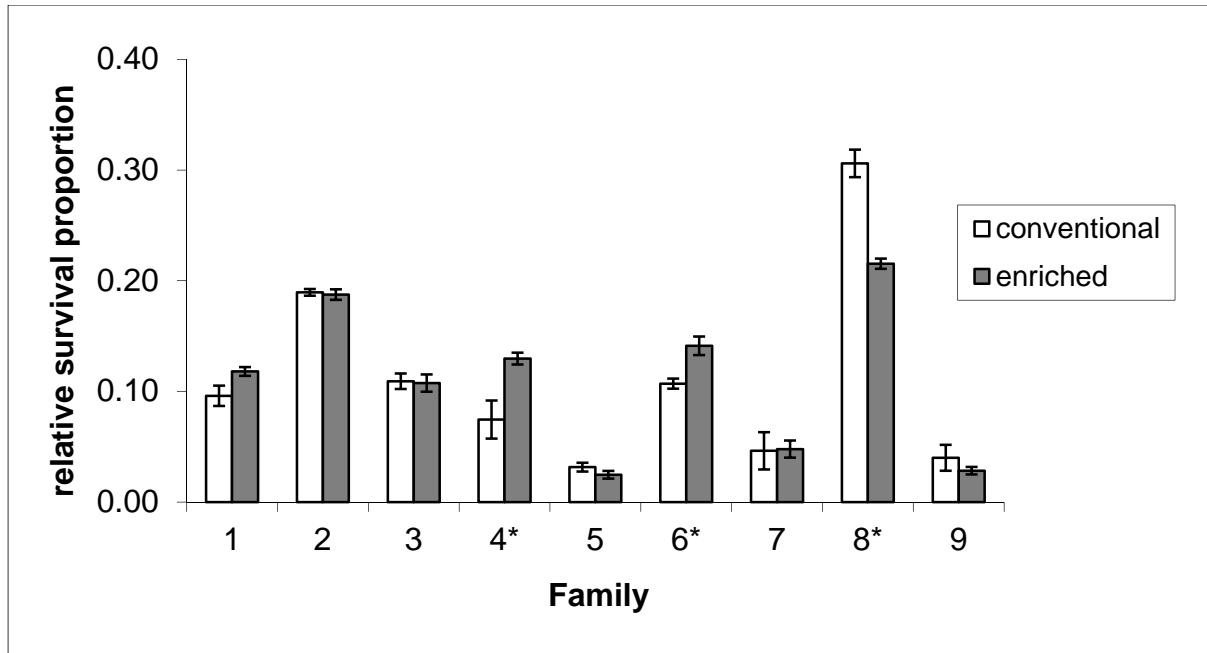


Figure 9. The mean (\pm SE) relative survival proportion (the number of abalone of one family that survived after 7 months divided by the total number of abalone at the beginning of experiment) for families reared in conventional (open bars) and habitat enriched tanks (solid bars). An asterisk (*) indicates there is a significant difference between proportions at $\alpha = 0.05$.

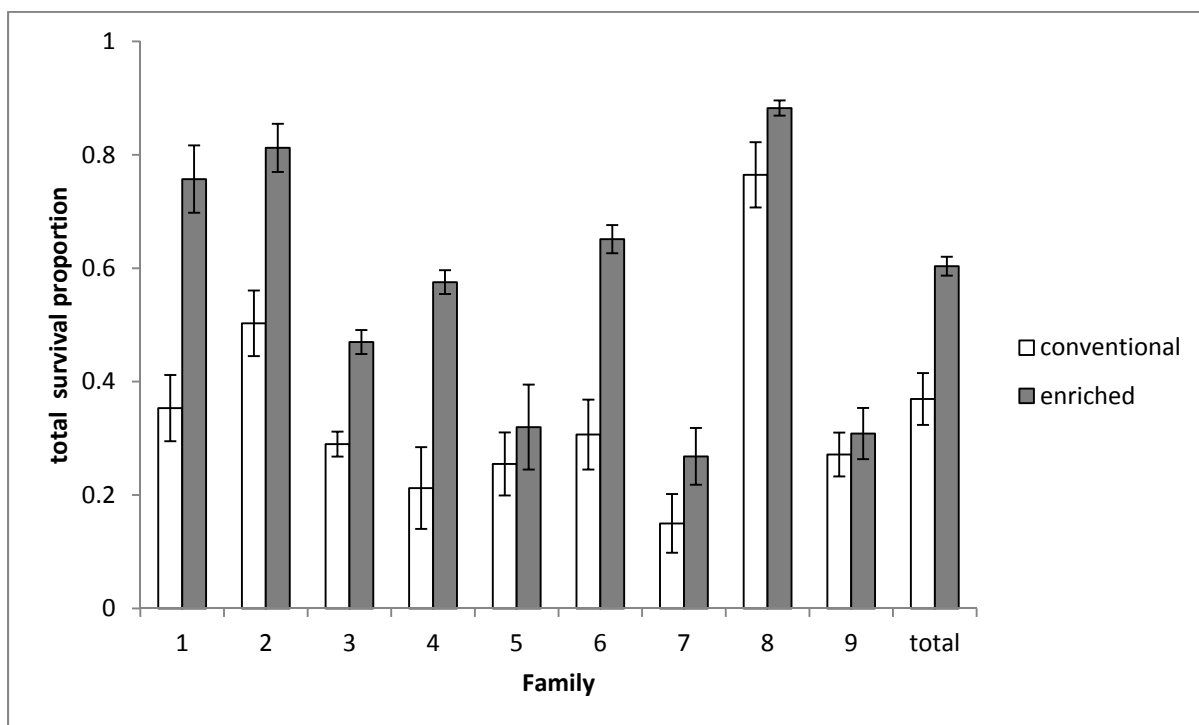


Figure 10. The mean (\pm SE) total survival proportion (the number of abalone of one family that survived after 7 months divided by the number of abalone of that family at the beginning of experiment) for families reared in conventional (open bars) and habitat enriched tanks (solid bars).

Table 2. The total (3 sites combined) proportion of juvenile pinto abalone (*Haliotis kamtschatkana*) by family that were initially outplanted and that survived one year post outplant. Unknown indicates that the family could not be determined due to tag loss or damage. An asterisk (*) indicates there is a significant difference between proportions at $\alpha = 0.05$.

family	proportion at outplant (n = 713)	proportion of survivors (n = 47)
1	0.10	0.02
2	0.16	0.17
3	0.08	0.09
4	0.08	0.04
5	0.01	0.00
6	0.06	0.00
7	0.02	0.00
8	0.24	0.17
9	0.01	0.00
unknown*	0.23	0.51