An Integrated Assessment of the Red Sea Urchin (*Strongylocentrotus franciscanus*) Fisheries in Southern California and Washington State: Addressing Sustainability and Vulnerability in the face of Climate Change

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Abstract

An Integrated Assessment of the Red Sea Urchin (*Strongylocentrotus franciscanus*)
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Integrated assessments are a tool that can help decision makers understand complex environmental problems. Climate change has the potential to vastly impact managed resources in terms social and economic factors, as well as causing changes to biogeochemical cycles and subsequent changes to species that these cycles affect. The U.S. west coast red sea urchin (*Strongylocentrotus franciscanus*) fisheries stand to be affected by climate change-related stressors such as ocean acidification, rising ocean temperatures, and changes in the frequency and intensity of ENSO events. An integrated assessment of the institutional and ecological contexts of two red sea urchin fishery case studies outlines existing knowledge gaps that may impede management efforts when it comes to the sustainability and vulnerability of these marine resources under threats from climate change.

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Chapter 1. INTRODUCTION

Anthropogenically-driven climate change has accelerated in the last 45 years (Pachauri et al. 2008), causing changes such as increases in sea level and average global temperature. The environmental consequences of these changes are the focus of much scientific research (Labat et al. 2004; Simpson et al. 2011; Pike 2013; Gerber et al. 2014; Toth et al. 2015). However, changes affecting the oceans, such as warming ocean temperatures and ocean acidification (OA) often are not fully considered when it comes to marine resource management (Chen et al. 2014). For example, the inclusion of CO_2-driven ocean acidification in fisheries management plans is relatively new (Cooley et al. 2015; Ekstrom et al. 2015). Recent studies (Cooley et al. 2015; Ekstrom et al. 2015) have focused on the effects of OA on the social, ecological and economic aspects of some U.S. shellfish fisheries and their management. The authors developed frameworks to assess the vulnerability and adaptability of shellfish fisheries to the increasing threat posed by OA. While these studies offer insightful analyses of specific cases, the analyses are data-intensive, and the necessary information is often unavailable. By examining currently used frameworks, resource managers can determine and adapt the necessary overarching structural components to form a more general management tool to be used to identify, and subsequently address, existing knowledge gaps in managed resource systems threatened by ocean acidification, as well as by the greater issue of climate change.

As defined by Janssen et al. (2004), a socio-ecological system (SES) comprises of a "subset of social systems in which some of the interdependent relationships among humans are mediated through interactions with biophysical and non-human biological units" (Janssen et al. 2004). The general framework developed by Ostrom (2007; also see Ostrom 2009) decomposes the analysis of SESs into four interacting components (resource system, resource units, governance systems, and users) that influence how an SES functions. Each of these components, as well as the interactions between them and the outcomes they produce, are further divided into a second tier of variables that can be combined analytically to determine the impact of each of their respective main components. Ostrom's framework can be used to identify the factors affecting managed resources and to determine how they can be adapted to achieve sustainability of that resource. This thesis aims to describe and understand the institutional and ecological settings of two managed resource systems, using the analysis of SESs developed by Ostrom (2007) to address the problem of sustaining sea urchin fisheries under conditions of climate change, with a focus on ocean acidification. Specifically, I use Ostrom's framework to compare the resource and management systems comprising the state-managed fisheries for the red sea urchin (*Strongylocentrotus franciscanus*) in southern California and Washington.

Sea urchins have a global distribution, with over 700 species found in habitats varying from intertidal zones to deep ocean floors (Stanford University 2008). There are 18 countries with at least one sea urchin fishery, and around 20 species are commercially harvested (Andrew et al. 2002). Sea urchins play a significant role in their ecosystems, especially in regard to predator-prey interactions. High densities of sea urchins can result in ecosystems with very low algal biomass due to urchin grazing, while ecosystems with low densities of sea urchins can suffer from disruptions of trophic level dynamics such as prey switching or decreases in urchin predator abundance (Harrold & Reed 1985). Sea urchins are also sensitive to their surrounding environmental conditions, and are used by the U.S. Environmental Protection Agency as indicator organisms for nearshore water quality testing (US EPA Regional Laboratory Network 2014). I chose to investigate sea urchins because, as calcifiers, they are directly impacted by OA, as well as other environmental conditions affected by climate change, such as increasing ocean temperature. I focus on two sea urchin fisheries, specifically the Washington state and southern California red sea urchin fisheries. I selected the Washington fishery for its proactive history of management, and I chose the southern California fishery to provide an example of a more reactive history of management. Although the management of the two fisheries currently includes many similar rules, I ask whether differences in the institutional and ecological context of two red sea urchin fishery case studies are detectable, and if so, what do these differences imply for the vulnerability and sustainability of these fisheries under impending environmental change.

The SES framework developed by Ostrom (2007) was designed to be used for assessing the sustainability of terrestrial managed resources. However, by adjusting the definitions for the main components and their second-tier variables, it is possible for any type of resource management to use Ostrom's framework. Ostrom's SES framework divides the components of the resource system into ecological (resource system and resource units) and institutional (governance system and users) components.

This basic structure is discernable in recent studies that examine the role of assessment models and vulnerability analyses in the field of managed marine resources. For example, Cooley et al. (2015) discuss the development of an integrated assessment model (IAM) to aid decision makers regarding the adaptation of the U.S. sea scallop fishery management plan for the impacts of climate change. The IAM has three sub-models, two of which make up the ecological aspects of the model, while the last sub-model addresses the institutional aspect. Although the IAM does produce information that could be factored into the sea scallop fishery management plan, the researchers explicitly state the use of multiple assumptions within the model, and the need for further research in order to fully understand all of the sub-model variables and how they interact.

A recent study by Chen et al. (2014) approaches the topic of how fisheries are influenced by institutional change from the perspective of fishermen's livelihoods. The authors developed a livelihood vulnerability index (LVI) framework to assess how fishermen would be affected by the loss of fishing area due to the establishment of marine protected areas (MPAs). The LVI framework consists of six components in which four are related to the users of the managed resource and the remaining two represent the resource system. The framework can be used by resource managers to more accurately estimate potential impacts of planned management decisions, but ultimately, the application of the LVI framework is restricted to managed resource systems that are well understood. This is then further limited by the fact that the framework does not take into account the potential impacts related to climate change.

Ekstrom et al. (2015) present an analytical framework that allows the authors to identify how fishing communities respond and adapt to change associated with OA. They perform a multidisciplinary vulnerability analysis at the local level on shellfish fishing communities across the U.S. in order to determine levels of vulnerability and capacity amongst fishermen and their fishery's management system, and to understand the vulnerability of the shellfish resource system to OA. The analysis is divided into two main components, the marine ecosystem exposure to OA (ecological) and the social vulnerability (institutional). Ekstrom et al. (2015) goes one step farther than Cooley et al. (2015) and Chen et al. (2014), in that the authors identified the types of knowledge gaps that exist within each element of their analysis framework, as well as the level of effort required to fill these gaps.

I apply the basic structure from Ostrom's SES framework to identify the level of understanding of a managed resource and the potential impacts that OA may have on the ecological and institutional aspects of two red sea urchin resource systems, while including aspects from the assessments and analyses presented in Chen et al. (2014), Cooley et al. (2015), and Ekstrom et al. (2015). This thesis provides an integrated assessment of two U.S. west coast red sea urchin fisheries. While the two fisheries share a target species, they are located in areas with different geographic characteristics, have different structures of governance and users, and potentially will have different impacts of OA on their resource systems. The goal of this thesis is to understand how these differences affect each fishery's management plan, and to compare existing knowledge gaps from both fisheries to investigate the vulnerability and sustainability of these fisheries in light of impending impacts of climate change. An assessment such as this could be used by resource managers as a starting point to identify areas of their management plans that are more vulnerable to impacts of OA, and climate change in general.

Chapter 2. METHODS

I adopted Ostrom's framework which consists of four main components, namely the resource system, resource units, governance system and users. I then specified multiple variables ('second tier variables') within each component that I scored to provide a semi-quantitative integrated assessment. For each second-tier variable, I reviewed the literature to assign an institutional score and a confidence score. The institutional score reflects the level of understanding and status of the fishery compared with past assessments, as well as the level of involvement of the governance and users of the managed resource. In comparison, the confidence score is an indication of how well understood the fisheries are, such as the amount of information collected about the managed resource system, its governance system, and the users of the resource itself, as well as the level of consensus regarding the available information. I used internet search engines, such as Google

Scholar and the University of Washington Libraries Search, to conduct a search for literature containing the terms “red sea urchin”, “*Strongylocentrotus franciscanus*”, “California urchin fishery”, and “Washington urchin fishery”, as well as other general search terms to address the localities where the fisheries occur. I filtered the search results to retain literature concerning the Southern California red sea urchin fishery focused around the Channel Islands and nearby ports, and the Washington State red sea urchin fishery based around the San Juan Islands. I obtained catch and permit data for the period 1991-2014 for both fisheries from their respective state’s Department of Fish and Wildlife (DFW). I used the DFW websites to access information about the fishery areas, and communicated by email and telephone with employees at both DFW offices to obtain information additional information about fishery management.

Using the information provided from the literature search, I coded each second-tier variable on a categorical scale from 0 to 3, where a score of 0 indicates insufficient information available to determine the status of the variable and a score of 3 indicates a “good” status for the variable. I opted for a simple categorical scale in order to be able to classify discernable differences between individual variables, as well as between each main component of the two fisheries. The scores differ in meaning across components. For both the resource systems and resource units, the institutional scores indicate how well understood are the state of the ecosystem where the resource is found and the status of the stock, in terms of both population size, as well as the conditions across life history phases. Higher scores were given when the available information provided a comprehensive understanding of the variable. For example, RU6 was given a score of 3 for both fisheries as the physical appearance of red sea urchins is well documented. Whereas for RU2, the southern California fishery was given a score of 2 while the Washington state fishery was given a score of 1. This is because the variable has multiple facets that need to be understood in order to receive a high score, and while the reproduction of red sea urchins has been extensively researched and is well understood for the species in general, there is more information available regarding the reproduction rate and migration patterns of urchin populations in southern California than in Washington state.

The ratings of governance systems variables signify the existence and functionality of management entities, and the users variables are rated according to how active or well-informed are the various stakeholders of the managed resource. An institutional score of 3 indicates a “good” (in the sense of “more sustainable” and “less vulnerable”) status in the context of active management of the managed resource. For example, for variable U1 – number of users, a rating of 3 indicates that the number of fishermen is actively controlled by the resource managers through methods such as limited access through permits and the requirement for fishermen to remain active within the fishery in order to keep their access. Whereas a rating of 2 would indicate that while the number of fishermen is partially controlled by the need to obtain a permit, there is no cap on the number of permits within the fishery. Similarly, for variable U8 – importance of resource, a “good” rating of 3 would indicate the existence of knowledge explicitly explaining the proportion of resource user’s household income that is dependent on the fishery and how this proportion varies amongst users within the fishery, as well as how important the fishery is to the culture of the community.

Lastly, the confidence score ratings denote the amount of information available concerning each variable, and the level of agreement between various literature sources. The ratings for confidence scores vary depending on the type of information and the type of source. For example, for RS1 and GS1, both states were assigned confidence scores of 3 despite RS1 having significantly fewer sources than GS1. This is because the sources for RS1 explicitly and fully addressed the variable, whereas the multiple GS1 sources each addressed aspects of the variable, and consensus and understanding were only reached when considering all of the sources. Similarly, variables CA RU5 and U2 for both states received scores of 1 also in spite of a difference in the number of sources. In this case, CA RU5 had multiple sources that stated poor understanding of the size of the red sea urchin population in southern California (institutional score), as well as poor understanding of the population dynamics as there was little consensus amongst sources (confidence score). This is compared to variable U2 where both states lacked information regarding the socioeconomic attributes of resource users and few sources that addressed the variable were found. The institutional and confidence scores revealed existing knowledge gaps within the two fisheries, and gave an indication of their vulnerability and sustainability as a managed resource.

Chapter 3. RESOURCE SYSTEM: HISTORICAL, ECOLOGICAL, AND OCEANOGRAPHIC CONTEXT

The red sea urchin fishery along the west coast of the U.S. began in the late 1960s in response to increasing demand for uni (sea urchin roe) from the growing numbers of Japanese restaurants in the U.S. (Kato & Schroeter 1985). At that time, sea urchins were considered to be pests by causing damage to kelp forests, which resulted in some communities in southern California organizing “urchin kills” (Kato & Schroeter 1985) to reduce urchin populations. As an alternative to unsystematic removal of urchins, the development of an organized fishery was seen as a way to create low-skilled labor opportunities and increase the position of the U.S. in the international seafood market. By 1972, the red sea urchin fishery along the U.S. west coast was established, with the largest fisheries being those in southern California and the Strait of Juan de Fuca in Washington State.

In southern California, the red sea urchin fishery occurs within the Southern California Bight – the coastal and offshore area extending from Point Conception to the U.S.–Mexico border (see Fig. 3.1a) – but, the majority of the fishery's catch within this region comes from the Northern Channel Islands. The ecological communities surrounding the Channel Islands have been studied extensively (Ebeling et al. 1985; Tegner & Dayton 1991; Davis 2005; Halpern et al. 2006; Polefka & Forgie 2008), resulting in a wealth of knowledge regarding the rocky benthic communities that dominate the area, including the kelp forests that grow in waters of depths of up to 30 meters (see CA ratings for variables RS1, RS2, and RS3 in Table 7.1). The fishing effort for red sea urchins is focused on these kelp forest areas, where the average depth reached by urchin divers is around 8 to 12 meters (Kato & Schroeter 1985; Kalvass & Hendrix 1997).

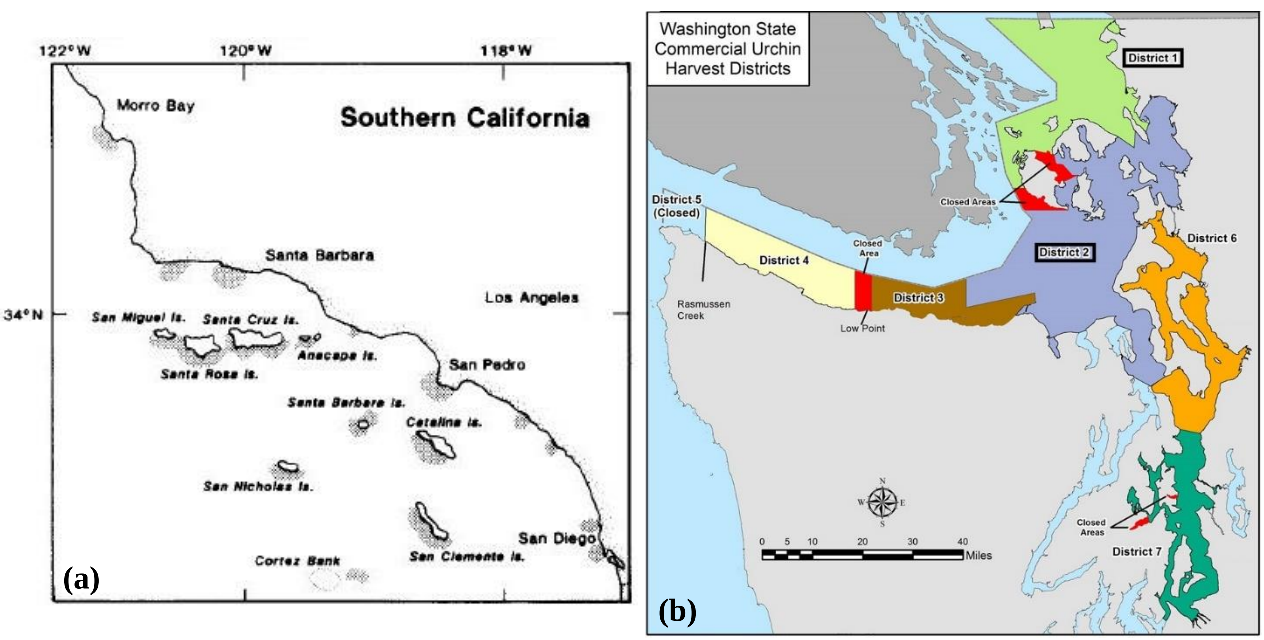

Figure 3.1 – (a) map of Southern California and the Channel Islands red sea urchin fishery area, and (b) map of Washington red sea fishery districts, with Districts 1 and 2 (labels have black boxes on the map) being the focus of this thesis

Unlike the ecologically-defined resource system boundaries for the red sea urchin fishery in southern California, the boundaries of the Washington fishery are defined by the users (see WA ratings for variables RS1 and RS2 in Table 7.1). Beginning in 1977, the Washington red sea urchin fishery created three harvest districts, which were then further divided into five districts in 1987 (see Table 7.1: WA RS3). These harvest districts were used to maintain a rotational harvest schedule to ensure each district was harvested every third year as an attempt to prevent overfishing in any given area (Pfister & Bradbury 1996; Carter & VanBlaricom 2002). The majority of catch occurs in Districts 1 and 2, which together represent the waters surrounding the San Juan Islands and extending northward to the U.S.–Canada border, and southward to Port Townsend, Washington (see Fig. 3.1b) (State of Washington 2015). A variety of habitat types occur in Districts 1 and 2, with the urchin population mostly occupying nearshore kelp and rocky slope habitats. While literature searches produced research from a variety of sources for the southern California fishery region, most research regarding the Washington fishery region has been carried out by the Washington Department of Fish and Wildlife (WDFW), with some studies also performed by researchers at the University of Washington in Seattle.

Section (a) of Table 7.1 shows that the type, size, and boundaries of both fisheries resource systems are well understood, with consensus between multiple sources. However, the Washington resource system has user-defined geographical boundaries, and therefore the size of the resource system does not change without management consensus. On the other hand the southern California resource system lacks explicitly defined boundaries and is instead delineated by the habitat of the resource units, kelp forests. Thus, while the size of the southern California resource system is well understood, it is interpreted as the distribution of the resource itself, and therefore as the fishery catch rates are declining, the size of the resource system is also slowly declining. While

Washington fishery catch rates are also declining, the management boundaries for the resource system are unchanged. For this reason, the RS3 – size of resource system variable is given a rating of 2 for southern California and a rating of 3 for Washington. Nevertheless, the actual area where harvesting occurs will decrease for both fisheries as catch rates decrease, but the clear boundaries of the Washington fishery allow for straightforward area closures for population recovery, or other executive management decisions.

Areas in which red sea urchins are commonly found usually vary between two alternate states characterized by either high macroalgal biomass and low herbivore biomass, or the converse (see Table 7.1: RS6) (Harrold & Reed 1985). Urchins are voracious predators that, in high concentrations, are capable of intense grazing that clears macroalgal growth, leading to the formation of what are known as urchin barrens (Rogers-Bennett et al. 1995). The southern California fishery began as a way of both addressing the growing market demand for uni, as well as a means to reduce the number of urchins in order to prevent the formation of urchin barrens, especially since reductions in macroalgae can cause cascading effects on the rest of the system (Kato & Schroeter 1985; Folke et al. 2004). While controlling the level of herbivory does help prevent the shift from kelp forest to urchin barren, an uncontrollable driving force does exist for the southern California coastal region, the El Niño Southern Oscillation (ENSO) (see Table 7.1: CA RS7). Characterized by warmer waters, reduced upwelling, reduced nutrient concentrations, and an increase in the number of storms along the U.S. west coast, ENSO events recur on an irregular and generally unpredictable timescale (Wittenberg et al. 2014; National Oceanic and Atmospheric Administration 2016) and can last anywhere from a few months to a few years. The warm, low nutrient water can cause growth rates and survivorship among kelp to decline, and can reduce the tensile strength of kelp fronds, leading to increased storm-induced mortality and reduced kelp abundance as a food source for urchins (Kato & Schroeter 1985). Studies have also suggested that the decrease in food, as well as the warmer water temperatures, causes a decline in red sea urchin feeding rates that then results in lower roe quality and yield (Kato & Schroeter 1985; Kalvass & Hendrix 1997). Moreover, El Niño storms discourage urchin divers from fishing, hence fishing effort declines. Therefore, strong ENSO conditions are seen to potentially lead to declines in total catch (Kato & Schroeter 1985; Kalvass & Hendrix 1997; Chen et al. 2014). In Washington state, there are no discernable patterns of El Niño impacting catch or urchin diver behavior in the Washington state managed fishery (Hank Carson, personal communication).

Although such environmental factors can be unpredictable and uncontrollable, a tool that can be used by resource managers to foster ecosystem recovery and improve resilience is the formation of marine protected areas (MPAs) or no-take areas. These areas prevent the removal of resources in order to conserve local biodiversity. In the case of the southern California and Washington red sea urchin fisheries, both states have designated MPAs, or no-take areas, located in close proximity to sea urchin harvest districts, although these sites were not designated for the explicit protection of urchins. In fact, in southern California, a goal of the marine reserves is to preserve or re-establish trophic webs dominated by urchin predators (e.g., the sheephead wrasse and spiny lobster), which are also commercially harvested in the region (Airamé et al. 2003; Lindholm et al. 2010). In southern California there are over fifty MPAS, which protect around 15% of the region's waters (California Department of Fish and Wildlife 2016). Benefits of designating areas as MPAs or no-take include the fact that organisms found in the area can keep reproducing (Airamé et al. 2003; Davis 2005). For example, the urchin fishery relies on harvesting urchins before spawning occurs since the roe is the harvested product of the fishery. By having an MPA/no-take area nearby to a harvesting area, it ensures that at least some urchins are not harvested and can still spawn for the purposes of population recovery (Lindholm et al. 2010). To increase the efficiency of MPAs/no-take areas as a management tool for population recovery, their designation can be planned to coincide with the larval source (i.e. where spawning occurs) and sink (i.e. where larvae settle) patterns (Airamé et al. 2003). These patterns may be determined by studying the life history parameters of the target resource, as well as the oceanographic conditions in the area.

In southern California, there have been experiments to determine whether it is possible to artificially increase urchin population sizes or reproductive success using methods of reseeding (e.g., juveniles reared in hatcheries can be released into wild populations to enhance natural recruitment). However, while the experiments did have some success, they found that juveniles needed to be reared until they measured around 15mm in order to offset high mortality upon release, which resulted in the proposed management method not being a cost effective option (Andrew et al. 2002). Other studies in southern California used artificial substrates to measure settlement patterns in the region (see Table 7.1: CA RS4[1]), allowing researchers to determine that strong localized recruitment has replenished populations since the peak in fishery catch rates in 1988. This strong localized recruitment has been hypothesized to be a result of high larval retention due to ocean circulation patterns in the Southern California Bight (Ebert et al. 1994; Andrew et al. 2002).

The California Current flows southward along the U.S. west coast, which when then combined with seasonal offshore winds, creates an upwelling zone that brings cooler, nutrient-rich waters

[1] CA RS4 received an "insufficient" rating for its' institutional score because the artificial substrates used to measure settlement were removed after the experiment had concluded. However, a confidence score was given because some literature addressed human-constructed facilities. There was no mention of human-constructed facilities in the literature pertaining to the Washington fishery, thus both scores received "insufficient" ratings. An "insufficient" rating indicates that there is not enough knowledge available to assess a variable, and therefore further research is needed to determine how resource management is addressing the vulnerability and sustainability of the fishery regarding the given variable.

upwards from deeper areas. This occurs to a lesser extent in the Southern California Bight because the coastline curves eastward at 34°N and the current continues southward, with lower levels of wind stress acting on the current when it is further from the shore (Ebert et al. 1994). The current then curves eastward at 32°N and forms an eddy with water flowing northward towards Point Conception at 34°N (see Fig. 3.1a). This eddy, called the Southern California Countercurrent, causes increased water retention within the Southern California Bight, with nutrient-rich waters being brought in by the California Current (see Table 7.1: CA RS5 and RS8). While the outer Channel Islands are closer to the California Current and therefore have coastlines that are exposed to the cooler, nutrient-rich water that it brings in, the waters that are part of the Southern California Countercurrent warm up and lose some of their nutrient load as the current moves northwards. However, the topography and bathymetry within the region causes the formation of multiple small-scale cyclonic eddies which induce localized upwelling of nutrient-rich water within the Southern California Bight (DiGiacomo & Holt 2001). These cooler, nutrient-rich waters allow for the growth of kelp forests, which are the basis for the ecosystem where red sea urchins are most commonly found in southern California.

Although upwelling also occurs along the Washington coast, the fishery is based in the waters around the San Juan Islands which are located within the Strait of Juan de Fuca. Waters upwelled along the outer coast of Washington enter through the Strait of Juan de Fuca in order to reach the San Juan Islands. The strait is 20 kilometers wide at its narrowest point, and it is the only outlet to the ocean for Washington State's Puget Sound, and British Columbia's Strait of Georgia, which together total almost 13,000 square kilometers of sea surface area (Mackas & Harrison 1997). These geographic features significantly influence the waters surrounding the San Juan Islands, with strong seasonal fluctuations that switch between upwelled water entering through the Strait of Juan de Fuca in the winter months and river discharge flowing from Puget Sound and the Strait of Georgia outwards to the open ocean in the summer months. The bathymetry of the area, as well as the shape of the coastline, contributes to a higher residence time of the waters in this region (Mackas & Harrison 1997).

The cold, upwelled water that enters through the mouth of the Strait of Juan de Fuca has properties such as high salinity, high nutrient-load, and high dissolved carbon dioxide (CO_2) (MacFadyen et al. 2008; Murray et al. 2015). In addition to this, anthropogenically-caused increases in atmospheric CO_2 over the last 250 years that have led to the sharp increases in atmospheric CO_2 concentrations (Feely et al. 2008). These high concentrations in turn increase the oceanic uptake of $CO_{2,}$ decrease seawater pH, and result in ocean acidification (OA). The dissolved CO_2 undergoes a series of chemical reactions that reduce the concentration of carbonate ions in seawater, and this, in turn, decreases the ability of calcifying organisms to form their calcium carbonate shells and skeletons (Doney et al. 2009). The northern portion of the U.S. west coast is particularly vulnerable to significant decreases in pH as upwelling in the region brings up water from the end of the ocean's thermohaline circulation current which, due to the accumulation of CO_2 at depths, has pH levels that are among the lowest in the ocean (see Table 7.1: WA RS7) (Ainsworth et al. 2011). Thus, the marine ecosystems along the Washington coastline are extremely vulnerable to the projected increase in OA.

Chapter 4. RESOURCE UNITS: CLASSIFICATION AND ATTRIBUTES

Sea urchins are classified in the phylum Echinodermata, a group of species with radially symmetrical bodies and calcareous plates making up a mesodermal skeleton. The calcareous plates in urchins form a hard shell called a ‘test’ which is covered in their characteristic spines. Urchins have five point radial symmetry, which can be seen in the five calcareous plates that make up their mouths found on their undersides, as well as their internal structure that has the five lines of roe and the five sections of their water vascular system (Kato & Schroeter 1985; California Department of Fish and Game 2003). Red sea urchins have a dark purple to reddish test and spines, and the length of their spines is dependent on the level of turbulence in their habitat (Kato & Schroeter 1985).

Red sea urchins are found in rocky habitats along the coast from Baja California to Alaska, and around Japan (Kalvass & Hendrix 1997; Lai & Bradbury 1998). In southern California, their habitat for the most part overlaps with the canopy cover of giant kelp (*Macrocystis pyrifera*), which spans roughly 45 km^2 from central California to the Mexican border (see Table 7.1: CA RS3) (Andrew et al. 2002). Red sea urchins congregate in areas with more food and form dense aggregations that are patchy throughout kelp beds, with lower concentrations found along the outside edges (see Table 7.1: RU7). In Washington, *Macrocystis* is uncommon, and red sea urchins are generally associated with bull kelp (*Nereocystis luetkeana*), but can also be found in areas that commonly have drift algae (Kato & Schroeter 1985; Pfister & Bradbury 1996), and exhibit similar patchy distributions based on food availability as red sea urchin populations elsewhere on the U.S. west coast (Rogers-Bennett et al. 1995; Andrew et al. 2002). The spatial patchiness of urchins benefits the population by increasing the concentrations of individuals and thereby increasing the fertilization success of spawning events. However, as populations decline due to fishing or other causes, depensation can occur, whereby the “smaller numbers of patches and reduced patch size can… limit fertilization success” (see Table 7.1: RU2) (Kalvass & Hendrix 1997).

Since the fishery is based on the harvest of roe from urchins, the optimal harvest time is before spawning occurs. Therefore, understanding fertilization success and recruitment rates is important for sustainable fishery management. Red sea urchins are a long lived species, with individuals living for over 100 years (California Department of Fish and Game 2003), and growth rates that are highly dependent on food availability and feeding rates (Pfister & Bradbury 1996). Studies have found that red sea urchins exhibit foraging behavior when water temperature is between 6 °C and 25°C, and feeding rates are greatest at 16°C (Kato & Schroeter 1985). Maturity is generally reached when an urchin is around 1-2 years old and its test measures roughly 50mm (Kato & Schroeter 1985; Pfister & Bradbury 1996). The production of gonads is also highly correlated with feeding rates, thus while spawning is seasonal, it can be highly variable based on location and year. For example, the red sea urchin population in southern California has annual spawning events that usually occur between January and April (Table 7.1: CA RS7 and RU2), while the spawning events of Washington populations are episodic (Table 7.1: WA RS7 and RU2) (Pearse & Hines 1987; Kalvass & Hendrix 1997). After spawning and fertilization, larvae pass through multiple free-swimming development stages before juveniles settle in the benthos, by which time the urchins are already 6-8 weeks old (Kato & Schroeter 1985). With the exception of the free-swimming larval stages during which urchins can travel up to hundreds of kilometers on currents, urchin movement is usually minimal (see Table 7.1: RU1). Juveniles will move towards adults after settlement in order to gain protection from adult spine canopies, where they remain until they reach maturity (see Table 7.1: RU3). Adults are relatively stationary, with rates of movement increasing as food abundance decreases (Kato & Schroeter 1985).

Red sea urchins have mostly been studied for the purposes of their fisheries (see Kato & Schroeter 1985; Pfister & Bradbury 1996; Kalvass & Hendrix 1997). Understanding of the resource varies across factors (section (b) Table 7.1), with some variables such as the taxonomic identification of red sea urchins being obvious to resource users (RU6). Other variables, such as RU1 and RU3, have a medium level of understanding, which is functional for resource management because enough is known about the mobility and interactions of red sea urchins that informed management decisions can be made. Yet, some knowledge gaps that impede management still exist (e.g. RU5). Given that both the California and Washington fisheries are targeting the same species, some knowledge about red sea urchin life history can be shared. The free-swimming larval life stage of red sea urchins suggests that geographically distant subpopulations of juvenile and adult urchins have the potential to be linked by larval dispersal. These dispersal mechanisms can create metapopulations consisting of several subpopulations over a large geographic ranges (Botsford 2001). While the California Current does traverse through multiple red sea urchin subpopulations as it flows southwards along the U.S. west coast, studies have shown that there is significant genetic differentiation between neighboring adult subpopulations, as well as differentiation between adults and recruits found in the same areas (Moberg & Burton 2000). However, there are no detectable geographic patterns for the genetic differentiation of either adults or recruits, suggesting that larvae may be travelling southward along the California Current as well as northward whenever the flow direction of the current changes during strong ENSO events (Debenham et al. 2000). Despite this, the genetic differentiation observed in red sea urchins along the U.S. west coast indicates that there is less mixing within the larval pool than would be expected given the life history characteristics of urchins and the oceanographic conditions of the California Current region (Moberg & Burton 2000; Debenham et al. 2000). Furthermore, with regards to the urchin fisheries in southern California and Washington state, the differences in the oceanographic and ecosystem context in which the two fisheries exist, combined with the genetic differences that occur within regional subpopulations, does lead to behavioral, growth, and reproductive rate differences between the two urchin fishery populations, and thus each fishery must develop a management approach that addresses the existing knowledge specific to red sea urchins in their resource system.

Mortality rates in red sea urchin populations are highly size-dependent, with individuals with tests measuring smaller than 60mm having the lowest chances of survival (Kalvass & Hendrix 1997). This includes larval stage urchins that fall prey to suspension feeders often found in rocky habitats. In southern California, mid-sized urchins exhibit elevated mortality rates due to predation by California sheephead (*Semicossyphus pulcher*) and spiny lobster (*Panulirus interruptus*) (Kato & Schroeter 1985). This higher mortality rate is not seen as disruptive to the population size since the region experiences higher recruitment and growth rates than other U.S. west coast populations (see Table 7.1: RU2) (Kalvass & Hendrix 1997). The higher recruitment and growth rates are seen to be balanced by the higher mortality rates, resulting in a stable population size. However, despite higher recruitment rates, the average size of catch and the number of urchins caught has been declining since a peak in fishery catch rates in 1988 (Andrew et al. 2002). The Washington fishery catch also peaked in 1988, with surveys then showing that the San Juan Islands red sea urchin populations declined by almost 50% between 1988 and 1989 due to the significant harvest in 1988 (see Table 7.1: RU5) (Lai & Bradbury 1998).

Even though the fishery has been in existence for over 40 years along the U.S. west coast, there still are knowledge gaps regarding the life history parameters of red sea urchins. The southern California fishery uses fishery-dependent catch data to estimate urchin abundance (Kato & Schroeter 1985), while the Washington fishery has used fisheries-independent data from diver and video surveys to perform stock assessments and determine catch quotas (Kalvass & Hendrix 1997), which allows for more accurate estimations of population size since the estimate is not confounded by fishing effort (see Table 7.1: RU5). Despite this, "the relationships between populations size and settlement/recruitment, and subsequent per capita survival and growth are unclear" (Table 7.1: RU1, RU2, RU5, and RU7) (Andrew et al. 2002). Research has shown that recruitment is density dependent, in that juvenile urchins require the presence of adult urchins for refuge, but also that a minimum adult density exists for successful spawning, with fertilization reaching negligible rates when adults are more than 1 to 2 meters apart (Kalvass & Hendrix 1997; California Department of Fish and Game 2003). Given the lower mortality rates among adult urchins, recruitment is considered to be the main limiting factor for population growth (Pearse & Hines 1987). Consequently, the fishery is prone to recruitment overfishing and a requirement for management plans to account for existing knowledge gaps is likely to help increase the sustainability of the fishery.

The growing effects of human-induced environmental change, and in particular that of OA, create knowledge gaps with respect to sustainability of the red sea urchin resource system. Scientists are working to learn more about how OA will affect coastal and marine ecosystems, but it is a relatively new field of study, with most research having occurred in the past decade (O'Donnell et al. 2008). Recent studies have shown that decreasing pH could impact multiple life stages of sea urchins, including a negative effect on fertilization success, higher mortality in larval development stages, increased mortality and decreased growth in juveniles, and a general decrease in adult calcification rates (Dupont et al. 2010; Kelly et al. 2013). Studies have also shown that while future OA levels have the potential to cause detrimental effects in sea urchins, these effects can vary between populations and sub-groups. This suggests that the genetic variation for successful adaptation to survival in elevated OA conditions could occur through spatially-varied selection (Dupont et al. 2012; Kelly et al. 2013). The predictions for future OA conditions indicate that detrimental effects on sea urchins will result reduce population size and growth rates among red sea urchins. In terms of fisheries, this suggests that the value of urchins will go up as the resource becomes scarcer and fishermen catch fewer and fewer sea urchins. In 1984, red sea urchins in southern California were valued at around $0.90 to $1.12 per kilogram[2] (Kato & Schroeter 1985), and since then their value has, on average increased (see Fig. 6.1). This is further supported by trends between 1989 and 2015 suggesting that price per kilogram and kilograms of catch landed are inversely related (Kalvass & Hendrix 1997; California Department of Fish and Wildlife 2015; Washington Department of Fish and Wildlife 2015).

Chapter 5. GOVERNANCE SYSTEM

In the early years of management of the southern California red sea urchin fishery, the only requirement for fishermen was that they needed to purchase a commercial fishing license from the California Department of Fish and Game (now the California Department of Fish and Wildlife, or CDFW). As of 1985, fishermen were required to possess an urchin permit issued by the CDFW

[2] Prices are in U.S. dollars and are adjusted for inflation calculated using Consumer Price Index annual average values (U.S. Bureau of Labor Statistics 2016)

(Kalvass & Hendrix 1997). Urchin permits were not location- or species-specific, but the catch contribution of purple sea urchins in California was negligible compared to red sea urchin landings. Then in 1987, the rise in the number of urchin fishermen caused the CDFW to place a moratorium on the issuance of new permits, at which time there were 938 active permits in both the northern and southern California urchin fisheries. In the same year that the permit issuance moratorium was imposed, the CDFW Director's Sea Urchin Advisory Committee (DSUAC) was established with the goal of fostering reactive, consensus-based management of the fishery by the CDFW and the California Sea Grant, the two leading authorities within California on sea urchin and kelp forest ecology (Kato & Schroeter 1985; Andrew et al. 2002). In 1989, the DSUAC set the minimum size for urchin catch at 76mm. In 1992, the minimum harvest size was increased to 83mm and the number of days each fishermen could work was limited to four days per week in winter months and three days per week in summer months (Kalvass & Hendrix 1997). The fishery peaked in 1988 and has been slowly declining since then. In 1990, as an effort to curb overfishing, fishery managers decided that one new entrant would only be allowed into the fishery after 10 urchin permits were retired (Andrew et al. 2002). Other management efforts in the fishery included the 2004 creation of the California Sea Urchin Commission, which was formed as a means to cohesively represent divers with urchin permits and processors of urchins to the managing authorities. The committee is still in place today, but solely represents urchin divers because in 2009 the processors voted to leave the commission.

The red sea urchin fishery in Washington is substantially smaller than that in California, but has been hailed for having "the most active and complex management history" among sea urchin fisheries worldwide (Kalvass & Hendrix 1997). Over the years, the Washington fishery has undergone continual changes to the management tools applied by the WDFW to the management of the resource in order to remain up to date with information collected about the status of the fishery. While the fishery opened in 1971, it was only in 1977 that management went beyond requiring fishermen to purchase an urchin permit and report their catch to the Washington Department of Fish and Wildlife (WDFW) (Andrew et al. 2002). Starting in 1977, the WDFW limited the fishing season to the winter months, and set catch size limits to protect urchins smaller than 102mm (the smallest 20% of the population), and larger than 133mm (the largest 20% of the population), though this was changed to 140mm in 1992 (Pfister & Bradbury 1996). The Washington fishery is the only U.S. west coast fishery to have set both an upper and lower catch size limit. The function of the upper size limit is to protect the larger individuals in the populations for their disproportionate reproductive output (reproductive output scales with test size) and for their spine canopies, which provide shelter to juveniles (Ulrich 2011). These size limits pertained to the harvest area around the San Juan Islands (Districts 1 and 2), and differed from the size limits established for other harvest districts in Washington where population size distributions were different (Hank Carson, personal communication). In 2011, the District 1 and 2 size limits were changed again due to decreases in the observed average size of urchins in the area, resulting in uniform size limits across all Washington urchin harvest districts. These uniform size limits are currently established with 83mm as the lower limit and 127mm as the upper limit (Ulrich 2011). Given that recruitment events in Washington red sea urchin populations are strongly episodic, the upper and lower size limit management efforts are aimed at increasing the size and success of spawning events.

Another management decision of the WDFW in 1977 divided Washington's waters in the Strait of Juan de Fuca and Puget Sound into roughly evenly sized sea urchin harvest districts. This allocation of fishing areas was made with the knowledge that while higher catch rates could occur if all harvest districts were accessible to fishermen every year, “a rotational fishery would maintain [urchin] populations at a level less likely to cause irreversible decline” (Pfister & Bradbury 1996). The districts maintained a rotational harvest plan that allowed an area two years of population recovery between harvest events. A study by Carter and VanBlaricom (2002) showed high variability of recolonization rates among sites, but that urchin population density could be recovered to between 37% and 69% of the original density a year after harvest. This management scheme was in operation from 1977 until the 1996 sea urchin fishing season, when a federal court case ruling, referred to as the ‘Rafeedie Decision’, reserved the rights for the Northwest Indian tribes to harvest half of all shellfish from their accustomed fishing areas each year in Washington State (US v. State of Washington 1994). In the case of red sea urchins, this decision legally affirmed the tribes’ rights to half of the State’s total allowable catch and meant that the WDFW could no longer impose their rotational harvest strategy on all urchin fishermen (Andrew et al. 2002; Carter & VanBlaricom 2002). While the five harvest districts still exist, they are now used only to report catch volumes.

The fishery boomed in the early 1980s, growing from 12 vessels in 1977, to 189 vessels in 1989 (Andrew et al. 2002). This led to an increase in WDFW management efforts in the late 1980s, which included adjustments made to the length of the season based on observations of roe quality, as well as the introduction of total allowable catch (TAC) quotas. These WDFW efforts are an example of the different types of management tools which can be used by resource managers in order to have a higher level of control over the proportion of the stock that is removed in a given year. Multiple management tools such as TAC quotas, seasonal closures and catch size limits can be used in conjunction with each other to better manage stocks, this can be seen in Table 5.1, which shows the management tool combinations used by the Washington and southern California fisheries. The Washington fishery first set a TAC limit in 1988, which was based on catch per unit effort data voluntarily reported by fishermen. This method was continued until 1994, when a size-based yield model was developed by the WDFW (Pfister & Bradbury 1996), and drop camera surveys were used to estimate size distributions and abundance (Hank Carson, personal communication). In 2010, remotely operated underwater vehicles (ROVs) were used to conduct a more comprehensive survey, and the resulting biomass estimate was very similar to the 1994 estimate, at just over 4 million kilograms in Districts 1 and 2, leaving the TAC quota unchanged. Using these survey results, the TAC was then set at 4% of the biomass estimate, where 4% is an arbitrarily amount decided upon by the WDFW as a conservative proportion of the stock as to avoid overfishing (Hank Carson, personal communication).

By 2002, the Washington state fishery was the only major sea urchin fishery in the world to have carried out a complex stock assessment (Andrew et al. 2002). In comparison, in 2016 the southern California fishery has not carried out more than a basic stock assessment, and still does not set any kind of harvest quota. This difference in management efforts could be attributed to the differences in the scale of two fisheries (see Fig. 6.1 for catch rates of both fisheries over the last 25 years), where the southern California fishery catch rates are still seen as high enough that catch limits are not seen as necessary. However, poor levels of understanding of the population size and dynamics (CA RU5), and declines in the number of urchins in the southern California fishery area suggests that the southern California fishery could benefit by adopting the use of additional management tools (see Table 5.1). While the Washington fishery is also experiencing declines in the total number of urchins caught, the decline is mostly accounted for by the Rafeedie Decision. The 1994 federal court ruling allocated of 50% of the state’s TAC to tribal fisheries, thereby decreasing the state-managed fishery TAC by 50%. As the tribal fisheries do not report their catch to the WDFW, trends in Washington state red sea urchin fishery catch rates reflect a significant decrease in catch in the 1995 fishing season onwards (see Fig. 6.1) which is not necessarily because fewer urchins were caught, but rather because of a decrease in the state-fishery TAC.

Table 5.1 – Comparison of management methods used in 2016 by the Washington and southern California red sea urchin fisheries (asterisks indicate the use of a management tool)

Management Tool	Washington	S. California
Size limit (mm)[1]		
Lower	83[2]	83[3]
Upper	127	
Seasonal closures	*	*
Harvest quotas	*	
Limited entry	*	*
Area closures	*[4]	

[1] Test diameter
[2] Washington size limits set in 2011 (Ulrich 2011)
[3] California size limits set in 1992 (Kalvass & Hendrix 1997)
[4] Effort was suspended in 1996 due to the Rafeedie Decision

Adapted from (Kalvass & Hendrix 1997)

The main governing authorities of the Washington and southern California red sea urchin fisheries are their respective DFW offices (GS1 and GS3 in Table 7.1). These departments are responsible for determining the operational, collective-choice, and constitutional rules (GS5, GS6, and GS7 in Table 7.1) of the urchin fisheries (Kato & Schroeter 1985; Kalvass & Hendrix 1997; Andrew et al. 2002), with the ratings for the Washington fishery not including the tribal fishery. The governance of the Washington fishery is co-managed by the WDFW and the tribes. Despite the ongoing process of determining the legal details of the co-management arrangement, tribal fishing effort is presently not focused on sea urchins because other dive fisheries such as sea cucumbers and geoducks currently are more lucrative (Hank Carson, personal communication). For the state-managed sea urchin fishery, the season is officially determined by the WDFW, who set the management year to begin on the 1st of September. However, it is really the fishermen who decide the season start date since the small size of the fishery is conducive to communication between individual permit holders, causing harvest to usually begin during mid-October. Licenses are limited to 1360 kilograms of catch per week, and the fishery remains open until the TAC limit is reached, or by the 1st of March, whichever occurs first. A combination of factors such as higher roe quality before spawning in the early spring months and high demand from the Japanese market in winter months often drives fishermen to increase their catch efforts to the point where the TAC quota is usually reached by late December or early January (Kalvass & Hendrix 1997; Hank Carson, personal communication).

Chapter 6. USERS

The red sea urchin fishery on the U.S. west coast is a small boat fishery. Each vessel usually has between one and three crew members, and fishing trips can last from a few hours to a couple of days depending on the size of the boat and the distance required to travel from port to the fishing area (Lai & Bradbury 1998; Chen et al. 2014). All fishing is done using low-pressure air compressors connected to a hose and a face mask (Kalvass & Hendrix 1997; Andrew et al. 2002). Boats search for areas with high urchin densities and check the roe quality of one urchin before beginning to harvest (Kato & Schroeter 1985). One or two divers then collect urchins using a short-handled rake to pry them off the rocks and into mesh bags or wire cages. These bags/cages are connected to the boat by a tether and have a rubber tube at the opening which can be inflated to maintain neutral buoyancy while collecting urchins. The average bag/cage is full when it contains around 50 to 70 kg of urchins, but some fishermen will continue to harvest up to 120 kg. Once the bag/cage is full, the rubber hose can be further inflated to lift the catch to the surface. Due to the strenuous nature of the working conditions, fishermen usually do around 3 to 4 hours of diving per day. However, vessels are also able to stay out for longer by trading off between multiple divers (Hank Carson, personal communication). The catch is taken back to port where fishermen sell it directly to consumers or processors. The urchins are usually processed domestically and then sold or exported mostly as fresh product, but some urchin roe is salted, steamed, baked or frozen (Kato & Schroeter 1985).

The southern California fishery has declined from 320 licensed urchin divers in 1998 to 164 in 2013 (California Department of Fish and Wildlife 2015). Along with the overall decrease in catch totals, the fishery is not retaining its fishermen or attracting new entrants because of increasing operating costs (Chen et al. 2014) and a decrease in the average price per kilogram of red sea urchins from roughly \$3.33 in 1990 to \$1.52 in 2014, with the highest price in the last 25 years being \$4.12 per kilogram in 1994 (see Fig. 6.1) (California Department of Fish and Wildlife 2015). It is worth noting that the current value of red sea urchins in southern California is higher than the 1984 value when the fishery was still growing (see S. CA 1984 data point on Fig. 6.1). However, these declines in the number of fishermen does not mean that the use of the resource is more evenly distributed. For example, in 1995, 33% of fishermen caught 50% of landings, whereas in 2010, 56% of fishermen caught 90% of red sea urchin landings (Chen et al. 2014).

While the fishery is relatively small compared to fisheries for other target species, urchin fishermen in southern California are dependent on at least seven different ports in the region. Since fishermen and local processors are first to know the timing of urchin spawning events (i.e., when the quality of urchin roe decreases to a point it is not profitable to harvest), they often form local organizations that then can advocate to the California Sea Urchin Commission (CSUC) for season closures. The CSUC is made up of five elected commissioners (four from southern California fishery districts and one northern California district), and two non-voting commissioners (one from the CDFW and one from the California Sea Grant) (California Sea Urchin Commission 2015). The opportunity for fishermen to contribute to the management of the fishery is important because there is a high economic dependence of fishermen on the red sea urchin fishery (Chen et al. 2014), and given that the gear used is specifically for a dive fishery, it is hard for urchin fishermen to diversify into other fisheries except for the sea cucumber dive fishery (Chen 2014).

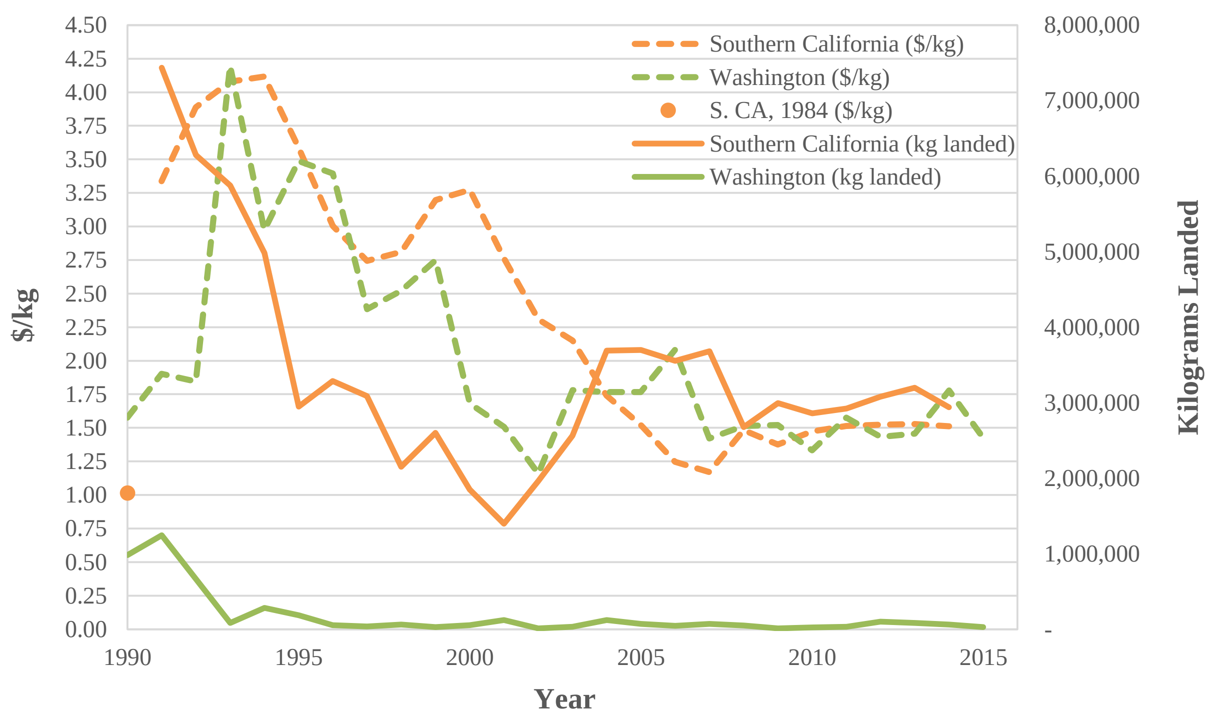

Figure 6.1 – Annual average value of red sea urchins per kilogram and annual catch values (kilograms landed) for the commercial fisheries in the Channel Islands, southern California (1991 – 2014) and San Juan Islands, Washington (1990 – 2015)

Similar to the southern California fishery, the number of fishermen in the Washington red sea urchin fishery decreased by roughly half in a comparable recent time frame, i.e. 2000 to 2015. However, it is important to note that the southern California fishery data were reported in terms of individual licensed fishermen, and though the Washington data are reported using the same terms, the WDFW does allow individuals to hold more than one license. The WDFW also allows fishing vessels to carry up to two licenses on board per fishing trip, but limits the number of divers in the water at any given time to one per license held aboard the vessel. Fishermen are also allowed to temporarily lease their licenses to others. Between 2000 and 2015, the number of non-tribal sea urchin permits decreased from 42 to 23, while the number of boats in the fishery decreased from 16 to 6, and the 23 permits in 2015 were held by 17 individuals (Hank Carson, personal communication). Given the flexibility of permit leasing, these numbers represent the lowest possible number of fishermen in the state regulated red sea urchin fishery in Washington. In 2000, the WDFW began a license buyback program in order to decrease the total number of existing licenses. The program purchased urchin licenses from fishermen who were no longer actively taking part in urchin harvesting. Between 2002 and when the program ended in 2013, the WDFW bought back 19 licenses (Hank Carson, personal communication). The decrease in the size of the Washington fishery can be attributed to similar causes as those for the southern California fishery. The average price per kilogram for red sea urchins in Washington decreased from $1.57 in 1990 to $1.43 in 2015. The price per kilogram did spike around 1993, where the price reached $4.19 per kilogram (see Fig. 6.1). The total catch for the San Juan Islands in Washington has fallen significantly in the last 25 years, from just over 900,000 kilograms of urchins harvested in 1990 to around 63,500 kilograms in 2014 (see Fig. 6.1) (Washington Department of Fish and Wildlife 2015). Accompanying this decrease has been the diversification of many fishermen into other dive fisheries, such as the sea cucumber and geoduck fisheries. In the WDFW managed fishery there are two main types of harvester, those with smaller vessels (9 to 12 meters) who stay in calmer waters and usually only have one urchin permit on board, and those with larger vessels who carry two urchin permits and take part in multiple dive fisheries in both Washington and Alaska. The non-tribal fishermen that harvest around the San Juan Islands mostly bring their landings to the port in Anacortes, though some fishermen are based at the Friday Harbor port on the islands and have to transport their catch to mainland processors by ferry (Hank Carson, personal communication).

The ratings for the users variables are potentially the most susceptible to bias compared to the other components in Table 7.1. This is due to a limited number of citable sources that provided information on selected sub-groups and did not necessarily address all aspects of each of Ostrom's variables. For both the southern California and Washington state fisheries, there were sources stating how fishermen are the ones first to know what urchin roe quality conditions are, and that fishermen communicate within their communities (see U7 in Table 7.1: local ecological knowledge, knowledge sharing). However, the few sources that discuss users were either potentially outdated or only briefly mentioned the current conditions of users in passing rather than specifically stating results of social research. For the Washington fishery, the available information explicitly discusses only the non-tribal users within the fishery, thus the ratings do not account for any information regarding the tribal fishery users.

Chapter 7. RESULTS

Both fisheries have been the focus of many studies over the course of the last few decades, resulting in some level of general understanding of the ecology of the resource systems and units. This can be seen in sections (a) and (b) of Table 7.1, where half of the variables were assigned at least one score of 3, indicating that the variable has been studied to a level of understanding which allows for management decisions to be founded on current scientific knowledge. However, this knowledge may not address uncertainties and knowledge gaps related to setting ecological goals for decreasing vulnerability and increasing sustainability of the fishery. In addition, 13 out of 16 variables had at least one score of 2, suggesting that many uncertainties and knowledge gaps still exist within the area of these variables. These results suggest that even for well-managed fisheries knowledge gaps and uncertainties are bound to exist, and thereby the effectiveness of management plans geared towards addressing vulnerability and sustainability may potentially be limited. Variables such as RS5 – RS8 received scores of 2 for both the institutional and confidence scores in both the southern California and Washington fisheries because the knowledge needed to fully understand these variables includes understanding the drivers of large-scale processes like ENSO, as well as the impacts of climate change on ecological communities, such as the effects of OA, warming ocean temperatures, and sea level rise. The uncertainties surrounding the topic of climate change are also the cause for the lower scores of resource unit variables. The total scores for the Resource Systems variables, section (a) of the table, are almost the same for both southern California and Washington, with the dissimilarities caused by the geographic differences of the two regions (e.g. RS3 and RS9). The southern California fishery encompasses the Channel Islands, which are somewhat dispersed, whereas the Washington fishery region comprises of the San Juan Archipelago which is less dispersed. This difference results in the California fishery encompassing a patchwork of smaller habitats where urchins are found, while the Washington fishery can access more continuous habitat.

Within Resource Units, three variables received a score of 1. While the lack of understanding regarding the effects of climate change did contribute to these lower scores, these variables scores also represent knowledge gaps that require a higher relative level of effort to fill (see Fig. 3 in Ekstrom et al. 2015). For example, to fully understand RU5 – number of units, time-intensive population surveys would need to be carried out, along with studies to determine the drivers of red sea urchin population dynamics in each fishery's region. It is worth noting that overall, Resource Units received the same totals for the institutional and confidence scores for both southern California and Washington despite the individual variables having multiple differences in scores. The scores mostly vary because of differences in urchin life histories (e.g. RU2 and RU3) between the two fishery regions caused by the different oceanographic conditions. However, management efforts, such as the WDFW population surveys and modeling, also result in different ratings (e.g. RU5). There was no mention in the literature of why the price per kilogram for red sea urchins differs so much between the two fisheries, though differences in roe quality due to growth rates and food availability could be large enough to account for the large gap in the prices (e.g. RU4).

The institutional aspects of resource management are depicted under the Governance (c) and Users (d) sections of Table 7.1, where in comparison to the ecological aspects, less variance is exhibited between the two fisheries. For the most part, in Table 7.1, the rating a variable received for the confidence score is a good indication of what rating it would receive for the institutional score, because higher consensus for a variable generally indicated that the variable had a higher level of understanding and fewer knowledge gaps. Differences in ratings for the two fisheries were mostly due to the availability of information or inherent differences among the regions. However, the ratings within the Governance section of the table do not follow this pattern, whereby the total governance system confidence score for the southern California fishery is lower than the institutional score. While the literature clearly reports on the organization of the governance system in place for the fishery, there was very little information available about how the governance structure and methods have changed since the inception of the fishery. This lack of current knowledge surrounding the governance system of the southern California red sea urchin fishery resulted in the confidence scores for multiple variables having a score of 2 while the respective institutional scores received a score of 3. This is contrasted by the Washington fishery having a history of consistently updating the structure and rules of their governance system, as well as consensus between a greater number of sources, and thereby leading to confidence scores that are the same as or higher than their respective institutional scores. For both states, variables GS2 and GS4 stand out as having lower ratings across both the institutional and confidence scores. While some information is available about property-rights (GS4) of red sea urchins – especially regarding the licenses needed by commercial fishermen in both states and the tribal rights in Washington – there are a limited number of sources discussing the property-rights systems for sea urchins as managed resource. There is even less information available about GS2 – nongovernment organizations (NGOs) related to the two sea urchin fisheries, with limited sources for each fishery briefly noting the existence of few NGOs.

The integrated assessment presented in Table 7.1 depicts the many similarities that exist between the two fisheries in the context of the status of their management efforts and the status of the managed ecosystem. This can especially be seen in the Users section (d) of the table where southern California and Washington scored identically for both the institutional and confidence variables. Despite differences in the numbers of users and the organization amongst users, both fisheries face similar challenges, such as decreasing numbers of fishermen (U1), increasing operating costs (U2), and limited knowledge sharing (U6 and U7). Much information regarding the users of the red sea urchin resource system comes from each state's respective DFW office, where records are kept of the number of fishermen active in the fishery in terms of number of sea urchin harvesting licenses, the catch reported per license, and the ports where catch is offloaded. These data indicate that variables U1, U3, and U4 are well managed, and thereby were assigned institutional and confidence scores of 3 for both fisheries. The variable U9 – technology used also received scores of 3 for both states and both types of score due to the technology in the commercial sea urchin fishery mostly remaining unchanged since the 1970s. Variables U5 – U8 were assigned lower scores because limited information was available regarding these aspects of the resource user group, as well as an indication of some understanding, however, there are still knowledge gaps. Lastly, variable U2 – socioeconomic attributes of users had almost no information available. Each fishery has one source that stated that operating costs are increasing and that these increases have played a role in the decrease in the number of fishermen. The combination of information suggesting that users may not have a good socioeconomic status, as well as the restricted number of sources resulted in both fisheries receiving scores of 1 for both institutional and confidence scores.

In general, the Resource System and Resource Units sections are dominated by scores of 2, the Users sections has an almost even split between scores of 2 and scores of 3, and the Governance System section is mostly scores of 3. This rough assessment suggests that there is a better understanding of the social and economic aspects of these two fisheries compared to the ecological aspects which have higher levels of uncertainty and more knowledge gaps. While looking at the ratings in Table 7.1, it is important to acknowledge that the two different scores are linked in a way that requires more knowledge or consensus on existing knowledge (i.e., attaining/maintaining higher confidence scores) before the ratings given to the institutional scores can be reevaluated. This can also be applied in the converse direction as a variable in the framework can be given a rating of 3 for an institutional score, thus indicating that the variable has a high level of understanding, but as circumstances affecting that variable change, both the institutional and the confidence scores for that variable will need to be reassessed.

Table 7.1 – Second-tier variables from Ostrom's (2007) framework for analyzing a socio-ecological system. Scores are indicative of status (institutional score) or level of knowledge/understanding (confidence score) for each variable (3 = good, 2 = okay, 1 = poor, 0 = insufficient). Colors are used to facilitate ease of comparison between scores. Bold numbers at the end of each section represent the total score for the component.

	Variable Definitions	**Institutional Score**		**Confidence Score**		**References (CA)**	**References (WA)**
		CA	**WA**	**CA**	**WA**		
(a) Resource System (RS)							
RS1 – Sector	Type of ecosystem(s) where resource is found	3	3	3	3	[13, 30]	[53]
RS2 - Clarity of system boundaries	Ecologically defined, e.g. watershed, habitats, etc.	3	3	3	3	[5, 13, 30]	[3, 53]
RS3 - Size of resource system	Geographical area size, shape and configuration	2	3	3	3	[5, 13, 30]	[3, 47, 53]
RS4 - Human-constructed facilities	Built structures that facilitate or impede ecological movement	0	0	1	0	[3]	
RS5 - Productivity of system	Resource dynamics (e.g. water, light, nutrients), community/species composition	2	2	2	2	[3, 22, 30]	[5, 38]
RS6 - Equilibrium properties	Successional stage/trajectory, alternative stable states, frequency/intensity of disturbances	2	2	2	2	[5, 30]	[47]
RS7 - Predictability of system dynamics	Uncertainty of driving forces (e.g. disturbances/population dynamics)	2	2	2	2	[5, 13, 30, 32]	[30, 35, 47]
RS8 - Storage characteristics	Nutrient source-sink dynamics	2	2	2	2	[15]	[15, 38]
RS9 - Location	Connectivity to nearby ecosystems	3	2	3	2	[30]	[47]
		19	**19**	**21**	**19**		
(b) Resource Units (RU)							
RU1 - Resource unit mobility	Inflows/outflows of individuals, patch dynamics	2	2	3	2	[11, 22, 32, 50]	[11]
RU2 - Growth or replacement rate	Reproductive maturity age, effective population size and reproduction rate, migration patterns	2	1	2	2	[3, 11, 22, 30, 32, 46, 50]	[3, 11, 30, 35, 47]
RU3 - Interaction among RUs	Competition, predation, trophic interactions	2	3	2	2	[5, 30, 32, 50]	[3, 35, 47]
RU4 - Economic value	Absolute/relative value	3	2	3	3	[7, 30, 32, 50]	[35, 61]
RU5 - Number of units	Population size and dynamics	1	2	1	2	[3, 11, 30, 32]	[3, 11, 30, 47]
RU6 - Distinctive markings	Natural/artificial physical markings	3	3	3	3	[30, 32]	[30, 32]
RU7 - Spatial and temporal distribution	Spatial patchiness, phenology	2	2	2	2	[3, 5, 11, 30, 32]	[3, 11, 35, 47]
		15	**15**	**16**	**16**		

	Variable Definitions	Institutional Score		Confidence Score		References (CA)	References (WA)
		CA	WA	CA	WA		
(c) Governance System (GS)							
GS1 - Government organizations	Federal/state/local, enforcement/funding support, restoration efforts	3	2	3	3	[3, 6, 30, 32]	[3, 10, 11, 47, 53]
GS2 - Nongovernment organizations	Environmental, research, social/welfare, restoration efforts	1	1	1	1	[12, 30]	[3, 4, 11]
GS3 - Network structure	Number of levels, transparency, connectivity	3	3	2	3	[6, 32]	[10]
GS4 - Property-rights systems	Public, private, common, mixed	2	2	2	2	[32]	[10, 11, 30]
GS5 - Operational rules	What people are: allowed to do, required to do, and prohibited from doing	3	3	3	3	[3, 6, 30, 32, 50]	[3, 11, 30, 35, 47, 53]
GS6 - Collective-choice rules	Who makes the rules, how rules are established/changed	3	3	2	3	[3, 6, 30]	[11, 30, 47, 53]
GS7 - Constitutional rules	Who participates in the political system, how they are chosen, what authority do they have	3	3	2	3	[3, 6, 30]	[11, 30, 47,53]
GS8 - Monitoring and sanctioning processes	Social, biophysical	3	3	2	3	[3, 6, 30]	[3, 10, 30, 35, 47]
		21	**20**	**17**	**21**		
(d) Users (U)							
U1 - Number of users	Commercial, recreational, tribal	3	3	3	3	[3, 7, 13, 30, 32]	[3, 10, 30,47]
U2 - Socioeconomic attributes of users	Socioeconomic resilience, operating costs	1	1	1	1	[13]	[10]
U3 - History of use	Duration, crisis	3	3	3	3	[3, 13, 30, 32]	[3, 11, 30]
U4 - Location	Built infrastructure (e.g. ports, harbors), non-built/natural access (e.g. beaches)	3	3	3	3	[13, 32]	[10]
U5 - Leadership/ entrepreneurship	Local/regional	2	2	3	3	[3, 12, 30]	[35]
U6 - Norms/social capital	Spatially based (e.g. clubs), non-spatially based (e.g. online, publications)	2	2	2	2	[7]	[10]
U7 - Knowledge of SES/mental models	Traditional/local ecological knowledge, knowledge sharing	2	2	2	2	[30, 32]	[35]
U8 - Importance of resource	Economic/cultural dependence	2	2	2	2	[3, 13, 30, 32]	[10]
U9 - Technology used	Type, homogeneity	3	3	3	3	[3, 12, 13, 30, 32]	[3, 10, 35]
		21	**21**	**22**	**22**		

Chapter 8. DISCUSSION

Overall, the application of Ostrom's SES framework for analyzing the sustainability of managed resources allowed me to identify areas of the southern California and Washington red sea urchin fisheries that appear to require additional research or effort in order to maximize the sustainability and minimize the vulnerability of these fisheries under future climate change conditions. While the two fisheries included in my assessment are among the most actively managed sea urchin fisheries in the world, my assessment does call attention to the existence of knowledge gaps and identifies differences between the sustainability and vulnerability of these two fisheries. The differing locations for the two fisheries means that the effects of climate change will vary, with southern California being more affected by the increase in frequency and intensity of ENSO events, while Washington will be more affected by intensifying OA conditions. Similarly, though the fisheries are managing the same species, there is enough variance between the two resource systems that knowledge regarding the red sea urchins as resource units cannot be shared by the two fisheries without considering the inherent differences between populations. Given that the fisheries are different sizes in terms of both number of fishermen/boats and kilograms of catch landed, it suggests that there is a potential for the two management plans to diverge in the future. For example, in Washington the implementation of a more individual approach, such as the use of individual fishing quotas, would give a higher degree of control of the TAC to the WDFW.

The management history of the Washington fishery involves more updates and adjustments than that of the southern California fishery, indicating that the WDFW management approach is more adaptive, and thereby potentially decreasing the vulnerability of the fishery to future climate change. The Washington fishery also has benefitted from the division of the TAC between the tribes and the state. This splitting of the TAC, and the fact that the tribal fishery is currently not actively harvesting red sea urchins, has resulted in the fishery being more sustainable as it is less susceptible to overfishing. However, this aspect of the sustainability of the WDFW managed fishery is not represented in Table 7.1 because my analysis did not include the tribal fishery. While the southern California and Washington fisheries have active and well-recorded management histories, both fisheries have knowledge gaps that hinder the ability of resource managers to build resilience into the fishery management plans. My analysis could be extended and expanded through a socioeconomic study of the users, studies clarifying the potential impacts of climate change on both the resource systems and resource units, and the possibility of developing a management plan that accounts for the social, economic, and ecological factors that affect red sea urchin fisheries.

My motivation for this analysis was to determine whether Ostrom’s framework could be applied to fisheries that are likely to be affected by climate change, and if so, whether such an analysis would yield helpful insights into management practices to foster sustainability. I found that using Ostrom’s framework I was able to compare variables across the two fisheries, but the analysis is subjective and does not yield the same level of understanding that could be provided by a complete quantitative analysis such as that performed by Ekstrom et al. (2015). Despite this, my analysis of red sea urchin fisheries in southern California and Washington suggests that they have many similarities, from licensing practices to gear restrictions and management structures. However, many differences still exist and they influence the vulnerability and the sustainability of the two fisheries. The smaller size of the Washington fishery indicates that less effort may be required to fill the knowledge gaps concerning the governance system and users of the resource. Conversely, the southern California fishery is located in a region with a high capacity for research to reduce the knowledge gaps related to their resource system and resource units. Both fisheries are facing different stressors caused by climate change; Washington has a greater incentive to determine the potential impacts of OA on their fishery, while southern California will most likely address threats from ENSO events and increasing ocean temperatures. Moving forward, each of these fisheries should work towards filling existing knowledge gaps and understanding the potential impacts that climate change may have on their resource system in order to maximize the resilience of their managed resource.

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