

**HARBOR SEAL ABUNDANCE AND HABITAT USE RELATIVE TO  
CANDIDATE MARINE RESERVES IN SKAGIT COUNTY, WASHINGTON**

by

Adria S. Banks

Accepted in Partial Completion  
of the Requirements for the Degree  
Master of Science

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A Thesis  
Presented to  
The Faculty of  
Western Washington University

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In Partial Fulfillment  
Of the Requirements for the Degree  
Master of Science

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**ABSTRACT**

Marine reserves manage fisheries by increasing abundance and size of exploited species within their boundaries and in adjoining, non-protected sites. However, their effectiveness might be compromised if predators increase their numbers (aggregative response) or foraging frequency (functional response) in the reserve rather than in adjacent sites. The previous scenario assumes that predators respond to increased fish density at small spatial scales. In Skagit County, Washington, where harbor seals are year-round residents, several candidate marine reserves for rocky reef bottomfish conservation have been recommended by the Skagit County Marine Resources Committee. This study addressed the following questions: 1) What is the number of harbor seals in and around Skagit County candidate marine reserves?; 2) Is there seasonal and annual variability in seal numbers and distribution?; 3) Do harbor seals respond to differing prey distributions on a small spatial scale ( $< 1 \text{ km}^2$ )?; and 4) How do harbor seals utilize a candidate marine reserve (Burrows Channel) prior to protection? To address the first two questions, I conducted aerial surveys to count the number of seals within a geographic area encompassing eight candidate marine reserves. To address the last two questions, I a) described harbor seal presence relative to different bottomfish distributions in Burrows Channel by conducting land-based observations of seals coupled with SCUBA surveys; b) conducted focal observations to record the

locations of seal predation events and identify prey species; and c) tracked individuals and opportunistically photographed seals to determine the time spent in Burrows Channel within a 4-hr observation period, whether individuals revisited the channel, as well as seal behavior and frequency of occurrence while in the candidate reserve. Throughout Skagit County, the number of seals hauled out increased from April ( $1,249 \pm \text{SD } 193$  seals) to September ( $2,302 \pm \text{SD } 120$  seals), with more seals ashore during the pupping season than during pre-pupping months ( $t_3 = 5.144$ ,  $p = 0.01$ ). Haulout use varied annually (when compared with published seal haulout locations) and seasonally, the latter was in part due to haulout availability in relation to tidal levels. In Burrows Channel, bottomfish density was higher at Site A ( $461.5 \pm \text{SD } 115.4$  fish/ha) than Site B ( $239.6 \pm \text{SD } 44.2$  fish/ha; linear mixed-effects:  $\text{AIC} = -241.6904$ ,  $p < 0.001$ ), and the number of seals near Site A was greater than near Site B ( $\chi^2_1 = 7.53$ ;  $p = 0.006$ ). No bottomfish were recorded as harbor seal prey during focal observations; rather, small pelagic fish were the principally observed prey. Seal sightings were not evenly distributed in the channel: areas with high frequencies of seal sightings were associated with a strong tidal rip, shallow bottom depths, and locations of predation events. Seals spent up to 3.5hrs in Burrows Channel, where they predominantly milled, which is indicative of foraging behavior. Additionally, individual seals returned to specific areas within the channel through the study period. The potential impact of harbor seals on bottomfish populations is currently unknown. However, results indicate that the number of seals ashore and their distribution changed on a seasonal and annual basis throughout Skagit County and that their habitat use varied at very small spatial scales. Hence, this study sets the stage for continued monitoring of these sites if protected status is awarded, providing a critical baseline for the eventual examination of aggregative and functional responses of predators.

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# INTRODUCTION

## Background

The world's marine ecosystems are considerably less healthy due largely to various human activities, particularly fishing. In fact, some studies suggest that traditional commercial fishing practices may actually cause these ecosystems to collapse (Jackson *et al.* 2001, Pauly and Watson 2003, Worm *et al.* 2006). Many federal and local governments have instituted marine protected areas (MPAs) to protect and conserve renewable marine resources as well as to provide an additional option to existing fisheries management. MPAs are attractive management tools because there tends to be rapid increases in biomass, abundance, and average size of exploited species after an area is protected from fishing (Halpern and Warner 2002, Halpern 2003). Furthermore, protected areas appear to augment adjoining fisheries through 'spillover' of fish biomass as well as the transport of larvae to unprotected sites (Roberts *et al.* 2001, Gell and Roberts 2003). Despite the promise of MPAs, the development and implementation of this type of ecosystem-based management is relatively new and there is substantial need for information. One aspect that merits additional study is the potential for predators to impact prey species in protected areas.

Predators can affect the abundance, size distribution, habitat use, and biodiversity of organisms in protected areas. For instance, sea otters (*Enhydra lutris*) negatively affect the number and size-distribution of red abalones (*Haliotis rufescens*) in marine protected areas off the coast of California (Fanshawe *et al.* 2003). Additionally, a study modeling interactions between fish and seals predicted that such predator-prey interactions would reduce the benefit of a MPA to human fishers (Boncoeur *et al.* 2002). Due to the increase in size and quantity of species in MPAs, one would expect predators to respond by increasing

their abundance (aggregative response) and frequency of foraging (functional response; Solomon 1949, Holling 1959) in these areas of increased prey density. Although research endeavors are increasing our understanding of the inherent complexities of marine systems and the management thereof, I am not aware of any study measuring the aggregative or functional responses of top marine predators relative to the establishment of a marine protected area.

### **Marine reserves**

No-take MPAS (also known as marine reserves) are a specific type of protected area defined as “areas of the ocean completely protected from all extractive and destructive activities”, hence differing from MPAs, which generally allow some level of harvest (Lubchenco *et al.* 2003). Despite this difference both conservation areas focus on ecosystem-based management rather than traditional single-species management. Currently, there are 74 marine managed areas found in Washington State (NOAA 2007). Although many of these areas do not qualify as marine reserves, seven “complete no-take” reserves exist in southern Puget Sound (Palsson 2002). Of these sites, Brackett’s Landing Shoreline Sanctuary Conservation Area (Edmonds Underwater Park) has been protected since 1970 and is the state’s oldest established marine reserve and sanctuary. Studies conducted by Washington Department of Fish and Wildlife (WDFW) at the Edmonds conservation area demonstrate that protection from harvest resulted in increases in abundance and size of fish (Palsson 2001).

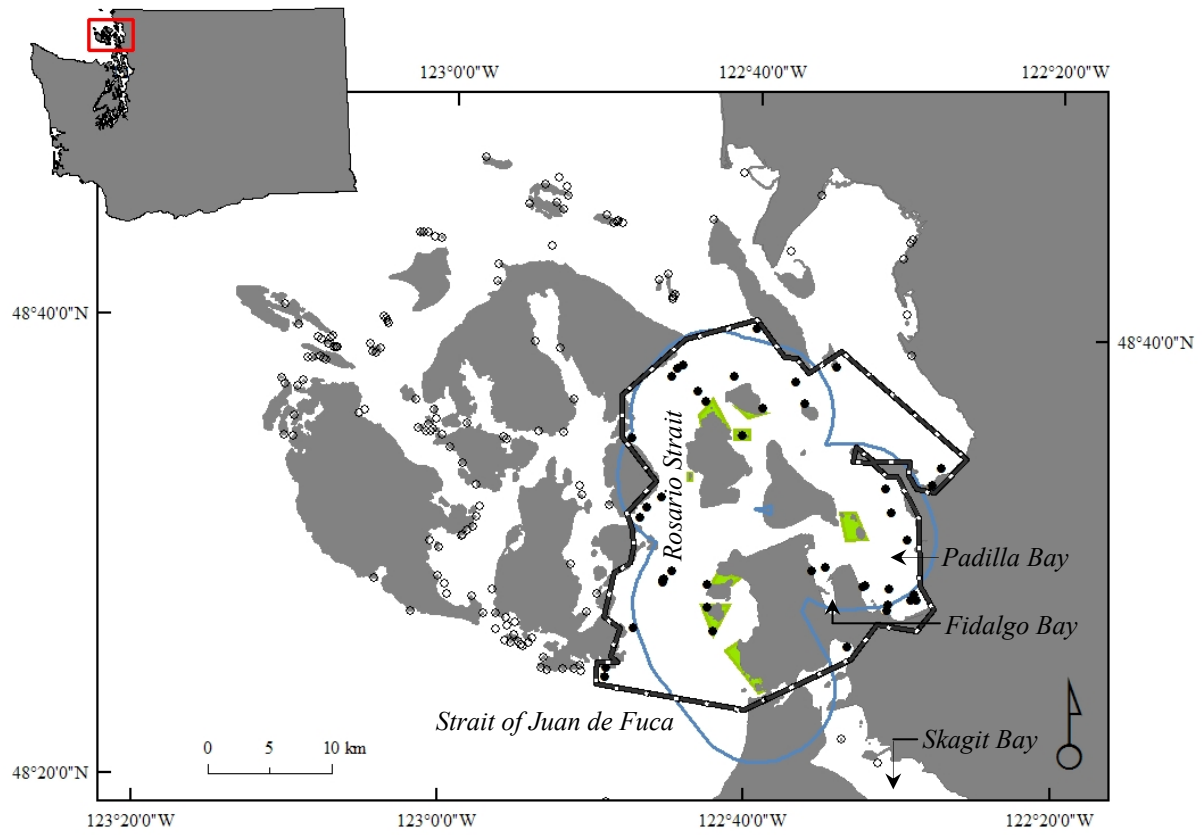
There are several established marine reserves and rockfish recovery areas in the San Juan Islands, Washington, that currently range in maturity from approximately 10 years to

more than fifteen years (Tuya *et al.* 2000). Recently, the Skagit County Marine Resources Committee selected eight candidate sites for future no-take reserves in the eastern San Juan Islands Archipelago (Figure 1; McConnell and Dinnel 2002). From this “long list” of eight potential locations, two to four sites may eventually be selected for marine reserve status (Dinnel, personal communication<sup>1</sup>). The possible establishment of Skagit County reserves is motivated by a grass roots recovery plan for rocky reef bottomfish, particularly rockfish species (Scorpaenidae) and lingcod (*Ophiodon elongatus*), but also kelp greenling (*Hexagrammos decagrammus*) and other species (McConnell *et al.* 2001, McConnell and Dinnel 2002).

Rocky reef bottomfish are excellent candidate species for habitat-based reserve protection. They are typically long-lived species (50-100yrs; Love *et al.* 2002) that require complex, high-relief substrate, especially as adults (West 1997, Pacunski and Palsson 2001). Because of their reliance on habitat, many rocky reef bottomfish also have restricted home ranges; for example, 95% of re-captured quillback (*Sebastes maliger*) and copper rockfish (*S. caurinus*) tagged in Puget Sound were recovered within 300m of their tagging site (Mathews and Barker 1983). However, studies also show that individual home range size is variable and often depends on the quality of the habitat (Matthews 1990). Because rockfish life history traits differ from many other managed stocks, MPAs are among the management strategies recommended by the American Fisheries Society and others (West 1997, Palsson *et al.* 1998, Yoklavich 1998, Parker *et al.* 2000). Additionally, existing local protected areas have been successful in increasing the size and density of rocky reef bottomfish within their boundaries (Palsson 2001, Eisenhardt 2002). With the possibility for the creation of no-take

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<sup>1</sup> Paul Dinnel; Shannon Point Marine Center; 1900 Shannon Point Road; Anacortes, WA 98221; January 26 2006.



**Figure 1.** The study site in the eastern San Juan Islands Archipelago. The extent of aerial surveys is indicated by the bold dashed line. Candidate marine reserves recommended by Skagit County Marine Resources Committee are shown in green boxes (McConnell and Dinnel 2002). The rounded blue line shows a 5.6km buffer around all the candidate reserves. Circles indicate known haulout sites of harbor seals (Jeffries *et al.* 2000; black circles were surveyed, open circles represent un-surveyed haulouts outside the study area).



areas in the eastern San Juan Islands, the need for baseline information on predator abundance and foraging habits is critical.

### **Marine mammal predators**

Pinnipeds (seals, sea lions, and fur seals) are considered a threat to local fisheries in many areas (Harwood and Croxall 1988, NMFS 1997, Bjørge *et al.* 2002). They have also been implicated as one factor in the failure to recover of Atlantic cod (*Gadus morhua*) stocks off eastern Canada (Bundy 2001, Fu *et al.* 2001, Trzcinski *et al.* 2006). California sea lions (*Zalophus californianus*), Steller sea lions (*Eumetopias jubatus*), and northern elephant seals (*Mirounga angustirostris*) are found in Washington State for only part of the year (Jeffries *et al.* 2000). Harbor seals (*Phoca vitulina richardsi*), however, are the most widely distributed and abundant pinniped species in Washington State; additionally, they are the only pinniped that breeds locally and is found year-round in the San Juan Archipelago (NMFS 1997, Jeffries *et al.* 2000). Hence, I focused my study on harbor seals given their great potential to affect local fish stocks in the San Juan Islands.

### **Harbor seals: Diet**

Harbor seals are mobile, opportunistic predators that consume seasonally or locally abundant prey, which include both commercial and non-commercial fish species (Olesiuk 1993, NMFS 1997, Browne *et al.* 2002). In Puget Sound, harbor seals are reported to prey on Pacific staghorn sculpin (*Leptocottus armatus*), Pacific whiting (*Merluccius productus*), salmonid species (Salmonidae), Pacific herring (*Clupea pallasii*), Pacific sand lance

(*Ammodytes hexapterus*), Pacific tomcod (*Microgadus proximus*), walleye pollock (*Theragra chalcogramma*), flatfish species (Pleuronectidae), shiner perch (*Cymatogaster aggregata*), northern anchovy (*Engraulis mordax*), plainfin midshipman (*Porichthys notatus*), as well as squid and octopi (Scheffer and Sperry 1931, Everitt *et al.* 1981, NMFS 1997, Zamon 2001).

A recent study in the San Juan Islands reported that rockfish species and hexagrammids (lingcod and greenling) were found in 2.3% and 1.0%, respectively, of 507 harbor seal scat samples (Lance and Jeffries 2006). Similarly, rockfish (2.3%) and hexagrammids (2.8%) cumulatively were present in less than 6% of 2,917 samples collected year-round in the Strait of Georgia, British Columbia (Olesiuk *et al.* 1990a). Although bottomfish are apparently not a primary prey in the San Juan Islands and the Strait of Georgia, they are important prey further south, where the prevalence of rockfish in harbor seal scat samples has ranged between 4-22% in Oregon and 19-50% in California, depending on location and season (NMFS 1997, Orr *et al.* 2004, Wright *et al.* 2007). The potential impact of harbor seals on bottomfish populations is currently unknown; however, even though they do not appear to consume many bottomfish locally, they could negatively impact these populations at low predation levels if harbor seal numbers are large enough.

### **Harbor seals: Foraging scale**

Marine mammals forage on different temporal and spatial scales depending on the availability of their prey and their bioenergetic demands (Boyd 1996, Thompson *et al.* 1996, Coltman *et al.* 1997, Boyd 2002, Boyd *et al.* 2002). Harbor seals exhibit individual foraging patterns on varying spatial scales (Iverson *et al.* 1997, Suryan and Harvey 1998). While a few individuals exhibit large-scale movements associated with foraging, radio-tagging

studies suggest that most seals forage locally (Bjørge *et al.* 1995, Frost *et al.* 1995, Suryan and Harvey 1998). Harbor seals in the San Juan Islands typically forage within 5.6km of their primary haulout site and exhibit haulout and foraging site fidelity (Suryan and Harvey 1998). Local foraging patterns and the fact that seals exploit locally and seasonally available prey (Brown and Mate 1983, Olesiuk 1993, Thompson *et al.* 1996, Marston *et al.* 2002, Middlemas *et al.* 2006) suggest that local prey aggregations are important food sources.

Most studies on harbor seal predation in relation to prey aggregations have relied on movements of harbor seals in relation to the typical timing of prey migrations (e.g., Olesiuk 1993, Browne and Terhune 2003, Middlemas *et al.* 2006) or on predator diet studies to infer local prey abundance (e.g., Browne *et al.* 2002). Although a few studies have concurrently documented prey abundance and predator abundance in small regions (Tollit *et al.* 1997a, Marston *et al.* 2002), none have been conducted in areas less than 1km<sup>2</sup>. Bottomfish densities are being quantified within the Skagit County candidate reserves (Dinnel *et al.* 2003, Weispenning *et al.* 2004, Weispenning 2006, Valz 2007, this study), providing a unique opportunity to study harbor seal foraging in relation to differing bottomfish densities on a small spatial scale. Due to the territorial nature of many bottomfish species (McConnell *et al.* 2001, Love *et al.* 2002), understanding fine-scale foraging habits of harbor seals is important to address their potential responses to local abundances of bottomfish.

### **Research objectives**

In addition to consuming locally and seasonally available prey (Olesiuk 1993, Tollit *et al.* 1997a, Hall *et al.* 1998, Browne *et al.* 2002), harbor seals exhibit aggregative (Marston *et al.* 2002, Browne and Terhune 2003, Middlemas *et al.* 2006) and functional responses

(Middlemas *et al.* 2006) to prey pulses. Exploitation of high prey densities is an efficient foraging strategy for many predators (Cornick and Horning 2003, Enstipp *et al.* 2007), but could be problematic to management and conservation goals if harbor seals demonstrated similar responses to increased densities in reserve boundaries. Thus, baseline studies from which numeric and functional responses may be measured are critical to the assessment of community level effects and marine reserve efficacy.

My first research objective was to estimate the number of harbor seals in and around Skagit County candidate marine reserves. Abundance data in Washington State is typically collected only at the peak of the pupping season (Huber *et al.* 2001, Jeffries *et al.* 2003), thus seasonal variability and seasonal trends that may affect predator-prey interactions cannot be examined. By conducting surveys over a longer period, I attempted to gauge inter- and intra-annual variability of haulout site use at the regional scale as well as the influence of the environment on the temporal and spatial distribution of seals among individual haulouts. I thereby provided a robust baseline by which the potential for an aggregative response of predators can be measured after reserve protection.

The second aim of my project was to assess whether or not harbor seal activity was correlated with bottomfish densities at a small spatial scale ( $< 1\text{km}^2$ ), i.e., to use different bottomfish densities as a proxy for marine reserve protection to gauge potential aggregative and functional responses of harbor seals. Due to technological developments, many studies are now able to investigate movements and foraging patterns of mobile predators at large scales (Boyd and Arnborn 1991, Guinet *et al.* 2001, Bjørge *et al.* 2002, Bradshaw *et al.* 2002). However, information regarding fine-scale habitat use and foraging strategies, both temporally and spatially, is equally important to understand predator-prey dynamics.

The final goal of my project was to describe the use of a candidate reserve by a marine predator prior to protection, providing a baseline against which future studies may measure aggregative and functional responses of seals to bottomfish reserves. In addition to setting the stage for continued monitoring of these sites after protected status has been awarded, these three research goals contribute to our understanding of the fine-scale predator-prey dynamics that should be considered in marine reserve design and management.

## **METHODS**

### **Number of harbor seals in and around candidate marine reserves**

#### **Study site**

Harbor seals tagged in the northern San Juan Islands typically forage within 5.6km of their primary haulout site (Suryan and Harvey 1998). I buffered the Skagit County candidate reserves by this average foraging distance to identify haulouts from which seals would likely forage within or near the candidate marine reserves. Airplane-based surveys were flown over this 630 km<sup>2</sup> area, which encompassed the western part of Skagit County, Washington, the eastern San Juan Islands, and the eight candidate marine reserve sites (Figure 1). Surveys were conducted along coastal areas, including intertidal reefs and rocks, from approximately 48° 40' N to 48° 26' S, and 122° 28' W to 122° 49' E (Figure 1).

#### **Data collection**

A single-engine, high-wing plane was flown at approximately 1,000m altitude and 150km/h between two hours before to two hours after daytime low tides. Surveys were conducted during mid-day low tides when maximal numbers of seals are hauled out (Schneider and Payne 1983, Huber *et al.* 2001, Zamon 2001, Jeffries *et al.* 2003). When there were no mid-day low tides in a sampling period, I flew during morning low tides. Tidal heights were estimated from time of day using the program Tides & Currents Pro, ver. 2.5b (Nautical Software, Inc.).

Surveys were flown twice monthly between April and September, 2005 on consecutive days (weather permitting) during the lowest low-tide window each month.

These months include three life-history stages of harbor seals in the San Juan Islands: pre-reproductive season (April to mid-June), pupping season (mid-June through August) and molting season (mid-August through October; Huber *et al.* 2001). Because most harbor seals utilize haulouts during pupping and molting seasons (Thompson *et al.* 1997) and recent studies have focused on population trends (Jeffries *et al.* 2003), abundance data in the state of Washington are currently collected during the pupping season (Huber *et al.* 2001, Jeffries *et al.* 2003). By conducting surveys over a large geographic area and long temporal period, I was able to estimate the local seal population inside and within several kilometers of the candidate reserves as well as to measure interseasonal variability.

At each haulout site, I counted seals, including pups, on land. In addition, I recorded location, time of day, and cloud cover as variables that may affect the number of seals hauled out (Pauli and Terhune 1987b, Reder *et al.* 2003). Evidence of recent disturbance (many seals milling in the water or a high proportion of wet seals on land) was also noted. Visual counts often result in an underestimation of actual numbers of seals hauled out (Thompson and Harwood 1990). Hence, sites with greater than 20 seals were photographed using a hand-held digital SLR camera (Canon 10-D) carrying a 100-400mm image-stabilizing lens. Digital images were examined and individual seals were numbered (including pups) using Adobe Photoshop, ver. 7.0 (Adobe Systems Incorporated). All photographs were inspected by at least two people and photographic counts were then compared to visual counts to correct for possible underestimation.

## Data analysis

**Regional scale (interannual and intra-annual changes).** GPS locations of harbor seal haulouts were imported into a Geographic Information Systems (GIS) database along with haulout sites documented in the Atlas of Seal and Sea Lion Haulouts in Washington (Jeffries *et al.* 2000). The map I created was used to explore interannual changes in haulout site locations between historical surveys and aerial surveys in 2005, as well as interseasonal changes in individual haulout site use within 2005.

In addition to the spatial distribution of seals within the study area, I was interested in the total number of seals that utilize the study area encompassing the eight candidate reserves as well as temporal trends in seal numbers over the six months of surveys. Monthly counts from the two survey days were averaged for these analyses; when only a single count existed for a given haulout, due to weather or evidence of prior disturbance, that value was used as the monthly average.

Counts at low tide result in a minimum abundance estimate because seals under the water and away from haulout sites are not included in these tallies. To correct for the proportion of harbor seals not hauled out during pupping season, a previously estimated correction factor for the area (Huber *et al.* 2001) was applied to mean low-tide counts of seals ashore from this season. Following Huber *et al.* (2001) harbor seal abundance was estimated as:

$$N = n * C$$

where,

$N$ = total abundance of seals in the study area.

$n$ = mean low-tide count during the sampling period.



$C$  = combined correction factor for Washington and Oregon, based on the proportion of low tides on which males, females, subadults, and pups were hauled out during the pupping season ( $C=1.53$ ;  $SE=0.1$ ; obtained from Huber *et al.* 2001).

Correction factors are geographically and seasonally dependent, and in the study area they only exist for the pupping season (Huber *et al.* 2001), thus, only uncorrected counts of non-pups on land were used to examine interseasonal variability in the number of seals. Interseasonal variability was examined on two spatial scales: the entire surveyed area and haulout-specific comparisons.

A paired t-test was used to compare the number of seals hauled out during the pre-reproductive and pupping seasons. June and September counts were not included in this analysis as they represented “transition” periods at the onset and end of pupping, with only a few pups being born in June and seals beginning to molt by September. The paired t-test was performed using R, ver. 2.4.1 (R Development Core Team 2006).

**Haulout scale (influence of tidal availability).** To eliminate sites that were used occasionally by a few seals, only haulouts with a recorded average count of 20 or more seals in any of the survey months were included in haulout-specific analyses. Photographs of haulouts were used to categorize sites as “limited” ( $n=15$ ) or “unlimited” ( $n=20$ ). To be classified as tidally “limited” the haulout had to be fully (e.g., channels in bays) or mostly (e.g., small reefs and rocks) submerged at high tide; i.e., intertidal organisms appeared to cover the haulout’s surface. Haulouts that remained mostly uncovered at high tide (e.g., islands and large rocks) were coded as “unlimited”.

Mixed-effects models were used to explore whether haulout-site use varied over the survey period (April-September) with respect to tidal availability, while accounting for the repeated sampling of haulouts over time. Mixed-effects analyses are powerful and adaptable tests as they permit fixed and random effects to be estimated separately, and associated error terms to be included in models independently (Cnaan *et al.* 1997, Faraway 2005). Though similar to ANOVA and multiple linear regression, mixed-effects models allow for correlation between observations (i.e., repeated measures) and can handle unbalanced data sets (Cnaan *et al.* 1997, Pinheiro and Bates 2004), making the technique preferable for this study. Data for this analysis were square-root transformed to correct for non-normality and heterogeneous variances (Zar 1999).

I fitted a series of mixed-effects models to determine the best estimation of model parameters using maximum likelihood (ML) estimation. ML estimation is the preferred technique to compare models with different fixed effects (Faraway 2005). Haulout site by month was treated as a random variable and included in all models. Average seal counts were the response and month a fixed effect; tidal classification was added (as a fixed factor) to determine its influence on the model. Both linear and quadratic models were considered.

Models were tested in the following order: 1) linear *versus* quadratic for all haulouts regardless of tidal classification, and 2) excluding tidal classification *versus* including tidal classification. A quadratic model was tested because if harbor seal numbers increased in one season, a similar decrease would be expected at some point during the year as Washington's harbor seal population is currently close to its predicted carrying capacity and growing only slowly (Jeffries *et al.* 2003). Tidal classification was included because tide height influences haulout patterns (Schneider and Payne 1983, Pauli and Terhune 1987a, Reder *et al.* 2003,

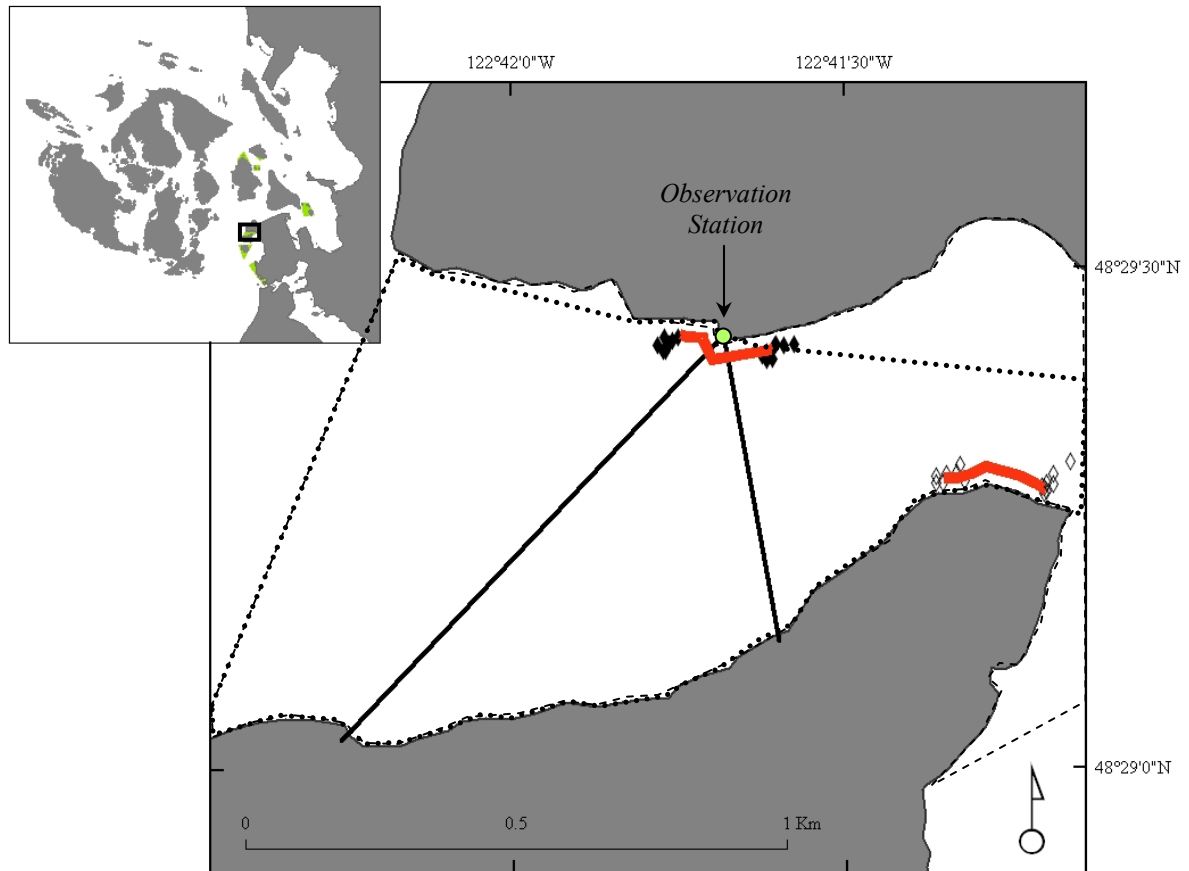
Hayward *et al.* 2005), and haulouts of differing tidal availability were surveyed in this study. I was also interested in the interaction of month and tidal classification as tidal extremes changed over the study period and would likely influence patterns of haulout use depending on the tidal availability of individual haulout sites. Akaike Information Criterion (AIC) scores were used to choose the most parsimonious model for each comparison (Burnham and Anderson 2001). The final mixed-model was fit using restricted maximum likelihood and an ANOVA test used to reveal significant effects.

The mixed-effect model analyses were performed using the linear and nonlinear mixed effects models (nlme) package (Pinheiro *et al.* 2006) of R, ver. 2.4.1 (R Development Core Team 2006). The lattice graphics package (lattice) was used to generate the figures associated with the mixed-effects analysis (Sarkar 2006). Lastly, monthly counts were summed for each haulout type within the study area to describe general trends by haulout type over the study period. Fifteen haulouts were randomly chosen from the "unlimited" classification to compare equivalent samples in this last case.

## **Bottomfish and harbor seal distribution within a candidate marine reserve**

### **Study site**

SCUBA surveys were conducted at two sites within Burrows Channel, one of the candidate marine reserves recommended by Skagit County's Marine Resources Committee. Divers entered the water at approximately 48° 29.417' N and 122° 41.605' W (Site A) or 48° 29.285' N and 122° 41.206' W (Site B; Figure 2). Site A and Site B experience similar tidal



**Figure 2.** Location of Burrows Channel candidate marine reserve in the San Juan Islands (inset). The detail map shows a partial extent of the candidate marine reserve (dashed line), visible area (dotted line) from the land-based survey observation station (○), dive routes (bold lines) as well as approximate start and end points of dives at Site A (◆) and B (◇), and the borders between sectors (bold lines).

regimes and are alike in bottom depth and rocky reef habitat. The easternmost point of Site A (near the north shore of Burrows Channel) is approximately 0.4km from the westernmost point of Site B (along the south coast of Burrows Channel).

Land-based observations of seals were conducted from a vantage point on the south side of Fidalgo Head in Washington Park, Anacortes, Washington at approximately 48° 29.438' N and 122° 41.682' W (Figure 2). Observer height was 13.8m above mean zero tide level. From this locale most of Burrows Channel was visible, including both the dive sites and 70% of the candidate marine reserve (or approximately 0.8km<sup>2</sup>). Burrows Channel is characterized by unequal, semi-diurnal tides.

### **Bottomfish distribution data collection**

To examine fine-scale differences in prey densities, bottomfish were counted at two sites (A and B) within a candidate marine reserve area (Figure 2). Fish densities were estimated using SCUBA visual strip transect surveys (Weispfenning 2006). Each survey consisted of eight, 25 x 4m transects between two depth strata, for a total surveyed area of 800m<sup>2</sup>. Surveys were run parallel to shore in a stair-step fashion, beginning with four deep transects between 19.8m and 13.7m, and ending with four shallow transects between 13.7m and 9.1m. Total lengths of encountered fish were estimated to the nearest 5cm using a set of parallel lasers spaced 20cm apart (details in Weispfenning 2006).

Surveys at each site occurred between June and August. In 2004, surveys were conducted once per month; in 2005, they were carried out three times per month. The same lead diver conducted bottomfish counts in all months during both years (except August

2005). All dive surveys were conducted during slack tide to maintain consistency between sampling dates, avoid the effects of tidal currents on diving safety and fish behavior, and maximize the period of best subtidal visibility (Weispfenning 2006). Surveys provided current bottomfish densities to be used in analysis with harbor seal surveys performed during the same months.

### **Seal distribution data collection – scan sampling**

To assess whether or not harbor seal presence was correlated with bottomfish density (i.e., whether there was an aggregative response), land-based observations were used to collect data on the frequency and location of harbor seal sightings within a candidate marine reserve encompassing differing bottomfish densities. Observations were conducted between June and August during 2004 and 2005 to cover the pupping season, when seals are both abundant and foraging locally (Boness *et al.* 1994, Coltman *et al.* 1997).

All observation periods took place between 06:00 and 21:00, were four hours in length, and centered around slack current to coincide with periods suitable for collecting bottomfish density data. All slack currents were observed within a given observation day unless 1) the slack current fell at a time of day when a four-hour observation extended beyond the hours of 06:00 or 21:00, 2) slack currents occurred too close together for the observers to have an adequate rest period, or 3) a slack current period was used for a SCUBA survey. When observation periods would have overlapped because slack currents were less than four hours apart, one was randomly selected. Efforts were suspended when visibility

was poor due to fog, steady rain, or sea state (Beaufort scale of 4 or more in any of the sectors).

Scan sampling was used to record the number and locations of harbor seals swimming in the water (Altmann 1974). The visible area was divided into three unequally sized sectors within the larger 0.8km<sup>2</sup> study area (Figure 2). Once every hour, each sector was scanned visually and with binoculars (7 x 50 power) to make instantaneous counts of all harbor seals seen swimming at the water's surface. Counts within a sector took 3 to 5 min to complete and seals rarely moved between sectors in that time. If a seal was believed to surface more than once during a scan, only the first sighting time and location were used. Within each sector, total counts of seals observed, sea state, cloud cover, visibility, and time of day of sightings were recorded.

To determine the exact position of seals within the candidate reserve, I employed a Leica TC605L theodolite to collect horizontal and vertical bearings to each seal. Using the observer's height above mean zero tide level and line-of-sight distance from the theodolite to the sighted seal, the distance along the earth's surface between the theodolite and the seal was calculated (Lerczak and Hobbs 1998). The theodolite height was measured directly and then corrected for the specific tide height at each given sighting event. Tide heights were estimated from a cosine prediction of tide height using Tides & Currents Pro, ver. 2.5b (Nautical Software, Inc.).

The great circumference equation (Lerczak and Hobbs 1998, Zwillinger 2003) was then used to determine the geographic position (latitude and longitude) of each seal sighting:

$$\tau = \eta - \rho$$

$$Lat_F = \sin^{-1}(\cos(\tau) * \sin(D/60/1852) * \cos(Lat_S) + [\sin(Lat_S) * \cos(D/60/1852)])$$

$$Lon_F = \cos^{-1} \left( \frac{\cos(D/60/1852) - [\sin(Lat_S) * \sin(Lat_F)]}{\cos(Lat_S) * \cos(Lat_F)} \right) + Lon_S$$

where,

$D$  = distance (m) between the two points along the surface of the Earth

$\tau$  = bearing or vertical angle from station to seal

$\eta$  = azimuth or horizontal angle estimated with the theodolite

$\rho$  = reference azimuth (bearing from station to reference point)

$Lat_S$  = Latitude of the station

$Lon_S$  = Longitude of the station

$Lat_F$  = Latitude of the fixed object (seal)

$Lon_F$  = Longitude of the fixed object (seal)

### **Tracking data collection**

To identify harbor seal prey and foraging locations I conducted opportunistic focal observations from land in addition to scan sampling. Using a random number generator, I chose one focal seal during an observation period and followed its movements while a second observer continued scan sampling. Time, location (as determined by theodolite), and



behavior (including predation events and prey species when identifiable) were recorded for each observed surfacing event. If a tracking session ended (i.e., contact with the focal seal was lost for 20 minutes) and sufficient observation time remained, further tracking sessions were attempted with new focal animals. If the observation period ended prior to losing contact with the seal, the tracking session was ended prematurely.

### **Data analysis**

For each dive site, bottomfish densities from all SCUBA surveys were enumerated per hectare. Mixed-effects models were used to determine whether bottomfish density differed among sites, while accounting for the repeated sampling of dive sites over time. This method was preferable to traditional repeated measures analysis due to the unbalanced design resulting from different survey frequencies between years. For the full model, fish densities were the response variable and year a fixed factor (block) while site by month was included as a random factor. Random factors were iteratively removed from the model to determine their contribution to model fit. Parameters were estimated using restricted maximum likelihood as all the models had the same fixed effects. AIC scores were used to determine the most parsimonious model (Burnham and Anderson 2001) and an ANOVA test used to reveal significant effects. A reciprocal transformation was used to correct for non-normality and heterogeneous variances (Zar 1999). Analyses were performed using R, ver. 2.4.1 (R Development Core Team 2006).

To compare the frequency of seal sightings relative to bottomfish densities at sites A and B, I imported the geographic positions of seal sightings, dive routes of the SCUBA surveys (as determined by GPS), and a coastal map of the survey area (DNR 2001) into a

GIS database ArcGIS 9.1 (ESRI 2005). Mean maximum error in theodolite locations was approximately 15.7m (n=58, SD=16.4m), based on calibration with the Washington State ShoreZone Inventory dataset (DNR 2001). Dive corridors were then buffered by 27.7m (2m of visibility on either side of the dive transect plus 5 seal body lengths plus the average sighting error), and only sightings that fell within the buffers were used in the analysis. I compared the number of seal sightings in proximity to either dive site with a Chi-square Goodness of Fit test (Zar 1999) using R, ver. 2.4.1 (R Development Core Team 2006).

Given that focal observations and photographic identification of individual seals indicated that animals could spend over an hour in the study area and that it was not possible to identify individual seals during scans, it was feasible that sightings within an observation period might not be independent. Hence, if more than one seal sighting fell within either buffer for a given 4-hr observation period, one sighting was randomly selected for this analysis to ensure independence.

Focal observations provided a better opportunity to identify harbor seal prey than scan sample surveys. Because I was following the movements of specific animals and attempting to record every surfacing, I was more likely to witness a predation event and had more time to attempt prey identification. Recorded predation events were extracted from all tracking records (whether of individual or multiple seals; n=45; 78.5hrs) during 2004 and 2005 to plot observed predation relative to the two dive sites as well as to quantify the commonly observed prey species. Predation events were thus compared to known bottomfish densities and used as indicators of the seals' functional response.

## **Seal habitat use of a candidate marine reserve**

### **Scan sample data**

Scan sampling was conducted as described in ‘Seal distribution data collection’. To examine habitat use of the entire channel, I included all sightings rather than those that fell in proximity to either dive site.

### **Tracking data**

To further assess seal habitat use in Burrows Channel and to describe behavior patterns of individual seals using the candidate reserve area, I used a subset of focal observations (n=20; 27.5hrs) where I judged the track to be of a single seal given distinguishing marks or the observer’s judgment at the time of the track. Additionally, focal seals had to be observed for at least 20mins to be included in these analyses. Focal observation protocol is detailed in the ‘Tracking data collection’ section.

### **Photographic identification data collection**

In 2004 and 2005, photographs were taken opportunistically of harbor seals that surfaced in proximity to the observation station using a hand-held digital SLR camera (Canon 10-D) with a 100-400mm image-stabilizing lens. "Photographic sets" consisted of photographs taken within several minutes of each other and were typically of a single surfacing event. I categorized the digital images based on view of the seal (e.g., right side, left side, ventral, etc.), quality of the image (e.g., excellent, good, fair, poor), and whether or

not distinguishing marks could be seen. I then compared photographs of excellent to fair quality to determine whether individual seals were seen repeatedly between years, from day-to-day within a field season, or within a day. Photographic identification of individual seals has been used for various pinniped species such as Mediterranean monk seals (*Monachus monachus*; Forcada and Aguilar 2000), Hawaiian monk seals (*Monachus schauinslandi*; Harting *et al.* 2004), leopard seals (*Hydrurga leptonyx*; Forcada and Robinson 2006), and grey seals (*Haliocoerurus grypus*; Karlsson *et al.* 2005). Further, individual harbor seals have been identified for over ten years based on distinct markings or scars (Johnson and Jeffries 1983) and pelage patterns have been used to distinguish between North Pacific harbor seal populations (Yochem *et al.* 1990). However, I am not aware of any studies that have used photographic methods to identify individual harbor seals in the water.

### **Data analysis**

**Scan sample data.** Swim speeds and dive times could easily allow seals to move anywhere within the survey area, or to leave the area entirely, in the space of 1hr (Pitcher and McAllister 1981, Bjørge *et al.* 1995, Suryan and Harvey 1998). Consequently, it was assumed that any individual seal could choose to move between any two sectors during the time between counts. It was subsequently assumed that seals moved independently of one another for purposes of the spatial analysis of seal sightings in the study area. A 50m<sup>2</sup> grid was overlaid on the seal location data plotted in ArcGIS, and the total number of seal sightings within each grid square was enumerated and then divided by 5.28, a fraction of the total hourly counts (n=528), to determine the frequency of seal sightings in the channel (i.e., sightings/100scans). The distribution of seal sightings was also plotted relative to

bathymetry (NOAA 2002), tidal activity, and areas of known foraging (from predation events recorded during tracking).

**Tracking and photographic identification.** Tracking sessions of individual seals were assigned general activity categories described by Suryan and Harvey (1998). ‘Milling’, which is typically associated with foraging, occurred when seals did not move in a specific direction, but rather remained in a localized area and dove repeatedly. Pursuit swimming and predation events were also used to characterize milling. Seals moving consistently in a single direction characterized ‘traveling’, while seals remaining in one location and spending approximately equivalent times diving and at the surface were classified as ‘resting’. A single track could be assigned multiple behaviors. These data were used to summarize the behavior allocation of individual seals while in the study area as well as the average time spent within the candidate marine reserve. To examine the spatial extent of individual seals’ visits to Burrows Channel, I mapped tracking paths using Hawth’s Tools Animal Movements extension, ver. 3.26 (Beyer 2004).

Photographic matches of distinguishing marks were used to quantify, 1) the minimum amount of time that seals spent in proximity to the north shore of Burrows Channel within a 4-hr observation period, and 2) whether individuals were seen repeatedly between years or during the field seasons (2004 and 2005).

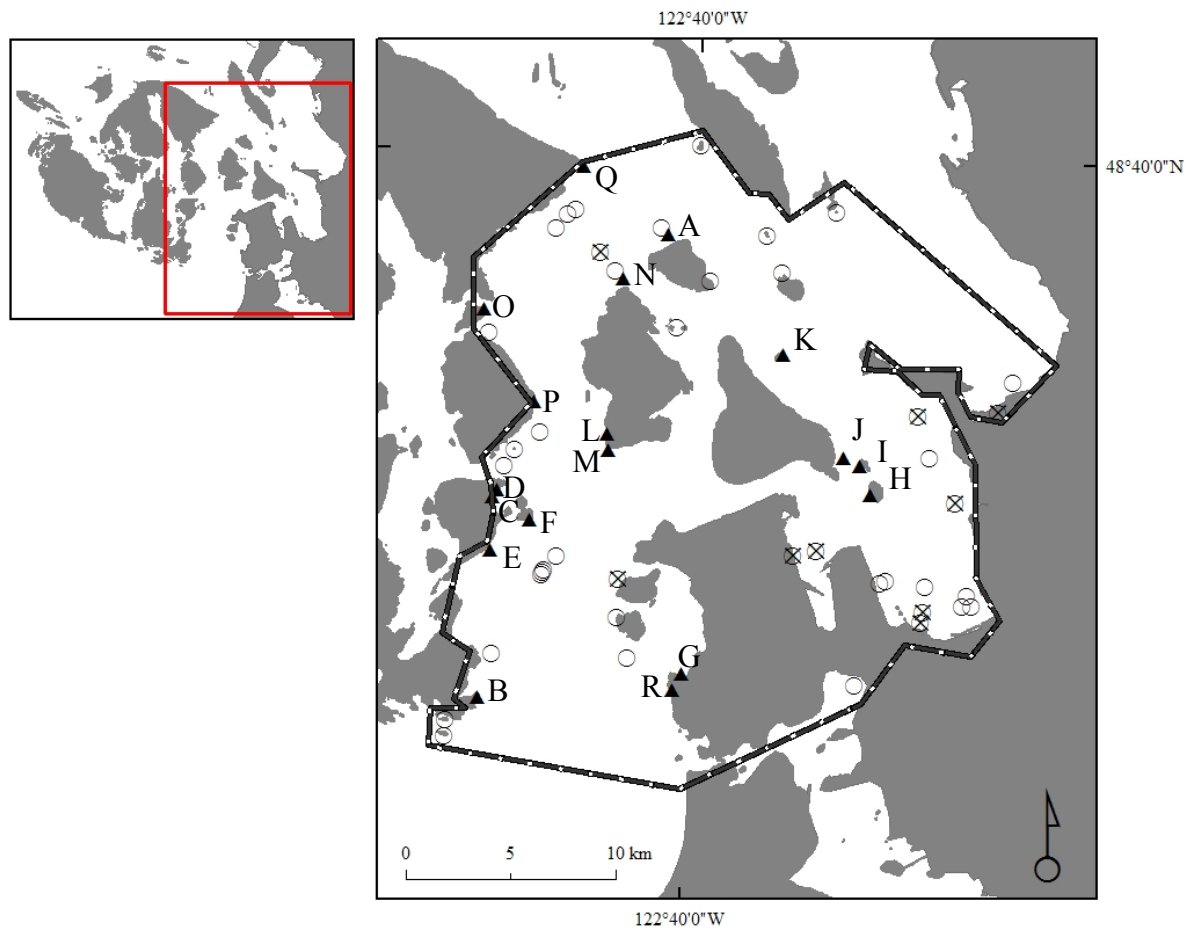
## RESULTS

### Number of harbor seals in and around candidate marine reserves

#### Regional scale

Aerial surveys in 2005 were flown twice per month on consecutive days, except in August when three days were used to get complete coverage of the eastern San Juan Islands due to fog and in September when inclement weather prevented consecutive surveys. Thirty-three of the 51 haulouts (65%) used by seals were known haulout sites (Figure 3). However, 18 of the 51 sites (35%) were not previously documented (Figure 3; Table 1), but only four of these haulouts (8% of sites used) may represent novel haulout locations (when compared with published accounts; Jeffries *et al.* 2000). Additionally, individual haulout sites were not used consistently from April through September in 2005, with a few haulouts only used for part of the field season, either between April and June or after June (Figure 3; Table 1).

The average number of hauled out seals (not including pups) approximately doubled from  $1,249 \pm \text{SD } 193$  seals in April to  $2,302 \pm \text{SD } 120$  seals in September; a period that encompasses the pre-reproductive and reproductive seasons for harbor seals in this area (Figure 4). The difference between counts of seals ashore during pre-reproductive months (April and May) and the reproductive period (July and August) was significantly different than zero in the surveyed region ( $t_3 = 5.144$ ,  $p = 0.01$ ). Additionally, the number of haulout sites used increased from 28 to 43 over this same period (Figure 4). Although the overall regional trend was of increasing average numbers of seals hauled out, individual haulouts showed unique patterns in mean harbor seal counts through the six months of aerial surveys (Figure 5).

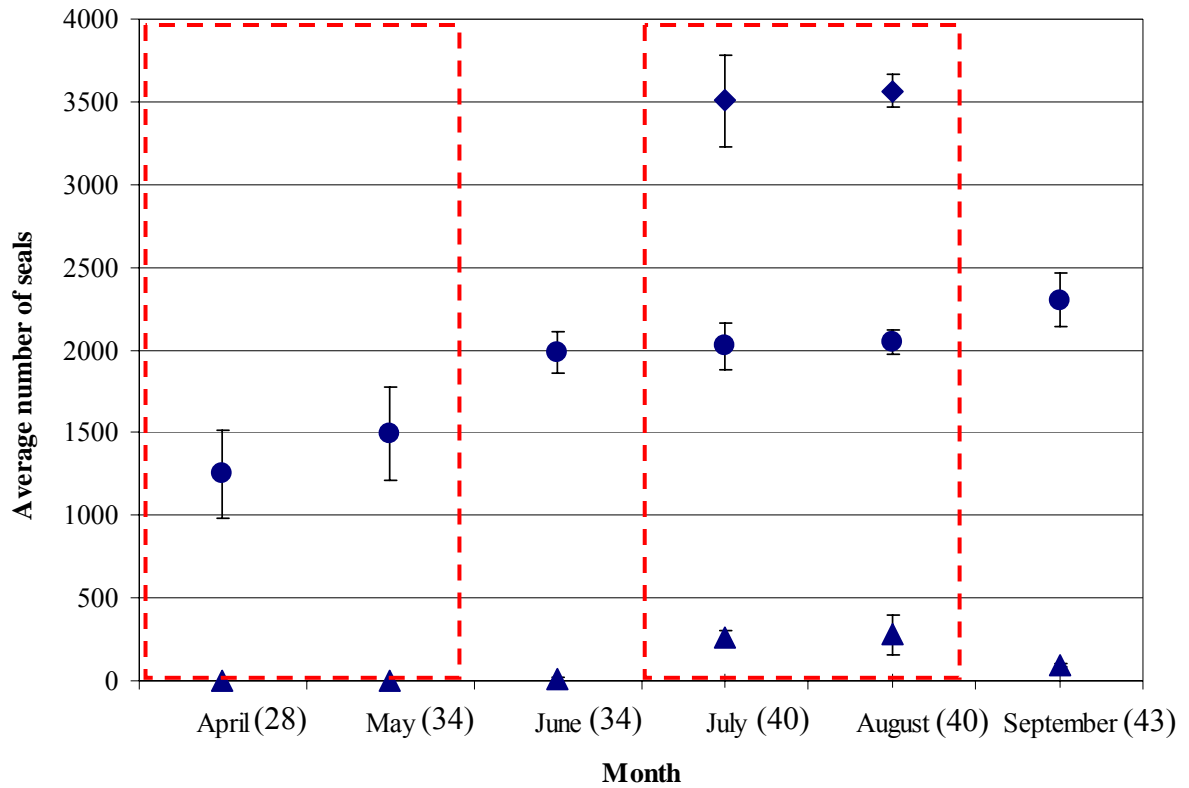


**Figure 3.** Map depicting the aerial survey area and haulouts historically recorded in the Atlas of Seal and Sea Lion Haulout Sites in Washington (Jeffries *et al.* 2000) as well as haulouts found in 2005. Open circles are haulouts recorded in surveys before 2000 (Jeffries *et al.* 2000) where seals were also seen in 2005. Crossed circles represent haulouts listed in the atlas, but where no seals were seen in 2005. Lettered triangles are novel locations where seals were found in 2005. See Table 1 for site descriptions.

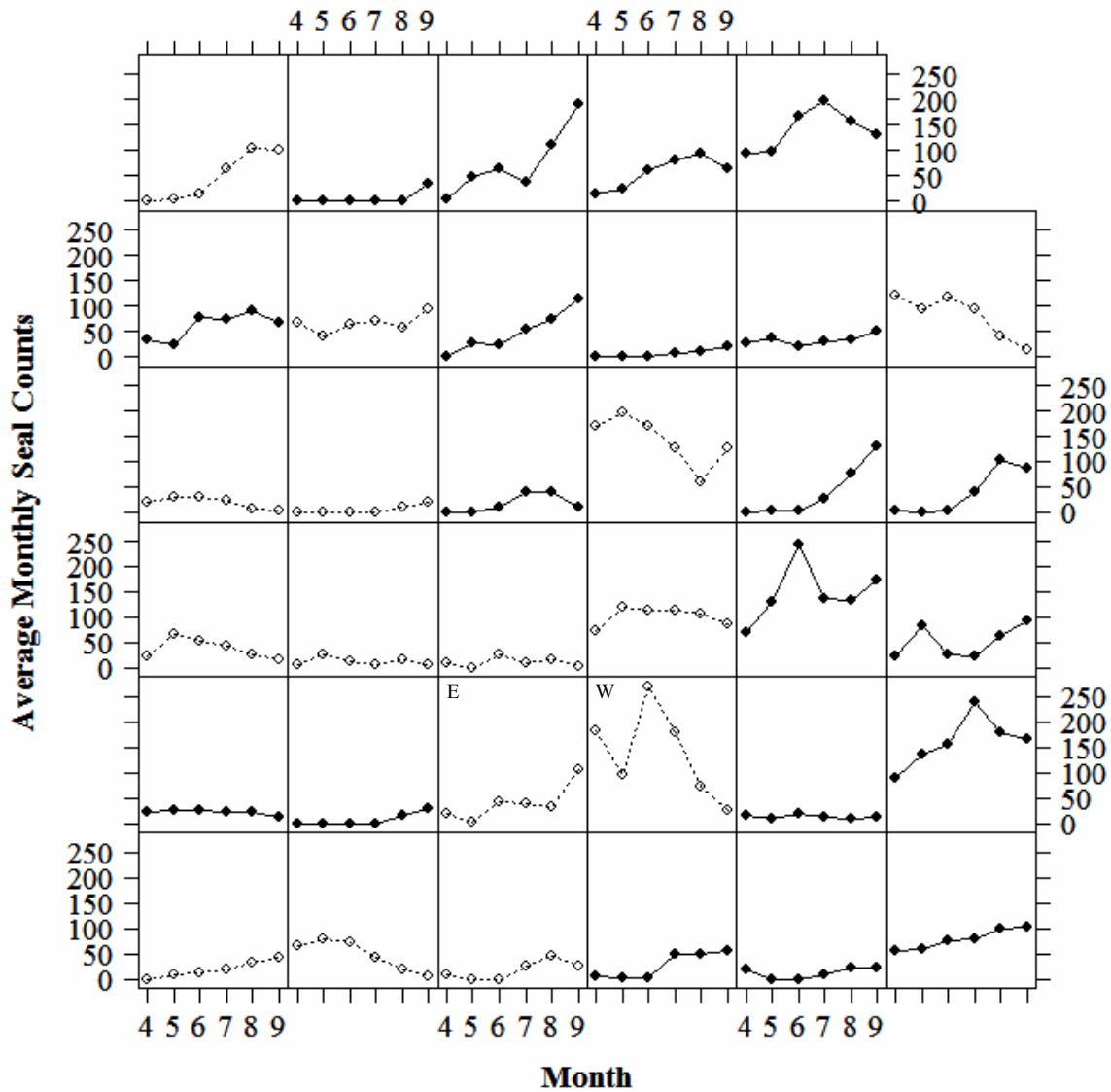
**Table 1.** List of novel haulout sites found in 2005. Labels refer to the labels in Figure 3. Status of haulouts are as follows: *L*, haulouts lumped in the Atlas of Seal and Seal Lion Haulout Sites in Washington and Oregon (Jeffries *et al.* 2000), but split out separately for the purposes of this study; *S*, haulouts used by seals outside of the months of July and August; *N*, haulouts not listed in Jeffries *et al.* (2000) but used during July and August 2005; and *I*, areas intermittently used by less than ten seals (see Discussion section). Labels in bold italics are sites strictly used either between April and June or after June 2005.

<b>Label</b>	<b>Status</b>	<b>Max. Count</b>	<b>Latitude</b>	<b>Longitude</b>	<b>Description</b>
A	L	< 200	48.63211	-122.68613	On intertidal rocks and reef area at Boulder Reef (East) <i>Note: lumped with Boulder Reef (West) by WDFW</i>
B	S, I	< 5	48.43148	-122.80190	On intertidal rocks and reef areas at Boulder Island.
<b>C</b>	S, I	< 10	48.51765	-122.79536	On intertidal rocks and reef areas in Decatur Bay.
<b>D</b>	I	< 10	48.52063	-122.79271	On intertidal rocks and reef areas in Decatur Bay - Fauntleroy Point.
E	S, I	< 10	48.49468	-122.79565	On intertidal rocks and reef areas near Dot Rock.
<b>F</b>	S	< 20	48.50816	-122.77109	On intertidal rocks and reef areas near James Island. <i>Note: not used by more than one seal until September</i>
<b>G</b>	N	< 30	48.44355	-122.67047	On intertidal rocks and reef areas in Burrows Bay/Langley Bay. <i>Note: used in August &amp; September</i>
H	N	< 100	48.52239	-122.55124	On intertidal rocks and reef areas on west side of Hat Island.
I	N	< 30	48.53773	-122.56911	On intertidal rocks and reef areas on north side of Huckleberry Island.
<b>J</b>	S	< 20	48.53490	-122.55867	On intertidal rocks and reef areas on southwest side of Saddlebag Island. <i>Note: not used by more than one seal except in May</i>
<b>K</b>	S	< 20	48.58158	-122.61030	On intertidal rocks and reef areas on Jack Island.
L	I	< 5	48.53651	-122.72070	On intertidal rocks and reef areas on southwest side of Cypress Island.
M	N	< 40	48.54560	-122.72188	On intertidal rocks and reef areas at southwest tip of Cypress Island.
<b>N</b>	S	< 50	48.61251	-122.71489	On intertidal rocks and reef areas on northwest side of Towhead Island.
O	L	< 30	48.59775	-122.80382	On intertidal rocks and reef areas on east side of Obstruction Island.
<b>P</b>	L	< 30	48.55880	-122.77028	On intertidal ledges and reef area along east side of Blakely Island. <i>Note: lumped with Spindle Rock and Obstruction Island reefs by WDFW</i>
Q	I	< 5	48.66127	-122.74095	On intertidal ledges and reef area of Lawrence Point - Orcas Island.
R	I	< 5	48.43928	-122.67404	On intertidal ledges and reef area of Sares Head - Fidalgo Island.





**Figure 4.** Average monthly counts of seals ashore in the study area. Seals >1 year-old are represented as circles, pups as triangles. The dashed boxes highlight the pre-reproductive (April-May) and the reproductive (July-August) seasons. Numbers in parentheses are the number of locations (i.e., haulouts) at which one or more harbor seals were seen on either of the survey days for a given month. Corrected abundance (♦) of hauled-out harbor seals (all age-sex classes) was only estimated for the pupping season. Error bars represent 95% confidence intervals.



**Figure 5.** Average monthly counts for a subset of haulout sites. Each square represents an individual site. “Unlimited” haulouts are solid lines with filled symbols, “limited” haulouts are dotted lines with open symbols. East and West Boulder Reefs are indicated with an E and W, respectively (see Discussion section).

Though a single pup was seen in both April and May, it was not until June that several pups were counted, with the peak number of pups recorded in August (280) and numbers decreasing by September (Figure 4). Using the combined correction factor, developed by Huber *et al.* (2001) for the pupping season in Washington and Oregon, abundance estimates of  $3,504 \pm \text{SD } 200$  and  $3,564 \pm \text{SD } 73$  seals were generated for the months of July and August, respectively (Figure 4).

### **Influence of tidal availability on haulout use**

The non-linear model did not significantly improve the fit of the model to the average monthly counts (AIC=967.4585;  $p=0.170$ ; Table 2a), providing evidence that the relationship was linear. Including tidal classification as a factor, however, improved the fit of the model as indicated by reduced AIC scores (AIC=967.3377;  $p=0.0162$ ; Table 2b). Month and the month-by-tidal classification interaction were significant descriptors for model fit (ANOVA;  $p<0.001$  and  $p= 0.0045$ , respectively; Table 2c). Slopes predicted by the final linear mixed-model were both positive and negative for “limited” haulouts, while all slopes were positive for “unlimited” haulouts (Figure 6).

Monthly average counts for each haulout type within the study area were approximately equivalent from April through June with counts increasing on “unlimited” sites (Figure 7). After June the average number of seals hauled out on “unlimited” haulouts surpassed the number of seals ashore on “limited” haulouts and continued increasing (Figure 7).

**Table 2a.** Comparison of the linear model (Model 1) with the non-linear model (Model 2). The non-linear model does not improve model fit.

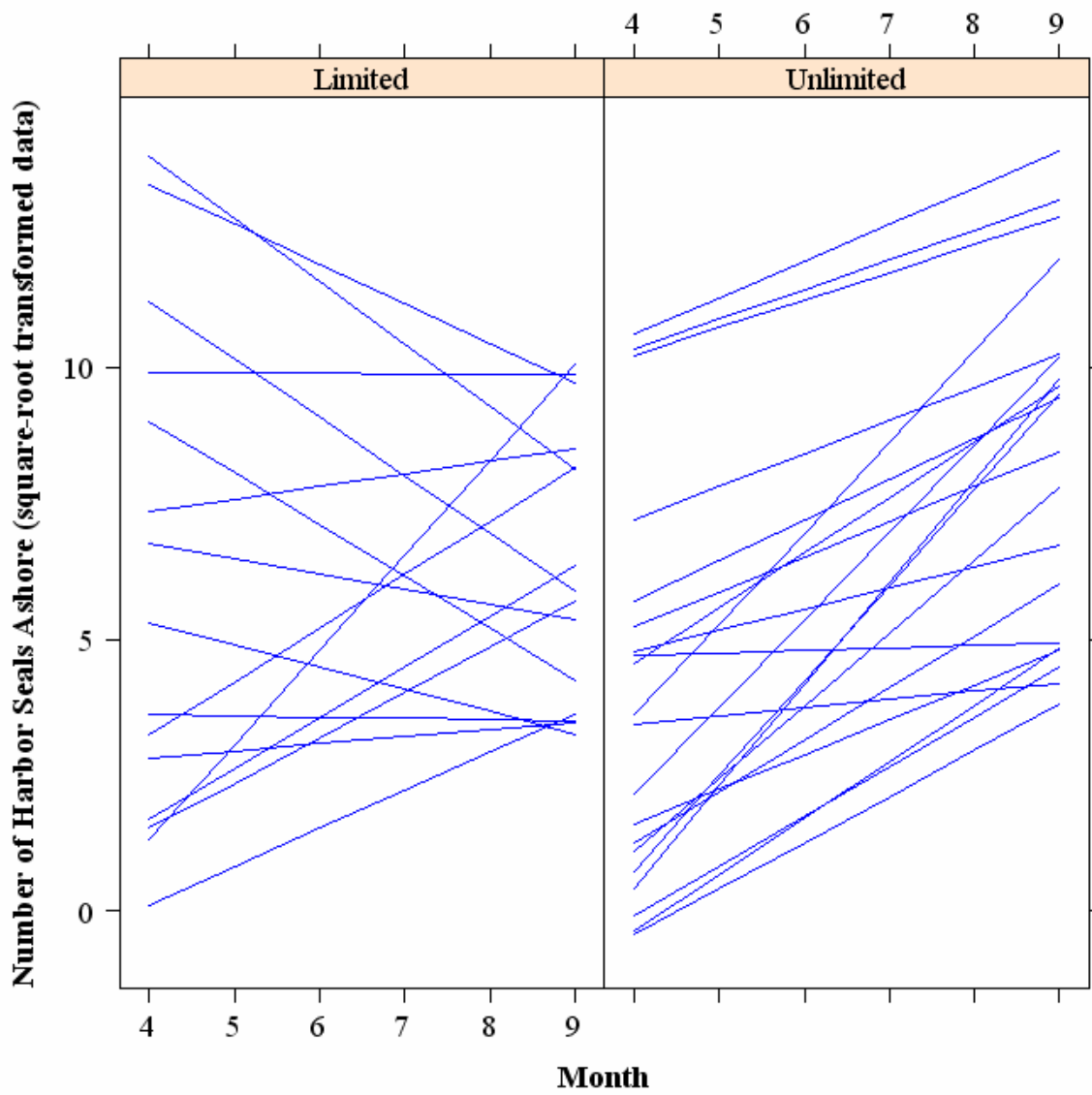
<b>Model</b>	<b>df</b>	<b>AIC</b>	<b>BIC</b>	<b>logLik</b>	<b>Test</b>	<b>L.ratio</b>	<b>p-value</b>
<b>1</b>	8	967.3377	994.1146	-475.6689			
<b>2</b>	9	967.4585	997.5825	-474.7293	1 vs 2	1.879178	0.1704

**2b.** Comparison of the overall model (Model 1) with the model including tidal classification (Model 2). Including tidal classification significantly improves the fit of the model.

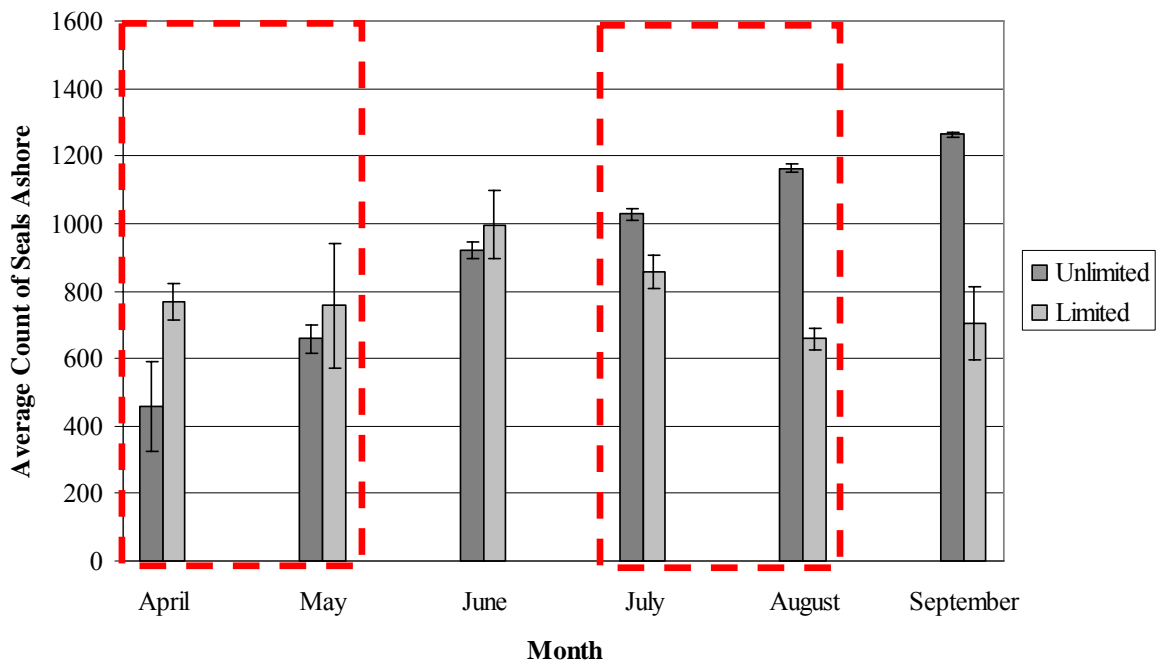
<b>Model</b>	<b>df</b>	<b>AIC</b>	<b>BIC</b>	<b>logLik</b>	<b>Test</b>	<b>L.ratio</b>	<b>p-value</b>
<b>1</b>	6	971.5791	991.6618	-479.7896			
<b>2</b>	8	967.3377	994.1146	-475.6689	1 vs 2	8.241403	0.0162

**2c.** ANOVA table for the most parsimonious mixed-effects model.

	<b>df</b>	<b>F-value</b>	<b>p-value</b>
<b>(Intercept)</b>	173	158.95824	<0.0001
<b>Tidal Classification</b>	33	0.57038	0.4960
<b>Month</b>	173	15.05684	0.0002
<b>Tidal:Month</b>	173	8.01735	0.0045



**Figure 6.** Predicted fitted slopes for tidally “limited” and “unlimited” haulouts for surveys conducted April-September, 2005.



**Figure 7.** Average number of non-pup seals hauled out on sites of differing tidal availability within the study area (n=15 for each haulout type). Dashed boxes highlight the pre-pupping season and the pupping season, which shared sampled tides of similar extremity (see Discussion section). Error bars represent  $\pm 1SD$ .

## **Bottomfish and harbor seal distribution within a candidate marine reserve**

### **Bottomfish distribution**

Bottomfish species identified during SCUBA surveys included copper rockfish, yellowtail rockfish (*S. flavidus*), black rockfish (*S. melanops*), quillback rockfish, yelloweye rockfish (*S. ruberrimus*), Puget Sound rockfish (*S. emphaeus*), unidentified rockfish (Scorpaenidae), lingcod, kelp greenling, painted greenling (*Oxylebius pictus*), and buffalo sculpin (*Enophrys bison*; Table 3). Only copper, yellowtail, black, quillback, yelloweye rockfish, lingcod, and kelp greenling were used in analyses, as they are the main species being targeted for protection within candidate reserves in this area.

Site was the only factor explaining the pattern of bottomfish density; removing it from the model significantly increased the model's AIC score indicating that it was explaining a significant portion of the variance in the model and should be retained (Table 4). Fish density averaged  $461.5 \pm \text{SD } 115.4$  fish/ha at Site A and  $239.6 \pm \text{SD } 44.2$  fish/ha at Site B (Figure 8). Removing month from the model did not significantly change the fit of the model demonstrating that month was not an important factor (Table 4). Additionally, year was not significant in the model of best fit (Model 2; ANOVA;  $p = 0.5898$ ).

### **Bottomfish and seal distribution – scan sampling**

During land-based observation in 2004 and 2005, a total of 231 and 297 hourly counts were performed on 36% and 50% of the days between 1 June and 31 August in 2004 and 2005, respectively. Two observation periods were completed during 45% of the days

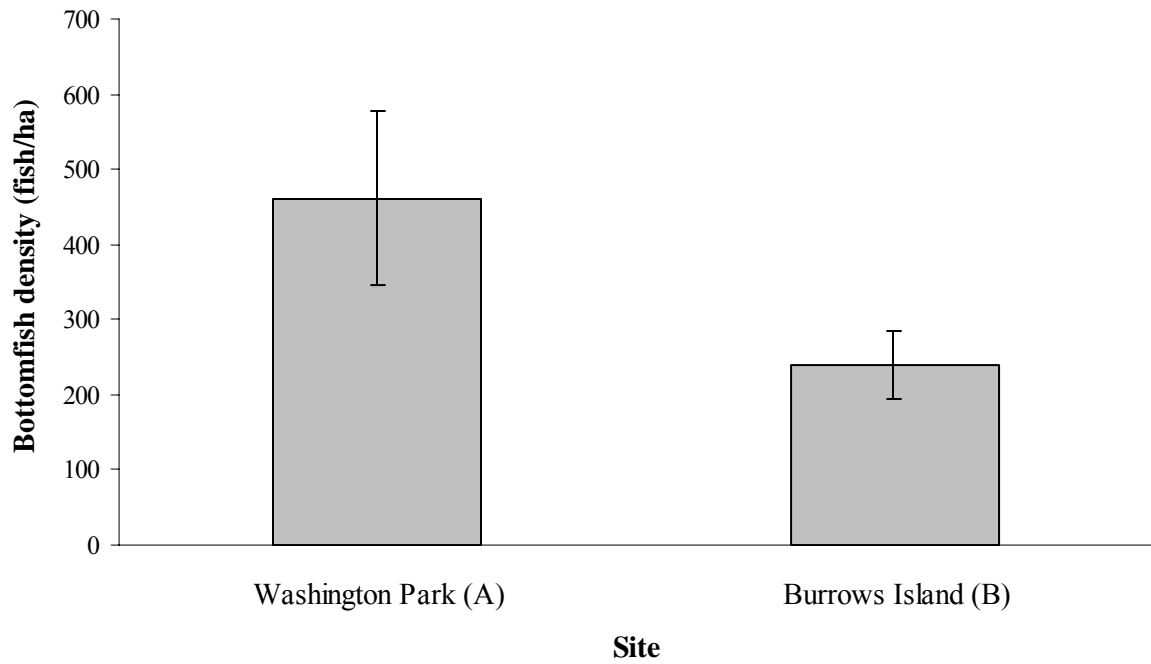
**Table 3.** Bottomfish species identified at Site A and Site B. Asterisked species were not included in density estimates or used in analysis (see Results section). Mean total lengths of fish recorded in Burrows Channel (see Discussion section).

<b>Species</b>	<b>Site A</b>	<b>Site B</b>	<b>Total Length (cm) <math>\pm</math> SD</b>
Black Rockfish	X	X	28.75 $\pm$ 6.37
Copper Rockfish	X	X	23.6 $\pm$ 4.97
Puget Sound Rockfish*	X	X	NA
Quillback Rockfish		X	15
Yelloweye Rockfish		X	10
Yellowtail Rockfish	X		25 $\pm$ 1.77
Unidentified Rockfish*		X	NA
Lingcod	X	X	55.55 $\pm$ 10.08
Kelp Greenling	X	X	28.73 $\pm$ 6.14
Buffalo sculpin*	X		NA
Painted Greenling*	X		NA



**Table 4.** Comparison of the overall model (Model 1) with the model excluding month (Model 2), and with the model excluding site (Model 3). The model including site (Model 2) is the most parsimonious model. Note: AIC and BIC values are negative due to the reciprocal transformation of the data.

<b>Model</b>	<b>df</b>	<b>AIC</b>	<b>BIC</b>	<b>logLik</b>	<b>Test</b>	<b>L.ratio</b>	<b>p-value</b>
<b>1</b>	6	-237.6904	-231.1441	124.8452			
<b>2</b>	4	-241.6904	-237.3262	124.8452	1 vs 2	0.0000	1
<b>3</b>	3	-221.2383	-217.9652	113.6192	2 vs 3	22.4521	<.0001



**Figure 8.** Mean bottomfish densities (fish/ha) at Site A ( $461.5 \pm \text{SD } 115.4$  fish/ha) and Site B ( $239.6 \pm \text{SD } 44.2$  fish/ha). Error bars represent  $\pm 1\text{SD}$ . Data were transformed for analysis.

surveyed in 2004 and 30% in 2005. After eliminating multiple sightings within a 4-hr observation period, a total of 34 seals were recorded in proximity (within 27.7m) of either dive site (Figure 9). Throughout the study, 74% of these seals were seen near Site A and 26% near Site B ( $\chi^2_1 = 7.53$ ;  $p = 0.006$ ).

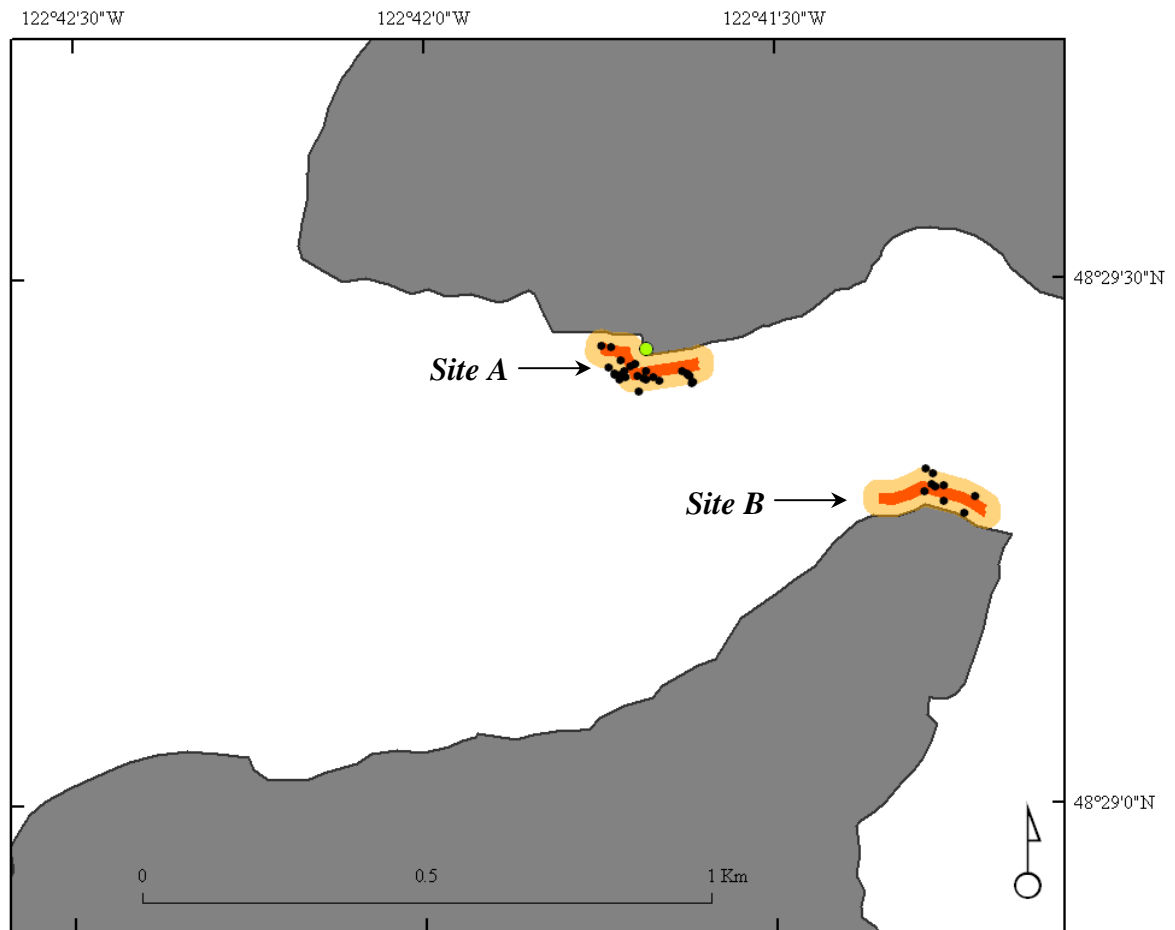
### **Bottomfish and seal distribution – tracking observations - predation**

Recorded predation events ( $n=34$ ) occurred mainly near the north shore of Burrows Channel, with a few documented mid-channel as well (Figure 10). Of the predation events, 32% were recorded within the buffer of Site A and none near dive Site B. Predation events were recorded up to 850m from the observation station (Figure 10), farther away than Site B (500-700m from observers). Overall, I identified the primary prey as small pelagic fish (50%). Of the remaining prey, 26% were unidentified, due to distance or handling by the seal, 21% were considered non-bottomfish, either due to prey size or body shape, and 3% were octopus. Of the prey consumed specifically in proximity to Site A, 73% were small pelagics, 18% were unidentified, and 9% were non-bottomfish.

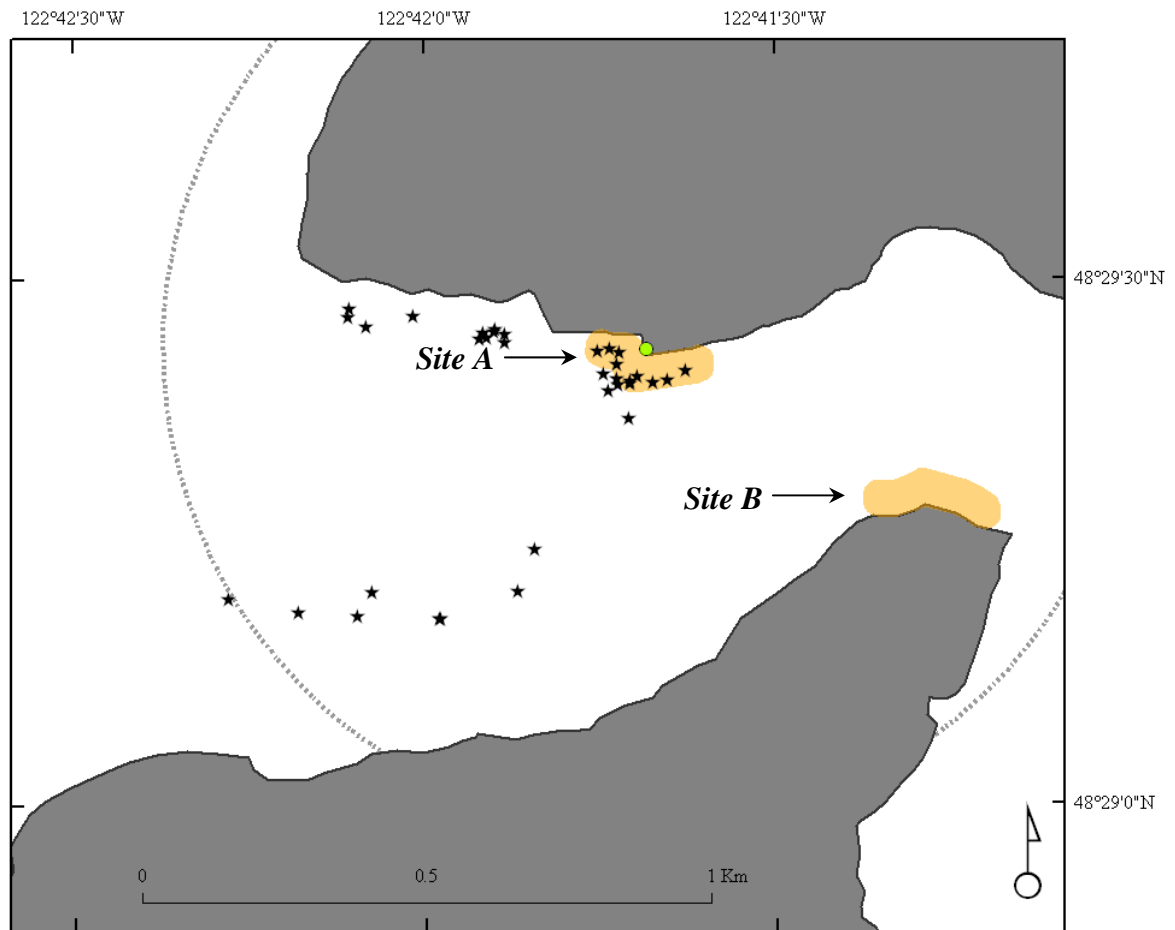
### **Seal habitat use of a candidate marine reserve**

#### **Scan sample data**

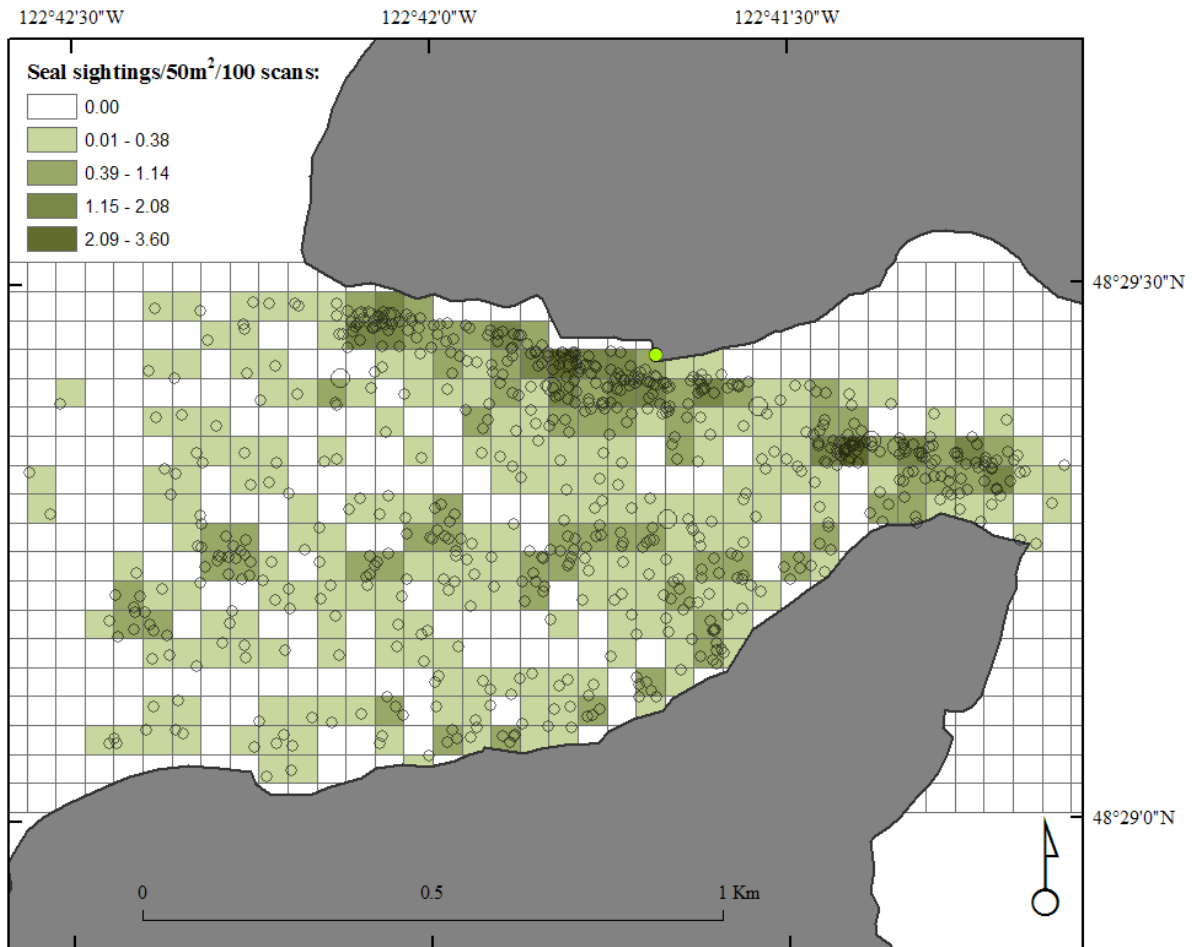
Throughout Burrows Channel, 149 seal sightings were recorded in 2004 and 519 sightings were made in 2005 (Figure 11). In both years, approximately 5% of the sightings occurred in Rosario Strait, outside of the candidate marine reserve. The maximum number of seals recorded in the study area during a given hourly count was five in 2004 and eight in



**Figure 9.** Seals sighted near either dive site in 2004 and 2005 (filled circles). Dive corridors are delineated by bold lines and buffered by 27.7m (shaded area). The observation station is represented by a circle.



**Figure 10.** Observed predation events (★) in the candidate marine reserve relative to buffered dive sites A and B. The observation station is represented by a circle and an 850m buffer is delineated by a dotted line.



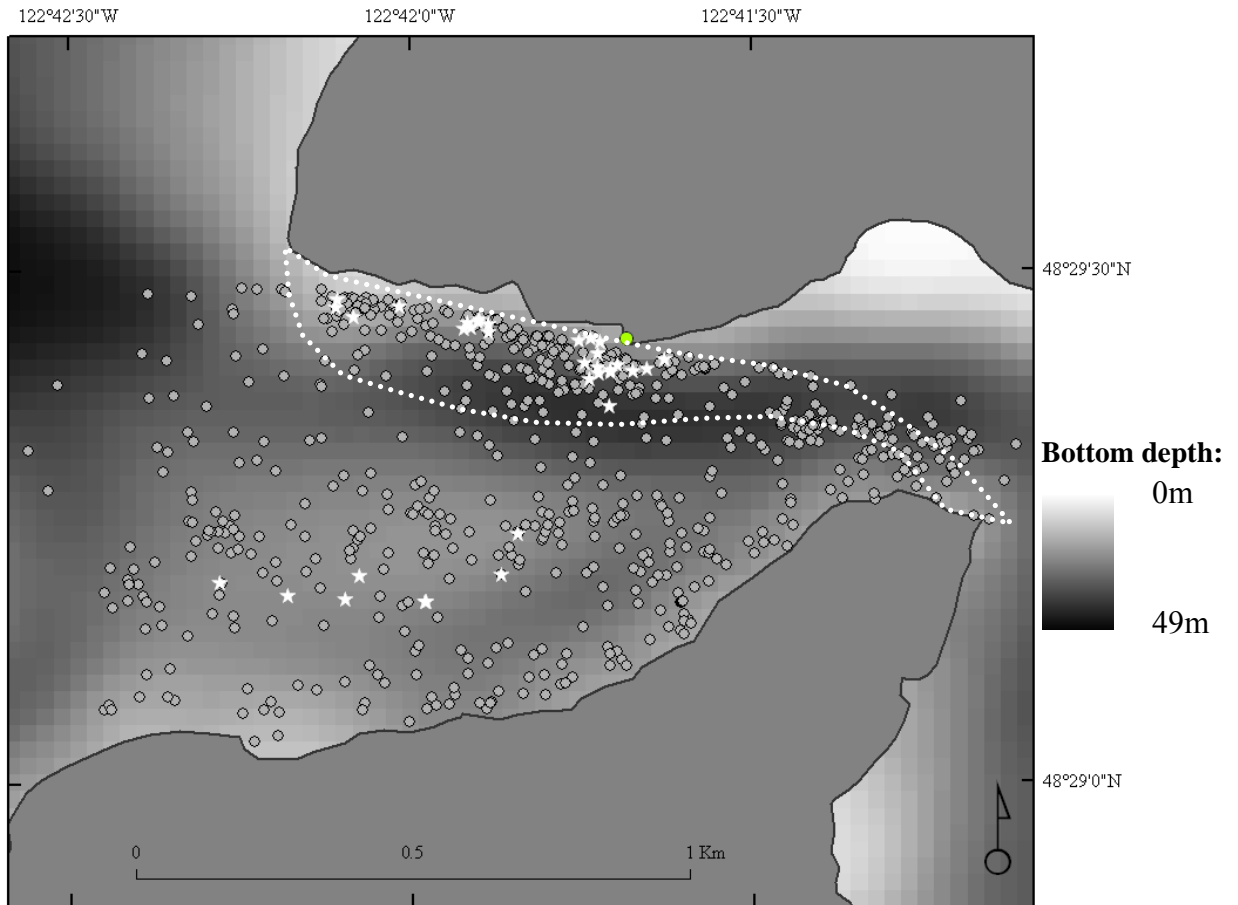
**Figure 11.** Harbor seal sighting distribution and frequency in the study area in 2004 and 2005 (using a 50m<sup>2</sup> grid). Small circles are records of one seal, larger symbols are sighting locations of two seals that surfaced simultaneously. The observation station is represented by a circle.

2005. Average encounter rates were approximately double in 2005 (2 seals per count) when compared to 2004 (1 seal per count).

Seal sightings were not evenly distributed over the study area (Figure 11). The highest frequencies of seal sightings were recorded in the eastern end of the channel and along the north shore (1.15-3.60 sightings/50m<sup>2</sup>/100 scans; Figure 11). Fewer sightings occurred near the south shore and mid-channel (0.39-1.14 sightings/50m<sup>2</sup>/100 scans), but these sighting frequencies were still greater on average than the frequency of sightings between the north shore and mid-channel, and between mid-channel and the south shore (0-0.38 sightings/50m<sup>2</sup>/100 scans; Figure 11). Seal sighting frequencies and predation events (recorded during tracking) appeared positively associated with the location of a tidal rip that is commonly present in Burrows Channel (Figure 12). In addition, the majority of sightings (56%) occurred in waters less than 25m in depth (Figure 12).

### **Tracking observations**

Twenty-one tracks of individual seals were recorded, totaling 17.5hrs and 9.82hrs in 2004 and 2005, respectively. Sessions averaged  $1.75 \pm \text{SD } 1.28\text{hrs}$  in 2004 and  $0.98 \pm \text{SD } 0.38\text{hrs}$  in 2005; the longest tracking session lasted 3.56hrs. Because tracking sessions were often ended because the observer lost contact with the focal animal (for 20mins or more), or because the observation period ended, these values are an approximation for the minimum amount of time that seals spent in the study area.



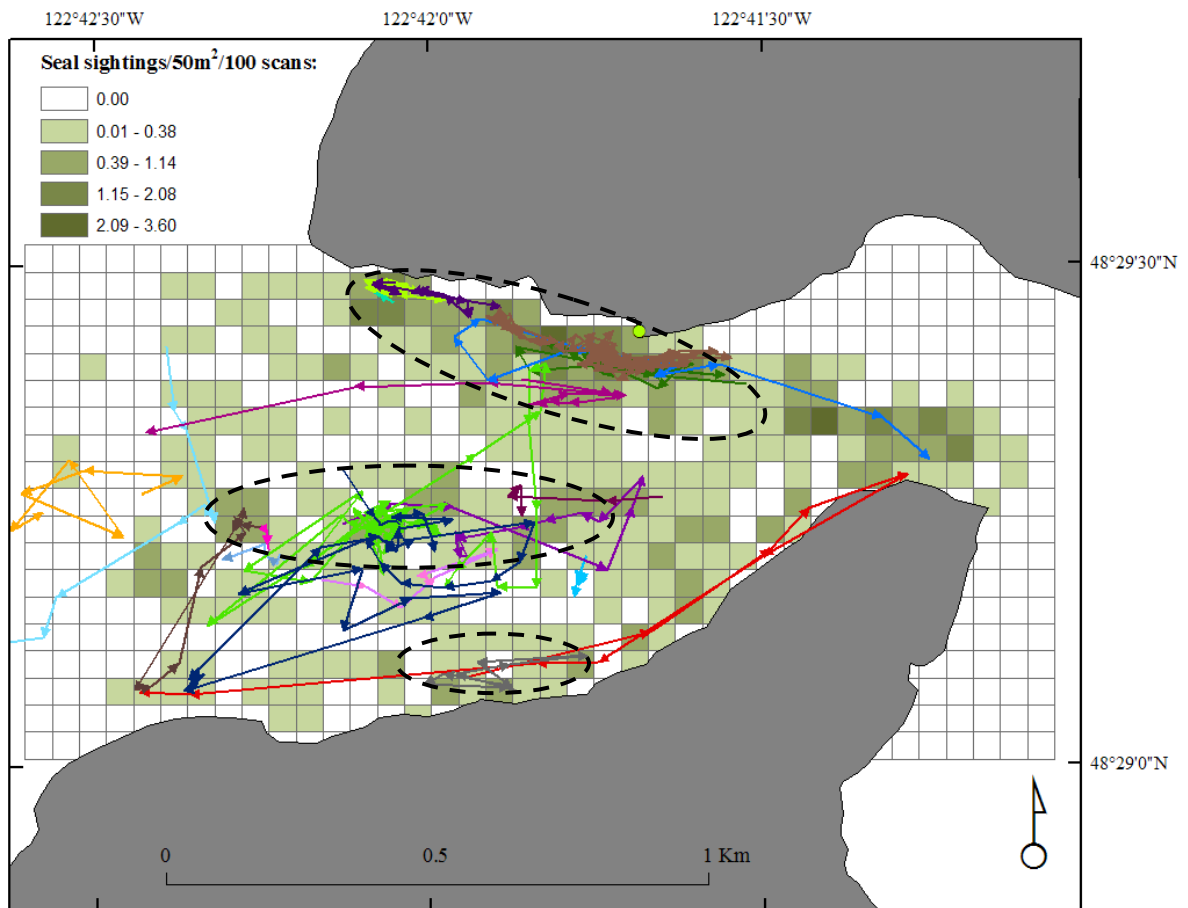
**Figure 12.** Harbor seal sightings (●) and predation events (☆) in relation to bottom depth (30m<sup>2</sup> grid; NOAA 2002) and the tidal rip (dotted outline) in the study area. Darker squares represent deeper bottom depths. The observation station is represented by a circle.



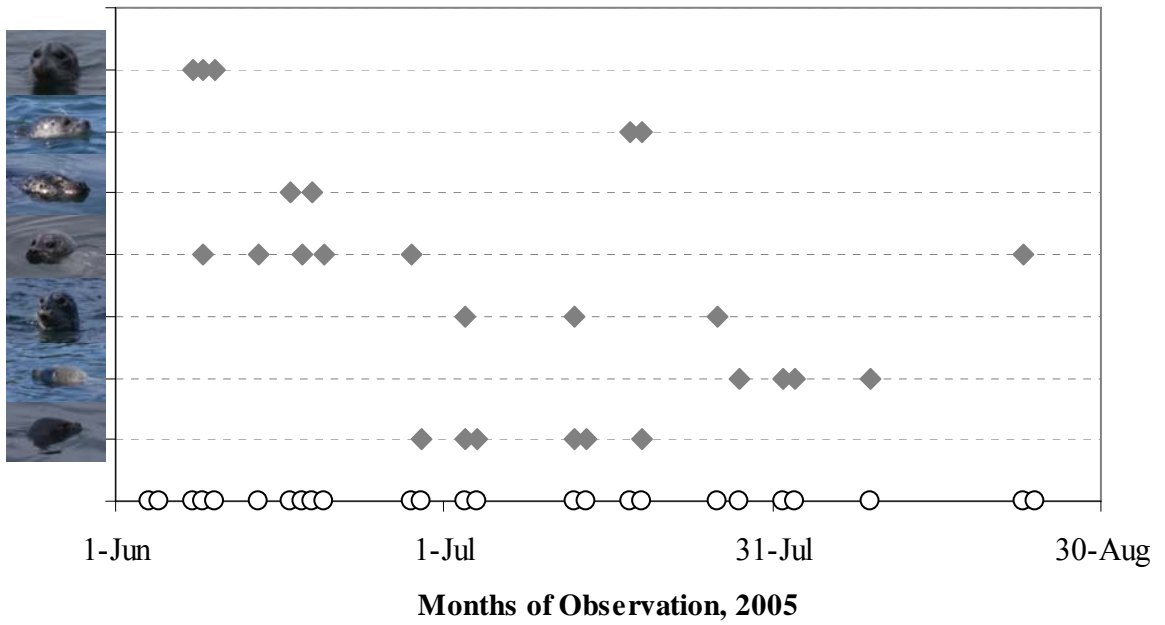
The distribution of focal observations agreed with the spatial distribution of seal sightings recorded during scan sampling (Figure 13). Out of 20 individuals tracked in the channel, 70% engaged in milling, 35% in traveling, and 5% in resting at least once during a track, while 15% of observations were categorized as unknown. Total frequencies sum to more than 100% because several tracked individuals engaged in more than one behavior while in the channel. Predation events, diving seabirds, and rapid subsurface pursuit swimming were frequently seen in association with milling seals, further supporting that this behavior was likely indicative of foraging activity. Milling occurred throughout the channel in areas of high sighting frequencies, except in the eastern end of Burrows Channel (Figure 13). Although tracking sessions were conducted in this area, they typically involved several seals and, thus, could not be included in analyses of individual behavior.

### **Photographic identification**

In 2004, photographs were taken opportunistically on 12 observation days in July and August. Of the 20 sets of digital photographs, 60% (12/20) could be used for identification. At least 7 individuals were photographed in 2004 and only one of these seals was photographed multiple times: once on 28 July and again on 4 and 5 August 2004. In 2005, 83 sets of digital photographs were taken of seals in the water close to the observation station, June through August. Of these sets, 68 (82%) contained images of sufficient quality to be used in analysis. At least 18 distinct individuals were photographed, and seven of these individuals were photographed on multiple days. The most frequently photographed seals ( $n=2$ ) appeared in the channel for 6 days each (Figure 14). In one case, all identifications occurred between 29 June and 19 July 2005. In the second case, the seal



**Figure 13.** Distribution of individual focal observations in Burrows Channel (2004 and 2005) overlaid on harbor seal sighting frequency in the study area (using a 50m<sup>2</sup> grid). Tracking efforts at the eastern edge of Burrows Channel were not overlaid because they were primarily of more than one seal and could not be included in analyses of individual behavior. Example tracks and areas where milling was recorded are highlighted by dashed ovals. The red track is an example of a traveling individual. The magenta track shows an individual that milled and then traveled west out of the channel.



**Figure 14.** Dates on which individual seals seen multiple times were identified, June-August 2005. Days on which opportunistic photographs were taken are represented by open circles.

was identified on 5 days between 9 June and 29 June, and then again on 23 August 2005 (Figure 14). No photographed seals were identified in both 2004 and 2005.

For seals that were photographed repeatedly within a given day, the average time spent near the observation station was  $1.6 \pm \text{SD } 1.1$  hrs. The seal that spent the longest time in proximity to the north shore of Burrows Channel within an observation period was photographed several times within 3.5hrs in 2004.

## DISCUSSION

### Number of harbor seals in and around candidate marine reserves

#### Regional scale interannual changes

The 2005 aerial surveys revealed changes in harbor seal haulout locales since the publishing of *The Atlas of Seal and Sea Lion Haulout Sites in Washington* (Jeffries *et al.* 2000). These shifts in haulout use are likely due to several factors: 1) new utilization or abandonment of specific haulouts, 2) the seasonal use of some haulouts, and 3) simple lumping or splitting of haulouts in geographic proximity by observers.

It may be difficult to assess the true extent of interannual change in haulout use by seals as the atlas is the product of a compilation of studies from the past several decades and is not a complete list of all sites; for example, sites used infrequently by a few animals have not been recorded as haulouts (Jeffries, personal communication<sup>2</sup>). However, new haulout sites (n=4) were recorded in this study and seals were never recorded at some of the historical haulout sites (Figure 3; Table 1). These results indicate that surveys should not be limited to known haulout sites, and suggest that harbor seals in this area may have shifted their haulout site use since publication of the atlas. Twenty-one percent of historical haulout sites were *not* used in 2005, which may indicate the abandonment of previously used haulout sites. Haulout abandonment has been attributed in part to human disturbance (Newby 1971, Allen 1991) and the San Juan Islands experience significant boat traffic during the summer months, which is a primary source of harbor seal disturbance (Suryan and Harvey 1999, Johnson and Acevedo-Gutiérrez 2007). Increased powerboat and kayak traffic may be contributing to

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<sup>2</sup> Steve Jeffries; Washington Department of Fish and Wildlife; 7801 Phillips Road SW; Tacoma, WA 98498; June 15 2007.

changes in haulout use in the study area. However, in Fidalgo Bay, removal of the haulout site (floating logbooms) is a more likely reason for not seeing seals there and may have contributed to the establishment of new haulout areas by displaced seals. Additionally, harbor seals appear to favor haulouts that increase their isolation (i.e., are farther from shore) and reduce the risk of predation by terrestrial predators (Nordstrom 2002). These factors may also be contributing to changes in haulout use in the study area. It is unlikely that site abandonment has occurred strictly since the atlas was published. Comparison with more recent survey data from WDFW may elucidate how much site abandonment or recruitment has occurred recently, as haulouts recorded as novel sites in this study are currently surveyed by WDFW (Jeffries, personal communication<sup>2</sup>). In addition, more surveys are needed to determine if the observed site abandonment is permanent and experimental approaches may be useful for describing factors affecting local haulout selection by harbor seals.

The seasonal use of haulouts, as well as the lumping or splitting of haulout sites by observers, tends to overemphasize the change in haulout site use (Table 1). Although some haulouts are not documented in the atlas, they may not necessarily be new, but rather seasonal sites (Table 1). Sites used exclusively outside of the pupping season (July and August) might not be recorded in the atlas because it is based largely on surveys conducted during August, especially since the 1990s (Jeffries, personal communication<sup>2</sup>). Year-long surveys were conducted historically (Calambokidis *et al.* 1979, Everitt *et al.* 1979) and those recorded sites are included in the atlas. However, at the time of those surveys, harbor seal numbers were at extremely low levels due to a state-financed bounty program in effect until 1960 (Newby 1973); thus, comparisons with historical surveys would be confounded by vastly different population sizes. Additionally, I tended to split haulouts that have been

lumped for the purposes of WDFW surveys (Table 1). These different reporting methods are in part due to different research goals: WDFW flies yearly surveys to examine population level trends over time (Jeffries *et al.* 2003), while this project concentrated on increased effort in a specific geographic area over an extended temporal window to look at seasonal changes in harbor seal numbers and distribution. Therefore, geographic specificity for haulout locations was preferred for this study. Lastly, inclusion of sites where few seals were seen irregularly in 2005 also overemphasizes these differences, as these sites are not new haulouts, but rather areas of infrequent use and are simply not represented as haulouts in the atlas (Jeffries, personal communication<sup>2</sup>; Table 1). Continued surveys throughout the year will provide information on the transience (or stability) of seasonally-used haulouts as well as the degree to which permanent shifts in haulout use have occurred.

### **Regional scale intra-annual changes**

The mean number of non-pup seals hauled out in the study area approximately doubled from April to September and was significantly greater during the pupping season (July and August) than during pre-reproductive months (April and May; Figure 4). Three potential explanations exist for this increase in total number of seals over time: 1) the same number of seals was present throughout the study period and the change in numbers hauled out was due to a change in behavior of the seals, 2) seasonal movements brought seals into the study area temporarily, or 3) animals migrated into the area.

Most studies report peak numbers of seals ashore at a time of year that corresponds with pupping and molting (Brown and Mate 1983, Johnson and Jeffries 1983, Harris *et al.*

2003). The sample period of this study encompasses parts of three life-history seasons of harbor seals in this area: the pre-reproductive period (April to mid-June), the reproductive period (July through August), and the molting period (from mid-August on). These seasons correspond to different priorities for harbor seals. They are more likely to spend time ashore during pupping, when mothers are tending pups (Huber *et al.* 2001), and molting, when seals experience a thermoregulatory benefit to resting on land (Boily 1995) although there are age and sex differences to these trends (Thompson *et al.* 1989, Härkönen *et al.* 1999, Huber *et al.* 2001). Thus, the increasing number of seals on land recorded in this study may simply reflect a change in haulout behavior during the pupping and molting seasons rather than a true change in the number of seals utilizing the eastern San Juan Islands.

It is possible that the increasing numbers recorded here could be attributed to seasonal movements of seals within the larger geographic area because the spatial extent of the area sampled in this study is only a portion of the San Juan Island Archipelago and the inland stock of harbor seals (Olesiuk *et al.* 1990b). Harbor seals are believed to shift their haulout-site use to be in proximity to seasonal or local prey aggregations (Brown and Mate 1983, Jeffries 1986, Thompson *et al.* 1996). Historical monthly surveys recorded increasing numbers of harbor seals in most, but not all, subareas within the inland waters of Washington from April through September (Everitt *et al.* 1979). Movements of seals into the eastern San Juan Islands area to take advantage of prey aggregations from June onward could explain the increased number of seals counted during those months. Studies examining harbor seal diet and prey distribution as well as seal tagging studies may be useful for determining whether seasonal movements explain the observed increase in seals in the study area.



Two genetic stocks exist in Washington State: the coastal stock and the inland waters stock, with little to no genetic exchange occurring between them (Lamont *et al.* 1996). Therefore, it is unlikely that harbor seals from the outer coast are immigrating into Puget Sound during the survey period. However, seals utilizing Puget Sound are more genetically diverse than other stocks in Washington, Oregon, and California (Lamont *et al.* 1996). These results, along with similar pupping phenology between Puget Sound and the Strait of Georgia, Canada (Bigg 1969, Temte *et al.* 1991), suggest the presence of gene flow between the two areas some time in the past. More recent genetic studies of seals north of Puget Sound indicate that harbor seals from Puget Sound and southern British Columbia comprise a single population (Burg *et al.* 1999). Thus, it is possible that seals from southern British Columbia could be immigrating into the study area. However, surveys in the Strait of Georgia and southern Vancouver Island (Olesiuk *et al.* 1990b) as well as Washington State (Jeffries *et al.* 2003) report an overall increasing population trend in harbor seal abundance, thus unidirectional immigration from British Columbia to the San Juan Islands is unlikely. Future tagging studies as well as surveys over a larger spatial scale may favor one of these explanations over the other.

**Molting season.** In various areas, both pupping and molting seasons have been reported as periods with peak seal counts (Jeffries 1986, Thompson *et al.* 1997). Historically, peak counts in the San Juan Islands and Eastern Bays have been recorded in August or September (Calambokidis *et al.* 1979, Everitt *et al.* 1979). The present study recorded the peak number of seals one-year-old or greater in September (Figure 4), which represented the onset of the molting period in this area in 2005. Approximately 20% of the seals I photographed during the September survey appeared to have molted, suggesting that

the molting season in 2005 likely continued into October. Continued surveys over longer temporal periods will clarify the possibility for interannual changes in the timing of molt as well as the time of peak counts of seals ashore in this area.

In addition to increased numbers of hauled out seals, the latter half of the survey months, July through September (pupping through the onset of molting), also corresponded with an increased number of haulout sites used by harbor seals (Figure 4). Surveys in the Columbia River area also found an increase in the number of sites used by seals during the pupping season, which were attributed to the establishment of nursery areas by mothers with pups (Jeffries 1986). After the pupping season, there was a decrease in the number of haulouts used in the Columbia River area (Jeffries 1986). Similar trends have been reported in Maine where the number of haulout sites used by seals decreased from the pupping to the molting season (Dow *et al.* 2005). In my study area, I did not see this decline in areas used by harbor seals, further supporting the idea that the molting season extended past September. Hence, surveys that continue into the fall in the San Juan Islands may also show a drop off in the number of haulouts used by seals. Conversely, if seals are moving into the study area to take advantage of local and seasonal prey aggregations, the number of areas used may not decrease in the fall (Brown and Mate 1983). Although historical aerial surveys did not record peak counts in the larger San Juan Island area past September, small-scale seasonal movements could redistribute seals into the eastern part of the archipelago in the fall, resulting in increased local counts. Continued long-term surveys and analysis at relevant spatial scales may clarify these possibilities.

**Pupping season.** Studies in the San Juan Islands report the maximum number of pups on land during the last week of July and first week of August (Suryan 1995, Johnson

and Acevedo-Gutiérrez 2007). It is unlikely that the maximum pup count recorded here (August 18-20) represents a delay in the timing of peak pupping; instead it is likely an artifact of my longer sampling interval compared to the minimum twice weekly effort of other studies (Suryan 1995, Johnson and Acevedo-Gutiérrez 2007). Additionally, the July and August pup counts from aerial surveys were similar and may have bracketed the true peak in pup numbers (Figure 4).

Application of the correction factor proposed for Washington and Oregon (Huber *et al.* 2001) to counts during the pupping season resulted in a mean of  $3,534 \pm \text{SD } 128$  seals utilizing the study area during July and August 2005 (Figure 4). I chose to use the regional correction factor rather than averaging local correction factors for the inland stock, or for the San Juan Islands and the Eastern Bays (Huber *et al.* 2001) - areas that were partially included in my surveys. Either of these treatments would have resulted in an elevated correction factor; thus, the approximation presented here should be considered a minimum abundance estimate.

If the correction factor applied to the pupping season counts provides an accurate abundance estimate of the resident population size, then approximately 3,100 seals may be found in the study area year-round (subtracting the average corrected count of pups ( $n=417$ ) from the abundance estimate). Consequently, one would expect an average correction factor of 2.3 for April-May based on aerial survey counts. Although correction factors do not currently exist for other times of the year in this area, this value is within the range for correction factors outside the breeding season in other geographic areas (1.2-11.1; summarized in Huber *et al.* 2001). Thus, it is possible that the increased number of seals from April through September was a result of a change in haulout behavior. Development of

local correction factors for other times of the year as well as long-term tagging of seals may help distinguish between behavioral changes or seasonal movements as explanations for increased numbers of seals ashore in this area.

### **Influence of tidal availability on haulout use**

I surveyed haulouts that were available at the majority of tide levels as well as haulouts that were available only at low tide (Figure 5). Tide height significantly affects harbor seal haulout behavior (Schneider and Payne 1983, Pauli and Terhune 1987a, Reder *et al.* 2003); and is an important influence in Puget Sound and the San Juan Islands (Calambokidis *et al.* 1979, Hayward *et al.* 2005) where diurnal tidal ranges commonly exceed 2.4m (NOAA 2004) and many haulouts are submerged at extreme high tides. Although some “limited” haulouts did show increasing trends in the number of seals ashore over time, this pattern was much more pronounced for “unlimited” haulouts (Figure 6). This suggests that the regional haulout patterns described above may be driven by the increased use of tidally “unlimited” sites later in the summer. The linear relationship from April through September is not surprising given that I did not fully sample the molting period, after which I would expect the number of seals ashore to decrease.

Including tidal classification improved the fit of the mixed-effect model (Table 2b). This is likely due to the high variability of the fitted slopes for “limited” haulouts, relative to the “unlimited” sites (Figure 6). The differing slopes for “limited” haulouts may be partially explained by the exposure profiles of individual haulouts in this category. The availability of these haulouts is likely unique given the unequal effect that the same tide level would have

on haulouts of different topographies (Jeffries 1986). This extreme variability is not seen in the “unlimited” haulouts, which should be largely available regardless of tide height.

The importance of tidal availability is further evident in the significant month-by-tidal interaction in the mixed-effect model (Table 2c). Over the sampled months, harbor seal numbers generally increased on “unlimited” haulouts but remained fairly constant at “limited” sites (Figure 6). Similarly, examining the summed average monthly counts for each haulout classification reveals that counts increased for “unlimited” haulouts, but that the number of seals ashore on tidally “limited” sites was approximately the same from April through September, resulting in a significant interaction term (Figure 7).

One possible explanation for the differing patterns between haulout types is that there are fewer seals on “limited” haulouts because that type of haulout is submerged. However, the April/May and July/August surveyed low tides were approximately equivalent, thus the number of seals ashore on each haulout type should have been similar. Yet there were more seals utilizing “unlimited” haulouts in July and August relative to April and May (Figure 7). Hence, the observed pattern is not solely due to differential tidal availability. Rather, the increased use of “unlimited” haulouts during the pupping season may reflect a preference for sites that are exposed for longer periods. This preference may extend into the molting season as suggested for other areas (Jeffries 1986).

Similarly, the significant month effect may be a proxy for a seasonal change in seal behavior (Table 2c). If seals are hauling out for longer periods during pupping and molting, as has been suggested in other areas (Thompson *et al.* 1989, Frost *et al.* 2001), then this may help explain the increased number of seals ashore during these seasons. It is possible that during the pupping and molting seasons, seals were looking for haulout sites earlier in the

tidal cycle. In that case, they would likely find “unlimited” haulouts with greater frequency than “limited” sites, and therefore be counted more frequently on “unlimited” sites when surveys were flown, explaining the steeper predicted slopes for “unlimited” sites.

Hauling out earlier in relation to the time of low tide may also help explain some of the differing trends associated with “limited” haulouts. Several of the “limited” sites in close proximity to each other exhibited opposing trends in the numbers of seals hauled out over time (Figure 5). Although these haulouts would be exposed to the same tidal level, the individual topography of the haulouts likely influenced seals’ specific use patterns of these sites. For example, harbor seals have been observed swimming in the water close to haulouts prior to low tide (Olesiuk *et al.* 1990b). In the study area, West Boulder Reef is a low, flat shoal and East Boulder Reef is sloping and more elevated. These haulouts are less than 500m apart. As East Boulder becomes exposed earlier in the tidal cycle, nearby seals waiting to come ashore may haulout there rather than wait for West Boulder to emerge. As with regional trends, tagging of individual animals to examine seasonal differences in local haulout behavior would be informative.

There are more sophisticated ways of examining the effect of tide on specific haulouts and haulout behavior (e.g., Jeffries 1986, Hayward *et al.* 2005). However, the assignment of haulouts into tidal availability classes provides an approximation of the tidal windows for which haulouts may be exposed and hence available to seals. Moreover, tidal availability and time of year appear to help explain the haulout use by seals in the study area. However, these factors may interact with others, such as additional environmental influences (Small *et al.* 2003), the reproductive phase of individuals (Huber *et al.* 2001), age-sex class differences in behavior (Thompson *et al.* 1989, Härkönen *et al.* 1999), and seasonal

movements in response to prey abundances (Brown and Mate 1983, Jeffries 1986), to shape haulout-use patterns in the San Juan Islands. Long-term tagging and observational studies may elucidate the range of haulout-use patterns and additional important covariates in the San Juan Islands.

### **Bottomfish and harbor seal distribution**

SCUBA surveys revealed differences in bottomfish density on a small spatial scale within the Burrows Channel candidate marine reserve (Figure 8). Non-metric multidimensional scaling ordination performed on a subset of these data as part of a related project indicated that bottomfish abundances and species compositions were different between these dive sites and that these differences persisted for the summer season (Weispfenning 2006), supporting the trends reported here. Further, differences between sites were maintained through several summers of data collection (Weispfenning 2006). These two conclusions mirror the results obtained by this study with the mixed-effects analysis.

More seals were sighted near dive site A than dive site B (Figure 9). Given that seals were frequently detected at distances of 500-800m (Figure 11), it is unlikely that the reduced number of sightings near site B (500-700m from the observers) was an artifact of distance from the observers. Although more seals were sighted in proximity to the high bottomfish density site, this correlation was not confirmed by the species recorded as harbor seal prey. No bottomfish were identified as prey during tracking sessions. Although scan sampling results suggested an aggregative response (i.e., more harbor seal sightings near an area of high bottomfish density), there was a lack of a functional response (using predation events as an indicator). During all observations (over 606 hours of scan sampling and focal

observations), only one potential bottomfish predation event was recorded during hourly scan sampling and a single confirmed bottomfish predation was witnessed off-effort. Instead, small pelagics were the predominantly observed prey during focal observations. Surface trawls from Padilla and Skagit Bays, areas immediately northeast and southeast of my study area, caught primarily Pacific herring, surf smelt (*Hypomesus pretiosus*), sand lance, and juvenile salmonids during the summer months of 2003 (Rice 2007). It is likely that these species were some of the harbor seal prey identified as small pelagic fish in this study. The high percentage of unidentified prey recorded during tracking observations could be interpreted as rocky reef bottomfish prey. However, the low occurrence of bottomfish in harbor seal scat samples from the San Juan Islands would suggest that it was unlikely that many of the unidentified prey recorded in this study were bottomfish (Lance and Jeffries 2006).

Because this study was conducted during the summer months and during daylight hours, it did not address the potential for bottomfish predation during other times of the year or day. In 2005, WDFW researchers collected scat samples during three seasonal windows, reporting a low overall weighted average of bottomfish in year-round diet of harbor seals in the San Juan Islands (Lance and Jeffries 2006). Instead, Pacific herring, salmonids, gadids, Pacific sand lance, and northern anchovy were the major local harbor seal prey (Lance and Jeffries 2006). Although some of these species were important year-round, salmon were a principal prey item during the summer/fall period, and of the identified salmonids, pink salmon (*Oncorhynchus gorbuscha*) were especially prevalent (Lance and Jeffries 2006).

Pink salmon run every other year (e.g., 2005), and years with low abundances of non-bottomfish prey may result in a shift in harbor seal diet and increased predation on



bottomfish given the behavior of the species to consume locally abundant prey (Olesiuk 1993, Tollit *et al.* 1997a, Hall *et al.* 1998, Browne *et al.* 2002). The prey recorded in my study were for predation events witnessed in 2004 and 2005, the former being a year that pink salmon did not return to spawn. However, the bottomfish predation that I recorded was not higher in 2004 than in 2005. Comparison of these results with additional diet data collected by WDFW in 2006 (the most recent non-pink year; Lance, personal communication<sup>3</sup>) should elucidate whether or not bottomfish are preyed on with greater frequency when other, more abundant prey are not available.

Similarly, the high frequency of rockfish in harbor seal diet samples in southern Oregon and California (NMFS 1997) compared to some sites in Oregon (Wright *et al.* 2007) as well as local results (Olesiuk *et al.* 1990a, Lance and Jefferies 2006) deserves further study. It may be that rockfish density is substantially higher in some regions of Oregon and California, which might explain the prevalence of rockfish in harbor seal diet in those areas as harbor seals are opportunistic predators and typically consume locally and seasonally abundant prey (Olesiuk 1993, Tollit *et al.* 1997a, Hall *et al.* 1998, Browne *et al.* 2002). However, at least two factors preclude a straightforward analysis of past studies: lack of concurrent bottomfish density data and changes in prey identification methodologies. Few harbor seal diet studies have concurrently documented prey abundance (but see Tollit *et al.* 1997); instead most rely on the timing of fish runs (e.g., Olesiuk 1993, Browne and Terhune 2003, Middlemas *et al.* 2006) or fisheries catches (e.g., Trumble 1995) to infer prey density, thus specific information on bottomfish density is frequently unavailable. Additionally, prior to 1990 most diet studies relied exclusively on otoliths for fish identification; however, more

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<sup>3</sup> Monique Lance; Washington Department of Fish and Wildlife; 7801 Phillips Road SW; Tacoma, WA 98498; June 4 2007.

recent studies have shifted to using numerous skeletal structures for fish prey identification. Using many structures increases the resolution of diet studies (Olesiuk 1993, Cottrell *et al.* 1996) and often affects the suite or importance of detected prey (Browne *et al.* 2002, Orr *et al.* 2004) because species with fragile otoliths are generally under-represented in otolith-only studies (da Silva and Neilson 1985, Dellinger and Trillmich 1988, Harvey 1989, Tollit *et al.* 1997b, Orr and Harvey 2001). Although rockfish otoliths are generally robust (i.e., all-structures methods may not greatly affect the frequency of rockfish detection), comparing different methodologies should be done with care. Nevertheless, meta-analysis of historical seal diet studies and regional rockfish catch-per-unit-effort data may be informative as to the range of functional responses that might be expected if local bottomfish densities were to increase.

Overall there were few predation events recorded during tracking (34 within 78.5hrs of focal observations). Several factors could have contributed to this result: 1) predation events occurring underwater, 2) distance between observers and seals, and 3) cryptic consumption of prey brought to the surface. Harbor seals consume many prey items underwater (Bowen *et al.* 2002). In general, surface predations appeared to underestimate harbor seal foraging, as foraging behavior, i.e., milling, was observed in other areas of the channel during focal observations of individual seals. This may be problematic for enumerating rockfish predation events as diet results indicate that the rockfish consumed by seals are juvenile fish; age 2 rockfish otoliths were recovered from seal scat samples in the San Juan Islands (Lance and Jeffries 2006). The size of juvenile rockfish varies by species (Love *et al.* 2002), but for the species seen in Burrows Channel (excluding Puget Sound rockfish; Table 3), age 2 fish range in size from 10 to 30cm (estimated from von Bertalanffy

growth curves published in Love *et al.* 2002). A study at Sable Island, Nova Scotia reported that male harbor seals commonly consumed 15-30cm flounder (Pleuronectidae) underwater; hence, it seems that harbor seals could consume juvenile rockfish without bringing them to the surface. Rockfish in Burrows Channel are within this size range (Table 3), with the largest rockfish recorded at 40cm. Secondly, distance to tracked seals may have influenced the observer's ability to detect predation events, especially if the predominant prey were small pelagic fish. This possibility is also supported by the presence of milling behavior in areas where no predation events were recorded. Lastly, because seals generally bring large prey to the surface and thrash the prey to subdue it and rip it into smaller pieces (e.g., Brown and Mate 1983), seal consumption of large fish is commonly considered easily detectable (Zamon 2001). During this study, however, several highly cryptic consumptions of large fish were observed. In one case, during scan sampling, a seal was seen with a suspected salmonid at the surface. The prey was handled immediately below the water's surface and, until the tail was swallowed, it was difficult to ascertain whether the seal actually had a fish. Although this was not always the case and typical consumption events of large prey were witnessed, cryptic predation of large prey may have contributed to an underestimation of large-prey capture based on surface observations.

It is clear that attempting to document predation events and determine prey type from land underestimated the frequency of occurrence of this behavior as well as the prey consumed in this study. Additionally, it was difficult to ascertain the identity of prey species unless the seal was close to the observation station. Nonetheless, general categories of prey could be identified based on body shape or prey size. Although year-round diet studies are critical for determining the amount of seal predation on bottomfish and other species

regionally, these studies provide no specific information on where prey items were consumed. Land-based observations of Burrows Channel, coupled with diet studies from nearby haulouts may be informative as would studies including underwater footage of foraging seals (e.g., Bowen *et al.* 2002).

### **Seal habitat use of a candidate marine reserve**

#### **Scan sample data**

Harbor seals were frequently seen in Burrows Channel throughout the study. Although there was a maximum of 8 seals recorded in the channel during a single hourly count, average counts were considerably lower (1-2 seals per count). Increased encounter rates in 2005, especially in June and July, relative to 2004 encounter rates, may be due to increased prey during the 2005 field season. Adult pink salmon, which were important seal prey in early September 2005 (Lance and Jeffries 2006), run only on odd-numbered years (e.g., 2005). Exploitation of this prey source in Burrows Channel could explain increased seal encounter rates. However, a concomitant change in methodology in 2005, increasing average scan times from 3min to 5min per sector, may have also contributed to increased encounter rates. Scan times were increased based on the average time between surfacings of tracked individuals in 2004, which was approximately 5.5min, to minimize the chance of missing underwater seals during scans. Future land-based observations with standardized methodologies may elucidate interannual differences in seal encounter rates in Burrows Channel.

Bathymetry and tidal currents influence the at-sea distribution of many marine predators including harbor porpoise (*Phocoena phocoena*; Raum-Suryan and Harvey 1998), Dall's porpoise (*Phocoenoides dalli*; Miller 1989), Black-legged kittiwakes (*Rissa tridactyla*; Irons 1998), and others. The spatial variation in sighting frequencies in Burrows Channel demonstrates differential habitat use on a very small scale, with seals being sighted most frequently in tidal rip areas and in shallower waters (Figure 12). Studies examining dive parameters of harbor seals demonstrate that these animals forage at a range of depths (Lesage *et al.* 1999, Gjertz *et al.* 2001, Hastings *et al.* 2004). However, harbor seal foraging behavior is also associated with both shoals and tidal currents (Suryan and Harvey 1998, Zamon 2001). These two factors appear to be influencing harbor seal habitat use, and possibly foraging, in Burrows Channel on a very fine scale (Figure 12).

Quantification of fine-scale habitat use is not common in marine mammals given the difficulties of following animals that spend much of their time underwater while at sea. Land-based observations provided information on seal habitat use at a scale of 10s of meters within a 0.85km<sup>2</sup> study area. Given the patterns of habitat use documented here, it is apparent that environmental variables such as tidal currents and bathymetry influence harbor seal habitat use even on very fine spatial scales; scales relevant to marine reserve design as reserves tend to be small in area.

Harbor seal associations with tidal and topographic features are likely driven by prey availability in these areas. In the San Juan Islands, actively migrating salmon have been reported to move along shorelines and orient along axes of tidal currents (Stasko *et al.* 1973, Stasko *et al.* 1976), which may concentrate these fish. Tidal currents may also influence small pelagic fish distribution or behavior, making them more accessible to foragers (Zamon

2003). Additionally, Burrows Channel is a constricted passage that might benefit harbor seal foraging efficiency. Zamon (2001) reported significantly more large fish captures than expected near a constricted passage between San Juan and Lopez Islands (with the narrowest point being 0.7km across) by harbor seals. Burrows Channel is less than 1km across at its widest point and the highest frequency of seal sightings occurred at the eastern end of the channel where the shores are less than 0.5km apart and there is high tidal activity (Figure 12).

Bathymetric and tidal features may also explain the high density of bottomfish at Site A, where high-complexity habitat and tidal influences on small pelagic prey similarly benefit rocky reef bottomfish. Hence, indirect effects of the environment and prey availability on both seals and bottomfish may more accurately explain the correlation between high seal numbers and high densities of bottomfish as well as the discrepancy between an apparent aggregative response and lack of a functional response. Additionally, many other marine predators (e.g., harbor porpoise, rhinoceros auklets (*Cerorhinca monocerata*), gulls (*Laridae*), common murre (*Uria aalge*), and others) were observed in Burrows Channel, suggesting that this area is important for numerous species and offering abundant opportunities for continued research in this area. Investigations relating prey abundance and distribution to bathymetric and tidal effects in Burrows Channel may elucidate the contribution of these factors to predator-prey interactions in the study area.

Burrows Channel also experiences significant boat traffic as a marina is located at the northern end of Burrows Bay (to the east of Burrows Channel). Although information on the rate of boat traffic was recorded during this study (approximately 10 power boats passed through the channel each hour during the summers of 2004 and 2005), no data on boat routes were collected. If boats were predominantly navigating the channel in deeper waters,

avoidance of boats could be a contributing factor to harbor seal distribution in Burrows Channel. Personal observations suggested that boats did not traverse the channel strictly through deeper waters, and harbor seals were largely indifferent to passing boats unless boats were extremely close. Although harbor seals are frequently disturbed by boats while they are ashore (Suryan and Harvey 1999, Johnson and Acevedo-Gutiérrez 2007), I am not aware of any studies examining the effect of boat traffic on seals in the water. Given the high rate of boat passage in Burrows Channel, future studies on harbor seal distribution in that area should examine boat traffic as a potential influence.

### **Focal observations and photographic identification**

Focal observations indicated that individual seals spent on average at least one hour in the candidate marine reserve, but could spend up to 3.5hrs in the area. This maximum track time was confirmed by photographic identification of individual seals in 2004 and 2005, where multiple sets of photographs were taken of identifiable seals over several hour periods. However, since both tracking and photographic efforts were suspended at the end of 4-hr observation periods, these should be considered minimum estimates for the amount of time that individual seals may have spent in Burrows Channel.

Photographic identification also revealed that known individuals were returning to the channel over the summer months, and that they were utilizing the same area within the channel over time. This fine-scale fidelity is likely related to foraging as harbor seals exhibit fidelity to one or more foraging sites (Suryan and Harvey 1998, Nickel 2003) and focal observations revealed a high incidence of predation near the observation station (Figures 10

and 12). The lack of matching identifications across years may have been due to the low sample size of quality photographs in 2004.

Photographic identification from shore has its limitations (e.g., seals must be in proximity to observers); however, it was useful for quantifying the visitation of seals to a restricted area within Burrows Channel. Although tagging of seals from haulouts near candidate marine reserves may provide similar information on the fidelity of individuals to particular areas (e.g., candidate reserves), the cost associated with tagging sufficient animals could be prohibitive. Instead, surveys of the reserves themselves may prove more informative. Digital images for this study were collected opportunistically. Systematic effort may reveal fidelity to Burrows Channel among years, as well as provide a photographic catalog of individuals that visit this area.

Tracking of individual seals indicated that milling was the predominant activity in Burrows Channel; however, resting and traveling behaviors were also recorded. Milling activity was not limited to the north shore of Burrows Channel, although that was the area with the most recorded predation events (Figure 10), further supporting the likelihood that surface predations were a minimum estimate of harbor seal foraging. The frequency of resting and traveling may have been underestimated, as they were harder to detect than milling. Resting and milling could be confused as both involve repeated diving in an area, and traveling seals were more difficult to track as they typically moved large distances between surfacings.

Using Suryan and Harvey's (1998) estimate that a seal's primary haulout site was typically less than 5.6km from its frequented foraging area, seals foraging in Burrows Channel are likely from nearby haulouts. However, a study in the San Francisco estuary in



California reported that all tagged seals, with one exception, foraged within 10km of known haulouts, while most foraging took place 1-5km from haulouts (Nickel 2003). Individual seals also exhibit long-distance movements, presumably between haulout sites and/or foraging locations (Pitcher and McAllister 1981, Thompson *et al.* 1996, Lowry *et al.* 2001). This would suggest that seals visiting Burrows Channel could come from outside the area covered by aerial surveys in this study. Future studies on the movements and foraging trips of harbor seals locally may generate a more comprehensive foraging range for seals in the San Juan Islands.

### **Implications for marine reserves**

#### **Regional scale**

**Spatial and temporal scale of aerial surveys.** Satellite and radio telemetry studies demonstrate that harbor seals generally exhibit a high degree of fidelity to haulout sites (Pitcher and McAllister 1981, Härkönen and Hårding 2001) and foraging areas (Suryan and Harvey 1998, Nickel 2003, Lesage *et al.* 2004). That is, seals frequently use one or several nearby areas on land or at sea, especially within a given season. Seals tagged and tracked during the breeding season typically have smaller ranges than at other times of the year (Thompson *et al.* 1989, Thompson and Miller 1990, Thompson *et al.* 1996, Lowry *et al.* 2001). Previous tagging studies in the northern San Juan Islands have been limited to a small number of males during the breeding season (Suryan and Harvey 1998) and the average foraging distance (5.6km) reported in that study was used to determine the extent of aerial surveys for this study. Future tagging studies in the study area should incorporate non-summer seasons as well as females (adults and subadults) to generate a more complete home

range extent for seals in the San Juan Islands. Coverage of baseline aerial surveys may need to be adjusted accordingly, as the area surveyed in 2005 may be too small to detect aggregative responses within the larger geographic area important to seals.

Monthly surveys were flown to examine intra-annual changes in seal numbers and distribution within the study area, thereby creating a baseline for future comparison after candidate areas have been protected. Because haulout behavior is influenced by many factors, including the age-sex class of individuals (e.g., adult females are more likely to be ashore during the pupping season (Huber *et al.* 2001) and males more likely to be on land during the molt (Thompson *et al.* 1989)), multiple surveys per year may provide a more complete picture of harbor seal population trends (Thompson *et al.* 1997). Additionally, multiple surveys per year have a greater chance of detecting change more quickly than single annual surveys (Thompson *et al.* 1997). If harbor seals do exhibit an aggregative response to established marine reserves, rapid detection may assist managers in realizing the conservation goals of the reserve.

**Predation on rocky reef bottomfish.** As many as 3,100 seals may be using the study area year-round and have the potential to exert substantial predation pressure on local fish stocks. Currently, harbor seals in the San Juan Islands do not appear to be consuming large numbers of rocky reef bottomfish prey (Lance and Jeffries 2006, this study). It may be that bottomfish occur in such low densities that seals do not regularly encounter them or perhaps seals in this area prefer other prey species. However, several fish species that appear to be important prey to seals in Puget Sound are species of concern or have depressed stock levels (Gaydos and Gilardi 2005, Stick 2005) and if they dwindle further, seals may focus on bottomfish (Lance and Jeffries 2006). Multiple year diet studies will better describe the level

of bottomfish predation as well as how inter-annual changes in the availability of more common prey may influence harbor seal diet.

It is possible that even low intensity predation could affect bottomfish stocks due to their reduced population levels and resultant low fecundity (Myers *et al.* 1999). Additionally, if bottomfish habitat is protected and the species recover, seals may encounter bottomfish more frequently and consume more of these species. In the event that candidate marine reserves are implemented and successful, either of these scenarios may create a situation in which seal predation impedes the recovery of bottomfish populations. Modeling studies may help managers gauge whether even the currently low level of seal predation is detrimental to bottomfish stocks. Models will also need to include other rockfish predators such as teleosts (e.g., other rockfish, kelp greenling (Hobson *et al.* 2001), lingcod (FishBase 2007)), seabirds (Burger *et al.* 1993, Nur and Sydeman 1999, Hedd *et al.* 2002), and other pinnipeds that migrate through the San Juan Islands (NMFS 1997), among others. Trophic relationships in northern Puget Sound are quite intricate (Simenstad *et al.* 1979) and neither harbor seals nor rocky reef bottomfish exist in isolation. Hence, well-parameterized models will also necessarily be complex and besides the previously mentioned predators they should also include current levels of human take.

In 2004 and 2005, human harvest of most rockfish species and lingcod was allowed at low levels, a practice that still continues (WDFW 2007). Regardless of existing fishing regulations, off-season lingcod catches and excessive rockfish harvest were witnessed in one of the candidate reserves during this study. Increased enforcement is expensive, but may be a necessary step to truly assess anthropogenic mortality of bottomfish. Because of rockfish life-history traits, even low levels of human fishing could provide the force necessary to keep

these populations suppressed. Thus, accurate measures of total human harvest and bycatch are critical to inform useful models. Overall predation pressure on rocky reef bottomfish by humans and other natural predators could also be used to model future scenarios of recovery and predator response, as well as forecast possible outcomes of various management and predation or harvest scenarios.

**Influence of tidal availability on haulout use.** Although tidal availability of haulouts may influence haulout behavior of harbor seals, it is unclear what effect this may have on the foraging patterns of harbor seals in proximity to the candidate marine reserves. As “unlimited” haulouts are widely available in the San Juan Islands, increased use of “unlimited” haulouts during certain seasons is not likely to substantially redistribute foraging activity of these central place foragers.

### **Within a candidate reserve**

The differential densities of rocky reef bottomfish within a candidate reserve indicate that reserve locales should be carefully chosen, and that bottomfish densities cannot be extrapolated over larger spatial extents. It also emphasizes that fine-scale predator responses may be important concerns in reserve design and management, especially given the generally small size of marine reserves. How predator response is measured (presence *versus* foraging frequency) is also an important consideration. As documented in this study, although seals were frequently seen in Burrows Channel, they did not appear to be foraging primarily on rocky reef bottomfish.

Diet studies are crucial for continued study of harbor seal impact on bottomfish stocks on a regional scale; however, continued observations at reserve sites before and after

protection are useful for determining 1) if seals increase visitation to the area after protection, and 2) if predation on protected species increases within the site, particularly for assessing functional responses to marine reserve protection. Harbor seals exhibit type-3 (sigmoid) functional responses to prey pulses (Middlemas *et al.* 2006). This type of response can result in the stabilization of prey populations at low levels (Andersson and Erlinge 1977). In the Atlantic, grey seals have been implicated as one factor that may be contributing to the failure of Atlantic cod to recover despite reduced fishing pressure (Bax 1998, Bundy 2001, Fu *et al.* 2001, Trzcinski *et al.* 2006). Thus, there is the potential that harbor seal predation could inhibit the recovery of rocky reef bottomfish. However, prey profitability and handling time may also influence seals' prey choice (Bowen *et al.* 2002) and should be considered in developing functional response curves for multi-prey systems. Although type-3 functional responses have been demonstrated for harbor seals in simple systems (Middlemas *et al.* 2006), more complex systems may not behave the same way. Careful monitoring of predator-prey interactions before and after reserve establishment may aid in assessing harbor seal functional responses and inform adaptive solutions to community interactions resulting from marine reserve protection.

The patterns of habitat use by seals in Burrows Channel, as well as return visits by individual seals, suggest that this area may be a preferred area as its bathymetric and tidal characteristics are favorable for foraging. Similar land-based observations should be conducted at other candidate reserve sites to determine if the sighting frequencies recorded here are exceptional or standard for this area. It may be that Burrows Channel is not an ideal location for a marine reserve as harbor seals frequently visit the area. However, without considering predator-prey interactions, Burrows Channel has the second highest bottomfish

density of all eight candidate sites, as well as relatively high scores for habitat complexity and species richness, making it a preferred candidate site based on those criteria (Weispenning 2006). It may be that some of the factors that make it attractive to seals also make it favorable for bottomfish. Habitat complexity coupled with the tidal activity along the north shore of Burrows Channel may contribute to the high density of rocky reef bottomfish there. Currently harbor seals do not appear to be targeting bottomfish, thus, it would seem premature to remove this site from consideration for protection. Although this association may change with increased bottomfish density, protection of the area and continued observations are necessary to evaluate this possibility.

#### **A larger view for marine reserves: *Community interactions & reserve size***

"An important principle of environmental science is that changes in single components of systems are likely to have consequences elsewhere in the same systems." - Pinnegar *et al.* 2000

Historically, fishing has exerted a directional pressure on fish stocks with targeted harvest of higher trophic level species (Pauly *et al.* 1998, Jackson *et al.* 2001, Myers and Worm 2003), and larger, older individuals (Beamish *et al.* 2006). This type of fishing has ecosystem-level effects (Goñi 1998, Agardy 2000, Blaber *et al.* 2000) as non-target species respond to the restructuring of communities and ecosystem processes (Jackson *et al.* 2001). Therefore, it is not unexpected that the numeric increase of exploited species may also have community- and ecosystem-level consequences.

Marine reserves are an extremely promising management tool as evidenced by the increase in biomass, abundance, and average size of exploited species within reserve

boundaries (Halpern and Warner 2002, Halpern 2003). In fact, reserves have been successful in increasing the size and abundance of rocky reef bottomfish in Puget Sound (Palsson 2001, Eisenhardt 2002). However, modeling exercises suggest that counterintuitive outcomes are possible when areas are protected without considering species interactions and related ecological processes (Salomon *et al.* 2002, Micheli *et al.* 2004, Baskett *et al.* 2006). The inclusion of trophic interactions in reserve design and management action scenarios is critical as protection can induce changes in community effects, such as trophic cascades (Pace *et al.* 1999, Pinnegar *et al.* 2000, Salomon *et al.* 2002).

Understanding predatory pressures and predator-prey interactions is essential to assess the influence of reserves on ecosystem processes as well as the efficacy of reserves in obtaining their aims (e.g., conservation and sustainable yield). However, these interactions are rarely straightforward, as they operate both directly and indirectly. For example, in the system studied here, seals may prey on rockfish directly; however, they also prey on rockfish predators such as other bottomfish (Hobson *et al.* 2001) and salmon (Beacham 1986), thereby indirectly benefiting rockfish. Moreover, seals and rockfish are also competitors that forage on similar prey such as Pacific herring and Pacific sand lance, especially in winter (Nagtegaal 1985, Murie 1995, Lance and Jeffries 2006). Thus, community models are needed to assess the direct and indirect effects of predator-prey interactions on reserve goals as well as to inform adaptive management decisions.

Predation pressures originating from outside reserve boundaries are to be expected, however, little work has been done in this arena (but see Boncoeur *et al.* 2002, Fanshawe *et al.* 2003). Marine reserves are often touted as “ecosystem management” in contrast to traditional fisheries management, which bases decisions on single-species statistics.

Nevertheless, many reserves originate because of concerns regarding target species and while it is true that by protecting habitat, reserves do shelter multiple species, organisms may not truly be protected at the ecosystem level. Considerations that apply to predator-prey interactions within reserves should also be applied to mobile predators that may visit protected areas. Additionally, marine ecosystems are quite complex and operate as open systems. Confounding effects of recruitment, pollution, disease, oceanographic events, illegal harvesting, and availability of shelters must be considered as additional potential causes of observed trends and should be experimentally tested (Sala *et al.* 1998). Lastly, to examine ecosystem effects of reserves, we must look at appropriate spatial and temporal scales, which are not normally addressed by short-term field projects. Longitudinal, multi-disciplinary efforts may provide more holistic information that should be considered in reserve design and management.

To improve fisheries sustainability, Botsford *et al.* (1997) proscribed a more holistic approach that considers strongly interacting species, marine habitats, and the physical environment. Yet top predators, which definitely “interact strongly” with their prey (Trites *et al.* 1997, Bax 1998), are not often considered in the design of fisheries-based reserves. Although some protected areas have been established for conservation of marine mammals and birds themselves (Hooker and Gerber 2004), the incorporation of top predators into reserves for fisheries management may make the objectives of those reserves more attainable. Trophic relationships, including predator-prey interactions, are extremely important in shaping communities and in processes of natural selection, and should be maintained to support healthy ecosystems (Soulé *et al.* 2003, Soulé *et al.* 2005, Boyd *et al.* 2006). Reserves that incorporate all ecosystem constituents and processes will necessarily be



larger protected areas (Walters 2000, Baskett *et al.* 2006), the establishment of which may meet some resistance.

The precedent for large protected areas and public support for that scale of conservation do exist. Large tracts of land in terrestrial ecosystems are protected, perhaps the largest in the United States being Wrangell-Saint Elias National Park and Preserve, Alaska covering over 3,600km<sup>2</sup> (WIN 2000). Unfortunately, protection of marine areas has lagged seriously behind terrestrial systems. In the United States, marine sanctuaries under federal jurisdiction were fewer in number, smaller in total area, and smaller in percentage of area covered than protected terrestrial areas (Lindholm and Barr 2001). This discrepancy between marine and terrestrial protection should be addressed at the federal level; however, community-supported, grassroots approaches are equally important if not more so (Agardy *et al.* 2003, Lundquist and Granek 2005, Kareiva 2006).

The candidate reserves in Skagit County are meant to contribute to an existing and developing network of protected areas in Puget Sound (McConnell and Dinnel 2002, Weispenning *et al.* 2004, Weispenning 2006). Because even small, local reserves lead to increases in the average size, biomass, abundance, and diversity of species within their borders (Halpern 2003), these reserves offer a near-term strategy for the potential increase of rocky reef bottomfish stocks. Yet, as indicated by this study, harbor seals and rocky reef bottomfish likely interact both directly and indirectly. Reserves that do not consider these types of predator-prey interactions may not fully meet reserve goals. Nevertheless, sites chosen as candidate reserves, especially Burrows Channel, present crucial opportunities to study the effects of marine reserves on target species, as well as the effects of protection on

community interactions, with the aim that reserves may be designed to truly protect local rocky reef bottomfish species at the ecosystem level.

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