

**THE DUNGENESS CRAB (METACARCINUS MAGISTER) FISHERY IN
BURRARD INLET, B.C.: CONSTRAINTS ON ABUNDANCE-BASED
MANAGEMENT AND IMPROVED ACCESS FOR RECREATIONAL
HARVESTERS**

by

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ABSTRACT

The British Columbia Dungeness crab (*Metacarcinus magister*) fishery is important to a diverse group of users, generating considerable value to coastal communities. While current management strategies have ensured sustainability and conservation of the species, persistently high exploitation by the commercial fishery limits access to the resource for First Nation and recreational crabbers. I evaluated the constraints on two possible management actions aimed at increasing access for recreational users. In chapter 1, I found that establishing abundance-based management using existing survey designs has potential for high use, multi-sector crab fisheries such as Burrard Inlet: provided that biases due to variable catchability are accounted for. In chapter 2, I demonstrated how discrepancies in requirements and responsibilities between the recreational and commercial sectors limit the scope of harvest rights attainable by the recreational sector. Reducing these discrepancies would help justify the changes to the management framework required to increase recreational access.

Keywords: Dungeness crab; multi-sector marine fisheries management; recreational access; abundance-based management; trap catchability; recreational sector legitimacy

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CHAPTER 1: UNCERTAINTY IN ESTIMATION OF DUNGENESS CRAB ABUNDANCE WITH TRAPS: THE ROLE OF SAMPLING DESIGN AND CATCHABILITY

1.1 Introduction

1.1.1 British Columbia Dungeness crab fishery

Crustacean fisheries have become an increasingly important resource, experiencing mounting fishing effort due in part to the global decline of many finfish stocks (Smith and Addison 2003). Of the invertebrate fisheries in British Columbia (BC), the Dungeness crab (*Metacarcinus magister*) fishery is the oldest and most important, exploited by commercial, First Nations and recreational harvesters (DFO 2009). In 2007, 222 commercial licenses landed an estimated \$37.8 million worth of Dungeness crabs, 11.6% of the total landed value of wild BC commercial fisheries (Oceans and Marine Fisheries Branch 2007). While some First Nation communities also harvest crabs for their commercial value, they additionally exploit this species for food, social and ceremonial purposes; managed by Fisheries and Oceans Canada (DFO) for community members under communal licenses. Additionally, over 300,000 tidal waters sport fishing licenses are sold annually in BC, of which crab fishing is thought to be a significant component. Because of the importance of this resource to a diverse group of users and its socioeconomic value to coastal communities, complex harvest strategies have been developed to meet conservation objectives and to accommodate the often conflicting objectives of the stakeholders.

1.1.2 *Current Dungeness crab management framework*

The commercial Dungeness crab fishery in British Columbia does not rely on estimates of abundance for management, but rather is managed using a so-called "3-S" strategy, which limits harvest by sex (male only), size (minimum 165 mm carapace width, measured from tip to tip of the longest lateral spines), and season (historically late-June to late-November). Such a strategy aims to maintain the reproductive potential of crab stocks by protecting all females and ensuring that sexually mature males are protected for at least one year prior to harvest (DFO 2009). Within the commercial fishery, limited licensing, area licensing, area closures, trap inventory limits (e.g., limit of 200 traps per commercial crabbing vessel) and restrictions on gear types and when traps can be hauled are also used to constrain fishing effort and mortality. The First Nations and recreational fisheries are subject to the same size restrictions as the commercial fleet, but these fisheries are open year-round, except for a few specific area closures. Recreational crabbers are additionally subject to male-only retention as well as have bag limits, possession limits and gear restrictions (DFO 2009).

Under this management strategy, Dungeness crab have been thought to be inherently resilient to recruitment overfishing (i.e. a rate of fishing that significantly reduces annual recruitment, or the entry of juveniles into the fishable size class). Cross their Pacific Coast range, catch rates have historically remained stable despite extremely high exploitation rates on legal sized males (Orensanz et al. 1998). This robustness has been attributed to the male-only fishery, size limit restrictions, and the relatively immobile and scattered populations, which act to create uneconomical refuge

populations, and to the ability of females to store sperm across reproductive seasons and thus skip poor reproductive opportunities (Jamieson, 1993; Orensanz et al. 1998; Swiney et al. 2003). However, small-scale serial stock depletions and collapses in Dungeness crab fisheries have been documented. These fishery collapses have occurred in part because of environmental effects such as climatic forcing and predation, but also due to overfishing resulting from increased fishing effort, expansion of fishing grounds and high incidental mortality of non-legal crabs (Orensanz et al. 1998). While isolated, these cases demonstrate that Dungeness crab populations are not as unaffected by fishing pressure as traditionally thought.

High exploitation creates a number of difficulties for managing these fisheries. In addition to risks of overfishing, equitable access to fishing opportunities among user groups has become an increasingly important issue. Commercial trappers are extremely effective harvesters, and are responsible for the majority of landings. In crab fisheries where exploitation rates of legal sized males can be as high as 90% (Zhang et al. 2002), this reduces opportunities for non-commercial crabbers. As a result, DFO is increasingly receiving requests to reallocate some of the stocks traditionally allocated to the commercial fishery to exclusive First Nations harvest, and to improve recreational user access to high quality crabbing opportunities (DFO 2009). Access to high quality crabbing opportunities is a particularly prevalent problem near urban centres, where the demand for access is high because of the high density of non-commercial trappers.

To address these conservation and access concerns DFO has recently undertaken an extensive consultation process to review the current Dungeness crab management

framework. The goal of this process is to address key issues of low catch rates, high discard levels, handling mortality and allocation among harvest groups (DFO 2009). To resolve these issues, a number of possible management actions have been proposed (DFO 2007). Most of these involve restrictions on commercial harvesters such as increasing area closures and gear restrictions, changing size limits, and strengthening license retirements. However, one of the proposed actions is to establish a total allowable catch (TAC) for Dungeness crab (DFO 2007), which would presumably be set by determining the abundance of available stock and setting aside a proportion of this for harvest. This strategy, generally referred to as abundance-based management, would represent a significant shift away from the current “3-S” management system towards active management based on abundance estimation, in-season stock assessments and harvest quotas.

1.1.3 *Estimating Dungeness crab abundance*

Critical to fisheries stock assessment and management is the ability to obtain an index of stock size that is proportional to abundance (Harley et al. 2001). Catch per unit effort (CPUE; biomass of crabs caught per unit of fishing effort) is one such index that is commonly used as a relative measure of stock abundance. The use of CPUE, however, requires catchability (i.e., the proportion of stock taken by one unit of fishing effort) to be constant across all fishing events if it is to be proportional to abundance. If this is not true, biased estimates of stock size result. In fact, recent studies of finfish have demonstrated that catchability is rarely constant, and varies across species, age classes

and between populations (Harley et al. 2001; Tsuboi and Endou 2008). Variable catchability is inherent to invertebrate trap fisheries as well because traps selectively catch target species depending on a variety of factors, including the fishing strategies employed (Miller 1990; Taggart et al. 2004), crab behaviour in and around the traps (Jury et al. 2001; Ihde et al. 2006; Barber and Cobb 2009), seasonality of the fishery (Tremblay 2000, Taggart et al. 2004) and gear saturation (Smith and Tremblay 2003). Such biases must be accounted for to accurately estimate abundance.

Depletion models are a common method for estimating abundance, particularly for fisheries where stock assessment data is scarce and exploitation is high (Smith and Addison 2003), as in the BC Dungeness crab fishery, and they have been shown to be particularly useful for crustacean fisheries (Dawe et al. 1993). However, the literature on depletion models repeatedly cautions that assuming constant catchability can be a serious error and it can lead to spurious estimates of absolute stock size or fishing mortality (Hilborn and Walters 1992; Smith and Addison 2003). Fishery CPUE will often stay high as abundance drops (hyperstability) or drop at a faster rate than abundance (hyperdepletion), biasing estimation of catchability and abundance. Both hyperstability and hyperdepletion indicate that catchability is varying between surveys and therefore the model has to be modified to account for this (Hilborn and Walters, 1992).

Determining whether hyperstability or hyperdepletion is occurring has typically been difficult because of the expense and effort required to obtain estimates of abundance that are independent of fishery CPUE. Methods for obtaining these estimates include fishery-independent surveys, swept-area trawl surveys and dive

transects. The latter methodology is an increasingly widespread approach to estimating density of populations (Tsuboi and Endou 2008), particularly invertebrates (Taggart et al. 2004; Tremblay et al. 2006). These studies usually involve surveying transects of known area and estimating the density (i.e. individuals per area) with SCUBA or snorkelling gear. Although less common, remotely operated vehicle (ROV) surveys provide a direct measure of absolute density (and therefore abundance), independent estimates of size distribution, and are not limited by depth, as are SCUBA diving transects. Comparison of absolute density estimates from ROV transects to trap CPUE is potentially useful method for detecting hyperstability and hyperdepletion, and therefore variable catchability.

1.1.3.1 Currently available data for assessing Dungeness crab abundance

In the lower mainland area of Vancouver, DFO regularly conducts fishery independent, fixed-station surveys (i.e. trapping locations repeated for all surveys) for Dungeness crab each spring and fall to collect biological data such as molt timing, population structure and injury rates in captured crabs (DFO 2009). The long duration (~20 years; A. Phillips pers. comm.) of these spring and fall surveys may provide a useful time-series for future abundance-based management, if catchability at these sites can be assumed constant. The fixed sampling site approach of the surveys may also significantly reduce bias in abundance estimates compared to a standard randomized survey design because the variance in catch rates can be calculated empirically by estimating covariance terms (Chen et al. 1998).

1.1.4 *Study area*

Burrard Inlet is a fjord largely located inland of Vancouver Harbour, BC, Canada (Figure 1.1). At its southern end, the inlet is almost entirely encompassed by the municipalities of Vancouver, Burnaby, Port Moody, and the Districts of North and West Vancouver. These relatively shallow and calm waters, close to urban centres are a popular destination for marine recreation. A secondary inlet, Indian Arm, stretches north of the main inlet and is characterized by steep mountainous shoreline and deep waters that have remained relatively undeveloped except for a few residential outposts, surrounded by the Indian Arm Provincial Park. The remainder of the Burrard Inlet is characterized by low topography and dense urbanization. The benthic environment throughout the inlet is mainly composed of un-vegetated, heavily silted, low relief muddy substrates, with steeply sloped rock walls near the shores, particularly along the north end of Indian Arm. Water depth ranges from 10m to 180m within the study area.

The fishery for Dungeness crab in Burrard Inlet is composed of a small commercial fleet of 2-3 vessels and a large, diverse group of recreational users drawn from the surrounding municipalities. The relatively small size of Burrard Inlet means that both sectors have access to the full range of crab habitat. However, the recreational fishing sector tends to fish the more easily accessible, shallow depths that do not require mechanized trap hauling equipment. Competition for resources between the commercial and recreational sectors is high, particularly around public boat launches and parks, where recreational crabbers tend to set their traps. This area

provides an opportunity to make a detailed assessment of a heavily exploited and highly competitive Dungeness crab fishery.

1.1.5 *Research Objectives*

The goal of this project is to investigate proportionality between absolute density and trap-based abundance indices for Dungeness crabs in Burrard Inlet, with the aim of providing information that will aid in determining the feasibility of abundance-based management for Dungeness crab. The study has two objectives:

- (1) To determine whether the mean trap CPUE from a fixed-station study design similar to the one implemented by DFO is significantly different from a randomized survey design, thereby evaluating whether fixed-station CPUE can provide an index of abundance for the entire inlet.
- (2) To evaluate whether mean trap CPUE at fixed stations is proportional to absolute abundance at these fixed sites as estimated from ROV transect surveys.

1.2 **Methods**

1.2.1 *Burrard Inlet Sampling Procedures*

Trap surveys targeting Dungeness crab were carried out in Burrard Inlet from May-October 2007 and from May-September 2008. Seasons were divided into five and three sampling periods (Appendix 1). Within the inlet, a study area was delineated to contain the fixed DFO survey locations and the majority of the current crab fishing effort (Figure 1.1). The sampling regime during each time period was divided among fixed

sites and randomly selected sites (Appendix 1). Six of the seven sites regularly sampled by the DFO survey were selected as our fixed sites. These sites were delineated based on the end points and the mean depth of the line of traps laid out at each site during the DFO survey in the fall 2006 (Figure 1.1).

Random site selection was treated differently in 2007 and 2008 because I was interested in minimizing the variance within a sampling site and at different depths. In 2007, random sites were selected by dividing the inlet into approximately 200m x 200m grid cells. Individual grid cells were randomly selected and multiple traps were set (4-5 traps) within the cell. The cell selection was stratified into 10-20m, 20-50m and 50-80m depth strata. A minimum of one cell in each depth strata was selected for each sampling period. In 2008, having found no significant differences in mean total catch caught within sites or between depth strata, individual traps were placed randomly throughout the study area during the random trap surveys.

Standard 34" diameter commercial grade Dungeness crab traps, fitted with wire mesh and baited with approximately 150g of thawed pacific herring, were fished overnight and soaked for 24 h. All escape hatches for undersized crabs were closed. Carapace width (CW), sex, shell hardness, and shell condition were recorded for all captured crabs. Carapace width was measured using the notch width format, by measuring the width across the carapace just anterior to the 10th anterio-lateral spine with slide callipers (Phillip and Zhang 2004). Using this format, legal sized crab are defined as ≥ 155 mm CW.

1.2.2 Objective 1: Comparison of fixed-station and randomized design

The trap data was analyzed to determine whether mean trap CPUE from a fixed-station study design is significantly different from a randomized survey design. Catch data from the fixed and random trap surveys were analyzed using generalized linear models (GLM) and mixture models. The number of crabs caught per trap (C_i) were modelled using a negative binomial model with probability $p(x_i)$ where $p(C_i | x, z) \sim$ negative binomial ($\lambda(z_i), \phi$). $\lambda(z_i)$ is the mean observed catch, and is linearly related to covariates using the log link function, expressed as a function of the explanatory variables z_i ($i = 1, 2, \dots, n$) (Martin et al. 2005). Estimation of the overdispersion parameter, ϕ , for the negative binomial model indicates the fit of the data to the model; in cases where $\phi = 0$, the model contracts to the Poisson. Poisson regression models were fit to the data, however the null deviance was greater than twice the degrees of freedom in both years, indicating overdispersion.

When trying to fit legal-sized catch per trap with the covariates, the proportion of zero catches was large, such that the data did not fit the standard Poisson or negative binomial distributions, invalidating the assumptions of the analysis and biasing the results (Lambert 1992). One approach for analyzing data with excess zeroes is to assume the response variable (i.e., catch per trap) follows a mixture distribution involving a Bernoulli process (i.e., generating either a positive or a zero count) and a count process (e.g., Poisson or negative binomial distribution). This class of statistical models is referred to as zero-inflated mixture models (Lambert 1992; Welsh et al. 1996; Martin et

al. 2005). Therefore, assuming that catches of legal-sized crab per trap (C_i) are independent, $p(C_i = 0 | x, z) \sim \text{Binomial}(\lambda(z_i))$ with probability $1 - p(x_i)$ and $p(C_i > 0 | x, z) \sim \text{Poisson}(\lambda(z_i))$ or negative binomial ($\lambda(z_i), \phi$) with probability $p(x_i)$. Just as $\lambda(z_i)$ is linearly related to covariates using the log link function for the standard negative binomial model (described above), the logit function is used to linearise the relationship between $p(x_i)$ and potential regressors for the Bernoulli model.

For both the 2007 and 2008 surveys, three explanatory variables were used to predict total catch of all captured size classes, and legal-sized catches (Table 1.1). A sampling period predictor was included to account for the depletion pattern of crabs as the fishing season progressed. Survey design (fixed or randomly selected) and depth stratum were used to assess whether there were significant differences in catch based on sampling design and fishing depth.

All parameters were estimated by maximum likelihood as implemented in R (R Development Core Team, 2008) using the `glm()`, `glm.nb()` and `zeroinfl()` functions in the `stats`, `MASS` and `pscl` packages, respectively. Model fits were compared with likelihood ratio tests.

1.2.3 *Objective 2: Evaluating proportionality of trap CPUE to absolute density from ROV transects*

ROV surveys of absolute density (expressed as crabs*100m²) in each site were undertaken to assess whether the relative indices of abundance measured by the 2007 trap survey CPUE at the DFO fixed sites were directly proportional. Sampling designs and

analysis methods for ROV surveys are summarized in Appendix 2. To maximize site contrast in terms of depth, two of the selected sites were at relatively shallow depths (Admiralty Pt. and Dan George); one was of intermediate depth (Bedwell Bay) and one was relatively deep (Deep Cove).

Our assessment of the legal-sized population of Dungeness crab from the ROV survey depended on accurate measurements of carapace width from ROV video. To assess the potential for bias in these measurements, the size distributions of crabs caught by trap at the fixed sites and measured during concurrent ROV surveys were compared. Differences in mean carapace width between the trap and ROV surveys were examined with t-tests, while the size distributions were compared with Kolmogorov-Smirnov tests. The size distribution of crabs caught at the fixed and randomly selected sites in the 2007 and 2008 season were also compared. Trap catches and ROV size estimates were aggregated into seasonal distributions.

The assumption of constant catchability was tested by modeling the legal-sized survey catch per trap ($CPUE_{trap}$) as a nonlinear function of legal-sized density from the ROV surveys ($CPUE_{ROV}$):

$$CPUE_{trap} = aCPUE_{ROV}^{b+1},$$

where b is a shape parameter of the function and a provides an estimate of catchability when $b=0$. Parameters a and b were estimated with functional (geometric mean) regression of the log transformed model (Ricker 1973; Peterman and Steer 1981; Hansen et al. 2000):

$$\ln CPUE_{trap} = b_0 + b_1 \ln CPUE_{ROV}$$

the intercept (b_0) = $\ln a$ and the slope (b_1) = $b+1$. When b_1 is equal to 1.0 catchability is constant and $CPUE_{trap}$ is proportional to abundance. For b_1 values significantly less than 1.0, hyperstability can be inferred whereas hyperdepletion is inferred for b_1 values significantly greater than 1.0.

1.3 Results

1.3.1 Comparison of fixed-station and randomized design

1.3.1.1 CPUE of all size classes

CPUE of all size classes remained fairly constant throughout the 2007 season (Figure 1.2a) and sampling period was not a significant predictor of catch ($p>0.05$). However, both sampling design and depth were significant predictors of total catch ($\chi^2=10.9$, $df = 2$, $p<0.05$; Table 1.2). The expected average catch declined at the random sites even though CPUE increased at deeper depth strata. For the 2008 survey, a negative binomial model revealed a similar pattern (Figure 1.2b) in which CPUE of all size classes remained constant throughout the field season. However, none of the explanatory variables were significant predictors of catch ($\chi^2= 4.32$, $df = 3$, $p>0.05$; Table 1.2). Overall, these results indicate that when all size classes are pooled, catch of Dungeness crab remains constant across the fishing season, although catch may vary with depth and sampling design.

1.3.1.2 CPUE of Legal Sized crabs

CPUE of legal sized crabs decreased across sampling periods in both years (Figures 1.2c and 1.2d), which, not surprisingly, indicates that the legal stock of crabs

was being depleted as the fishing season progressed. The potential for zero-inflation in the legal sized catch was observed in both years, indicated by a disproportionate number of traps having no legal sized crabs (Figure 1.3), particularly later in the season. A likelihood ratio test indicated that the ZIP model was a significant improvement over a standard Poisson fit for the 2007 survey (Vuong test= -2.97, $p < 0.05$). The ZIP model for crab catch indicated that sampling period was a significant predictor of declining catch ($\chi^2 = 102.00$, $df = 2$, $p < 0.05$; Table 1.3). Sampling design and depth were not significant predictors of catch. A zero-inflated negative binomial model (ZINB) fit yielded the same results, however the dispersion parameter (ϕ) was not significantly different from zero.

The 2008 data of legal sized catch was fit with a ZINB mixture model. The likelihood ratio test indicated that the zero inflated mixture model was a significant improvement over a standard negative binomial fit (Vuong test= -1.27, $p < 0.05$). The dispersion parameter was significantly different from zero (Table 1.3), suggesting that a ZIP model would be overdispersed. Average catch declined across sampling periods ($\chi^2 = 155.35$, $df = 6$, $p < 0.05$), however sampling design and depth were not significant predictors. None of the explanatory variables were significant predictors of excess zeros ($p > 0.05$) in 2008.

1.3.1.3 Size frequency

The mean size of crabs caught in traps at the fixed and random sites were different for both 2007 ($t = -2.364$; $df = 1130$; $p < 0.05$; Figure 1.4a) and 2008 field seasons ($t = -4.453$; $df = 1239$; $p < 0.05$; Figure 1.4b). The size distributions of crabs at the different

sampling design were also significantly different within each year (Kolmogorov-Smirnov test, $D=0.1$, $p<0.001$). The mean notch width of crabs caught at the fixed sites was 143mm and 141mm at the random sites. In 2008, mean notch width was 147mm at the fixed sites and 144mm at the random sites. The majority of crabs caught from both sampling designs had a carapace width range of 105mm to 175mm for both in 2007 and 2008. However, a greater percentage of legal sized crabs were caught at the fixed sites than at the random sites in both years; a difference of 5% in 2007 and 9% in 2008.

1.3.2 *ROV survey*

The size frequency of Dungeness crab estimated from the ROV transects was different from that estimated during the fixed-station trap survey (Figure 1.5). CW observations from the ROV survey ranged from 24mm to 189mm during the ROV survey, a much greater range than observed for the trap survey. The mean observed CW of 118mm was significantly smaller for the ROV survey ($t=-14.783$; $df= 442$; $p<0.05$), compared to 144mm for the trap survey and the distributions of CW observations from the ROV survey were significantly different (Kolmogorov-Smirnov test, $D=0.5$, $p<0.001$).

Only 13% of the observed Dungeness crabs in the ROV survey were legal-sized compared to 25% from the trap survey. CPUE of legal crabs observed during the ROV transects showed a sharp decrease throughout the 2007 season (Figure 1.6). Overall, ROV CPUE appears to have decreased sooner in the season than the trap CPUE at these sites.

1.3.3 *Trap catchability*

No relationship was observed between aggregated trap CPUE and the absolute abundance estimate from the ROV surveys, suggesting that catchability was not constant among sites, sampling periods and observed densities ($t= 1.34$; $df = 12$; $p>0.05$; Figure 1.7).

To test whether this was true for all localities, the slope coefficient (b_1) was estimated at each site (Table 1.4). Of the four sites, only Admiralty Point demonstrated a significant relationship between trap CPUE and abundance (Table 1.4). Additionally, the b_1 was not significantly different from 1, indicating that trap catchability was constant for this site. No significant relationship between trap CPUE and ROV density was observed at the remaining three sites. Trap CPUE tended to remain high at low ROV density in Bedwell Bay and no legal sized crabs were observed on the ROV transects at the Deep Cove site, despite high trap CPUE during some sampling periods. Interestingly, CPUE appeared to increase with abundance at the Dan George site and the b_1 estimate for this site was not significantly different from 1, However, the relationship was not significant despite a high correlation ($R^2 = 0.931$). A possible explanation for this lack of a significant relationship was that I was only able survey Dan George with the ROV three times, giving one degree of freedom for the regression. This likely resulted in low statistical power to detect a trend (i.e, a Type II statistical error). To evaluate this possibility, the Dan George estimates were pooled with the Admiralty Point site, which is in relatively close proximity to the Dan George site and has similar depth, substrate and tide regimes. A significant relationship was found between CPUE and abundance for

the pooled data and the slope coefficient was slightly greater than 1 (Table 1.4) This nonlinear relationship is an indication of hyperdepletion. However, the calculated p-value is only marginally significant ($p = 0.048$).

1.4 Discussion

Concerns over conservation and access have motivated an extensive re-evaluation of the management framework for Dungeness crabs in BC waters. A frequently considered alternative to the current “3-S” management strategy is to switch to a more active system of management, whereby a TAC is set, stock status and harvest is monitored, and fishing activity is adjusted periodically to accomplish some overall suite of fishery objectives (DFO 2009). Accurately estimating stock abundance is a critical component to such a management strategy (Walters and Martell 2004). I found that the fixed sampling site survey design currently used by DFO could also be used for stock assessment purposes. However, when I evaluated whether trap CPUE is proportional to the density of Dungeness crabs, I found conflicting evidence of proportionality between trap CPUE and abundance.

1.4.1 *Fixed station surveys*

While not specifically designed for stock assessment, the DFO surveys may have unintended utility for such use. While the long time series of data from these surveys clearly makes them an attractive option for stock assessment purposes, their fixed station characteristics, in particular, have a number of advantages. Previous work has shown that fixed station trap surveys for crab result in increased sampling efficiency,

precision, and reduced bias in abundance estimation when using change-in-ratio and index removal depletion methods (Chen et al. 1998). Our results corroborate the findings of Chen et al. (1998) by demonstrating that legal sized crab catches were not significantly different for the fixed stations compared to a randomized design.

Therefore, the legal sized catch observed at the fixed sites can be treated as an index of abundance for the entire study area. However, unlike 2008 surveys, in 2007 total catch was significantly lower at the random sites and significantly higher at deeper depths, which may be an indication of less than uniform distribution of crabs across the inlet. If this is the case, a fixed station design is particularly effective at reducing bias due to patchy distributions (Chen et al. 1998). Therefore, there is strong evidence that catch rates of legal crabs are unrelated to sampling design, suggesting that the DFO dataset may provide a good index of stock abundance in the spring and fall.

1.4.2 *Trap Catchability*

Using trap CPUE as a relative index of abundance has several advantages over other potential survey methods. Traps yield high catches for a relatively low cost and can sample a wide range of substrates with little damage to the benthic habitat or the catch (Corgos and Friere 2007). However, in using trap CPUE, it is critical to assess whether catchability is constant to ensure that CPUE is indeed a true index of abundance. A number of recent studies have evaluated bias in crustacean trap CPUE as an independent measure of absolute abundance (Dungeness crab, Taggart et al. 2004; American lobster, Tremblay et al. 2006; spider crab, Corgos and Friere 2007).

Collectively, these studies demonstrate that there is no clear relationship between trap

CPUE and abundance for crustaceans and that the relationship can vary between cohorts and within species. However, constant catchability is still assumed for Dungeness crab because large crabs are evenly distributed over their habitat rather than concentrated in high densities and fishers tend to spread their gear over entire habitats rather than concentrating gear in one area (Fong and Gillespie 2008).

Crabs were sampled with repeated, fixed station, fishery-independent trap surveys concurrent with a density estimate from ROV transects at the same sites. These are ideal conditions not generally found in fishery datasets. Under these conditions, a significant linear relationship was observed between trap CPUE and abundance at the Admiralty Point site. However, these relationships varied among sites. No detectable relationship was observed at three of the four sites when treated individually, and nonlinear relationships were observed when the two sites with increasing trends (i.e., Admiralty Point and Dan George) were pooled. Therefore, the assumption of proportionality between trap CPUE and abundance is not justified for our results.

1.4.3 *Limitations*

Similar to Taggart et al. (2004) and Corgos and Friere (2007), I believe low power in our dive transects hampered a definitive determination of the relationship between trap CPUE and abundance. No legal sized crabs were observed for a number of ROV sampling events during which legal crabs were caught in traps. This could be an indication of hyperstability, however it is more likely a reflection of the fishing power of our traps compared to the ROV. In looking for rare events, in this case a legal sized crab, the ROV did not have the same effective sampling area as a bait plume from a trap set

overnight. For example, during the second sampling period at Admiralty Point, each trap was attracting crabs from the area with an estimated CPUE of $1.0 \text{ crab} \cdot \text{trap}^{-1} \cdot \text{day}^{-1}$. If the crab density estimated from the ROV transects is true for that sampling period (0.12 crabs per 100m^2) each captured crab in each trap would have been drawn from a mean area of 833m^2 . The total area covered by the ROV during that sampling event was 979m^2 . Therefore, in setting multiple traps at a site, crabs were sampled from a much greater area than the ROV was able to achieve.

Measurement error may also have biased our results. ROV estimates of crab size, and therefore estimates of legal sized crab abundance, were heavily dependent on visual estimates of crab size during the post-transect video viewing. A greater range in size distribution was observed for the ROV compared to the traps (Figure 1.7), indicating that estimates of crab size may be biased slightly higher than the estimates for the traps and that the ROV observes more small crabs than are caught in the trap.

1.4.4 *Management Implications*

Despite the simplicity of use and relative ease of data collection with traps, accurate estimation of abundance is contingent on critical assumptions about catchability. If catchability varies systematically over sampling periods, as appears to be the case at the majority of our sites, use of simple depletion models such as the Leslie model (Leslie and Davis 1939) will produce biased estimates of abundance. More robust models for estimating abundance, that are robust to variable catchability, are required if abundance-based management is the goal for alleviating competition in these competitive, multi-sector fisheries.

For example, the change-in-ratio (CIR) method could be used for abundance estimation in Burrard Inlet. The model estimates initial abundance by observing the relative change in population composition of two distinct classes (e.g. male and female; legal-sized and sub-legal sized crab) following a known removal (Chen et al 1998; Smith and Addison 2003; Fong and Gillespie 2008). This class of models do not require constant catchability if only one class of is harvested (Chen et al 1998); therefore, they may be appropriate for estimating legal-sized Dungeness crab abundance with traps. Replicate CPUE samples have also been recognized as important for variance estimation within the classes (Eberhardt 1982; Dawe et al. 1993), because factors influencing abundance estimation (e.g. migration and mortality) can be assumed to apply equally to the two classes (Chen et al. 1998). CIR model precision is improved with fishery-independent data and by incorporating information about the variation of encounter probabilities among subclasses (Udevitz and Pollock 1991) and sampling effort (Udevitz and Pollock 1995). Therefore, the existence of a long term DFO data set with pre- and post-season, fixed station surveys makes CIR a strong candidate.

If the data is available, maximum likelihood estimation is also commonly used for depletion analysis involving complete model specification in which catchability is treated as a variable rather than a constant (Schnute 1983). However, these models require fairly extensive datasets for parameter estimation. More recently, hierarchical bayesian modelling has been shown to successfully estimate abundance with variable catchability and smaller datasets (Zhou et al. 2008).

In conclusion, abundance-based management has potential application in Burrard Inlet. Available DFO datasets are well suited to estimating abundance, despite not having been designed for such use. Given the importance of accurately estimating abundance of legal sized crabs for such a management strategy, addressing the bias arising from variable catchability, often a central assumption of depletion models, is central to success. I found no evidence of constant catchability, however these issues may be overcome using simple estimation methods such as the change-in-ratio method.

1.5 Tables

Table 1.1: Predictors used in the analysis of Dungeness crab catch per trap

Predictor	Type	Description
Sampling Period	Categorical	Discrete time period of sampling event
Sampling Design	Categorical	Fixed or Random Selection
Depth Stratum	Categorical	Traps set in shallow, intermediate or deep depth stratum

Table 1.2: Coefficient estimates, standard errors and the calculated probabilities for the negative binomial GLMs predicting total catch of all size classes of Dungeness crab catch in 2007 and 2008.

	2007			2008		
	Estimate	Std. Error	Prob.	Estimate	Std. Error	Prob.
(Intercept)	1.396	0.125	<0.001	1.663	0.131	<0.001
Sampling period	-0.053	0.028	0.056	-0.072	0.043	0.096
Sampling design	-0.238	0.087	0.006	-0.071	0.078	0.359
Depth stratum	0.120	0.051	0.018	0.028	0.052	0.598

Table 1.3: Coefficient estimates, standard errors and the calculated probabilities for the zero inflated Poisson (ZIP) and zero inflated negative binomial (ZINB) mixture models predicting legal sized Dungeness crab catch in 2007 and 2008. Estimates for the overdispersion parameter, ϕ , are also provided for the ZINB model.

	2007 (ZIP)			2008 (ZINB)		
	Estimate	Std. Error	Prob.	Estimate	Std. Error	Prob.
(Intercept)	0.871	0.276	0.002	1.690	0.423	<0.001
Sampling period	-0.422	0.103	<0.001	-0.748	0.200	<0.001
Sampling design	-0.209	0.202	0.300	-0.279	0.251	0.266
Depth stratum	0.188	0.096	0.051	0.004	0.132	0.976
Log ϕ	-	-	-	1.331	0.463	0.004

Table 1.4: Estimates of the slope parameter (b_1) for the goodness of fit test ($H_0: b_1=0$) and the test for constant catchability ($H_0: b_1=1$) for crabs caught by trap at fixed trapping sites and observed during concurrent ROV transects in Burrard Inlet, 2007. Statistics presented are the estimate of the slope coefficient, standard error and the calculated probability.

	b_1 Estimate	Std. Error	$H_0: b_1=0$	$H_0: b_1=1$
			Prob.	Prob.
Admiralty Pt.	1.123	0.054	0.002	0.150
Bedwell Bay	-0.036	0.115	0.782	-
Dan George	0.716	0.196	0.170	-
Deep Cove	-	-	-	-
Admiralty Pt. + Dan George	1.106	0.040	<0.001	0.048

1.6 Figures

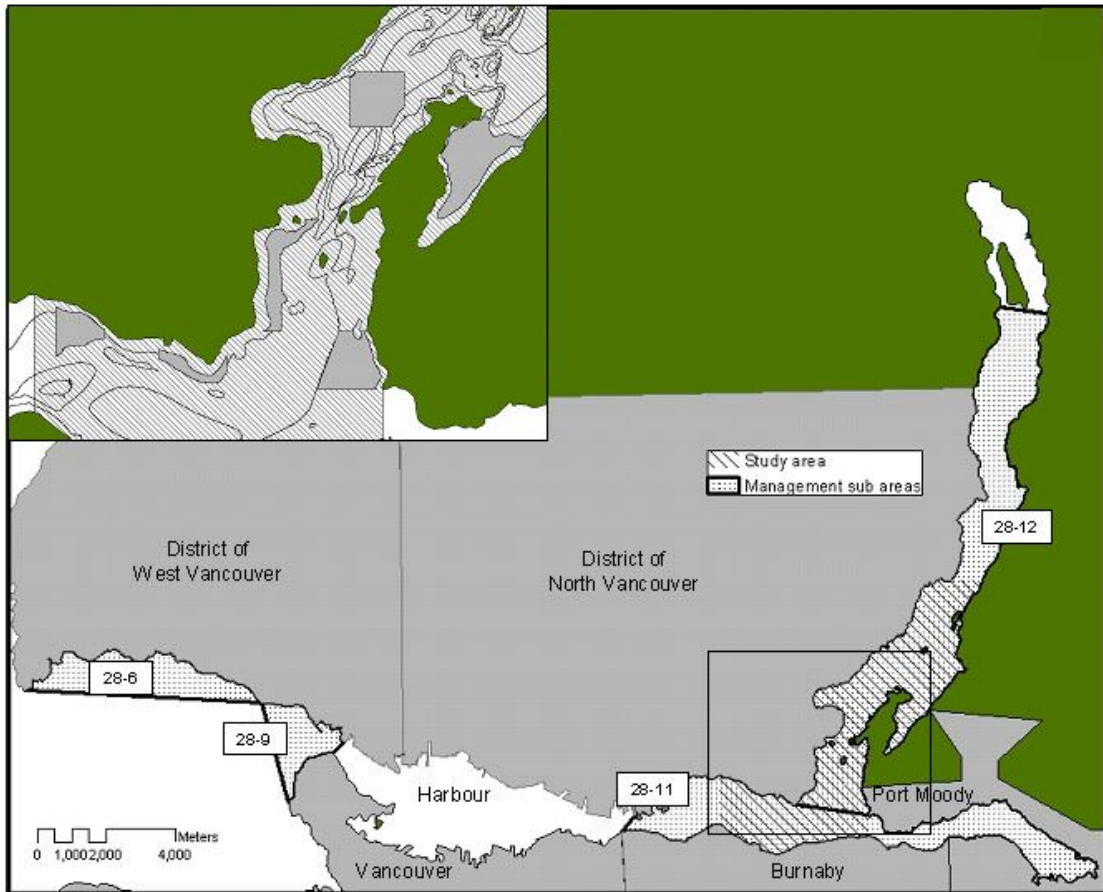


Figure 1.1: Location of the study area and management sub areas (numbers in boxes) in Burrard Inlet, Canada. Inset shows the fixed sites (solid grey areas) within the study area. 20-meter bathymetric contour lines are also shown in the inset.

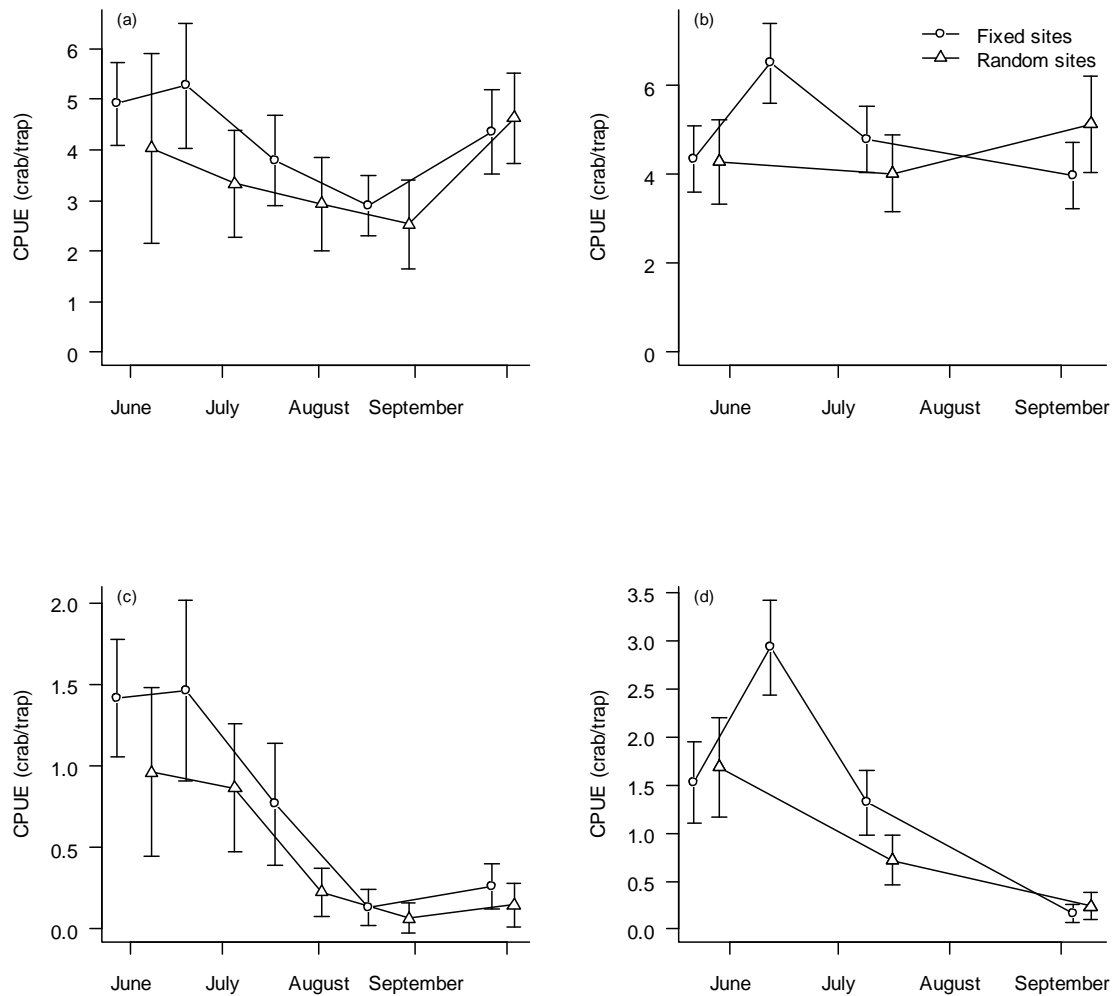


Figure 1.2: Mean CPUE (number of crabs per trap) by sampling period, with 95% confidence intervals at fixed and random trapping sites in Burrard Inlet, 2007 and 2008. Panels (a) and (b) show total catch, panels (c) and (d) show legal sized catch (≥ 155 mm CW).

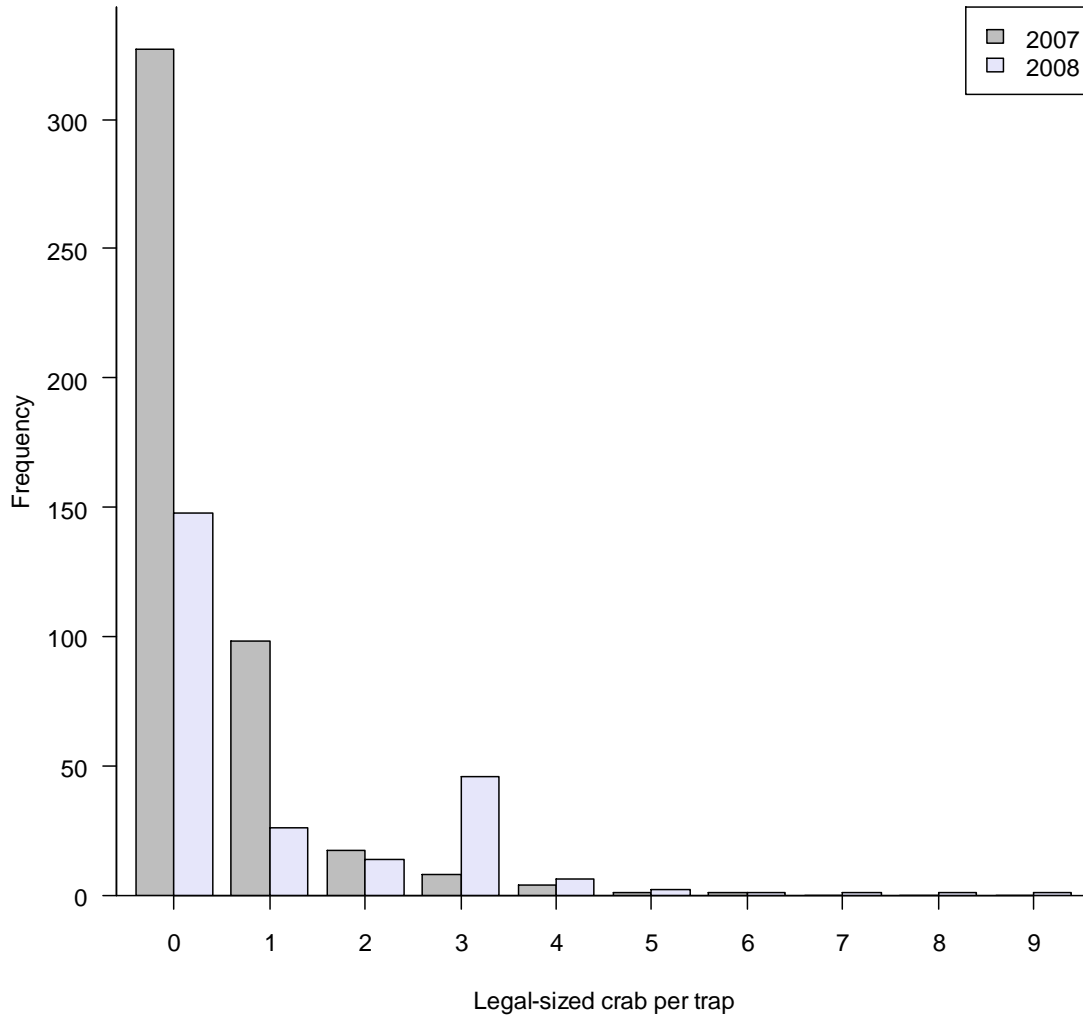


Figure 1.3: Observed number of legal-sized ($\geq 155\text{mm}$ CW) crabs per trap for aggregated catches at fixed and random sites in Burrard Inlet, 2007 and 2008.

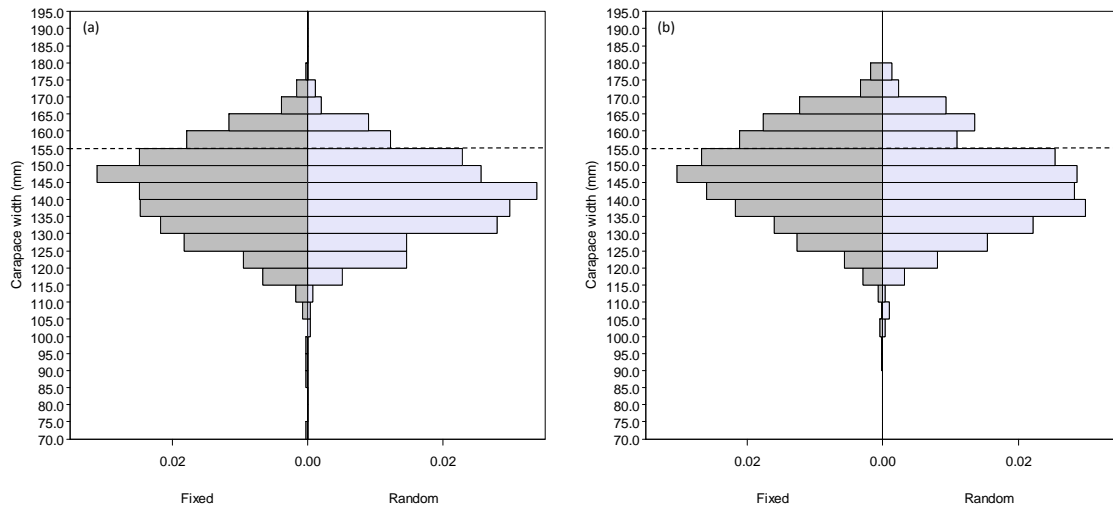


Figure 1.4: Comparison of crab carapace widths captured at fixed and random trapping sites during the (a) 2007 and (b) 2008 surveys. The dashed line represents the legal size division. Size distributions were significantly different between sampling designs for both years.

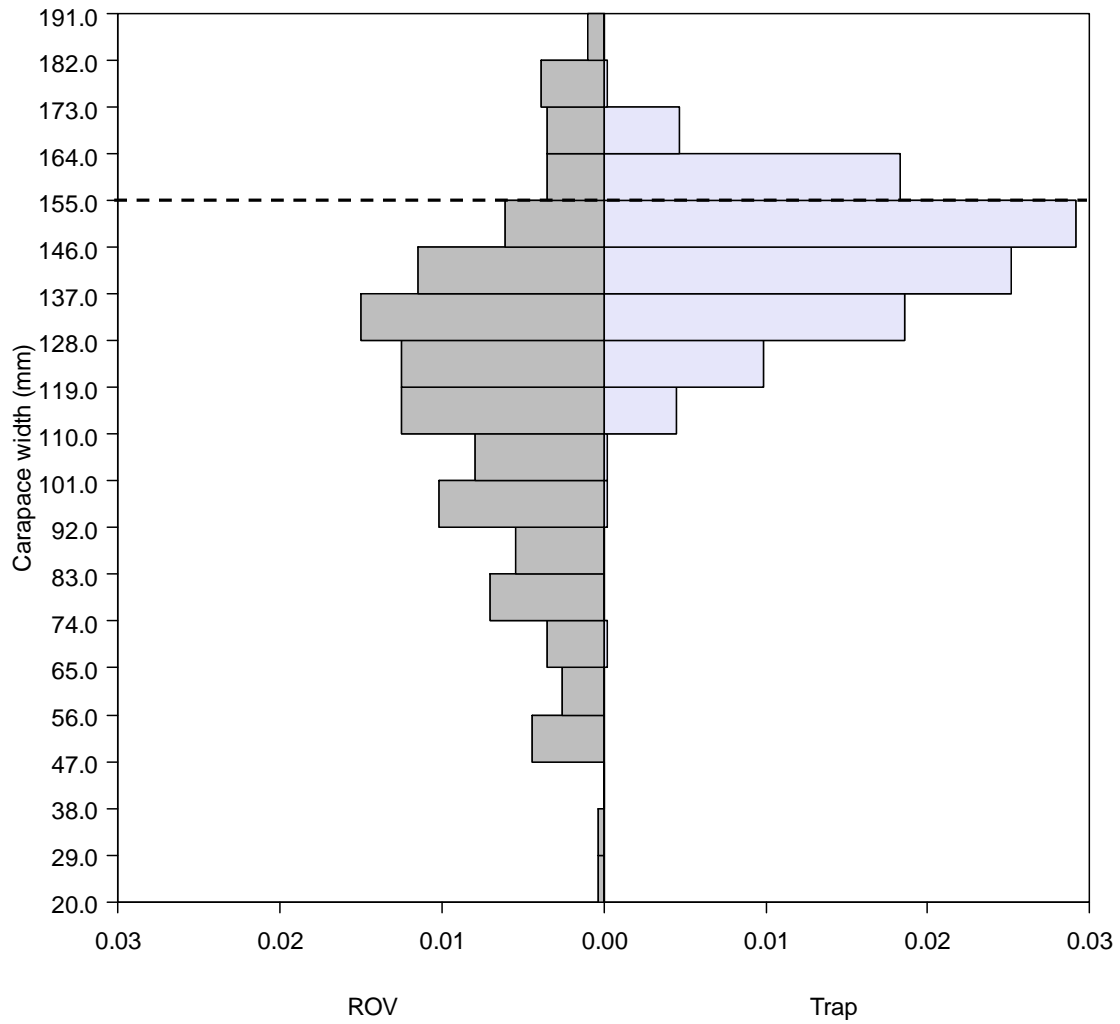


Figure 1.5: Comparison of estimated Dungeness crab carapace widths observed during the 2007 ROV transects and captured concurrently by trap in 2007. The dashed line represents the legal size division.

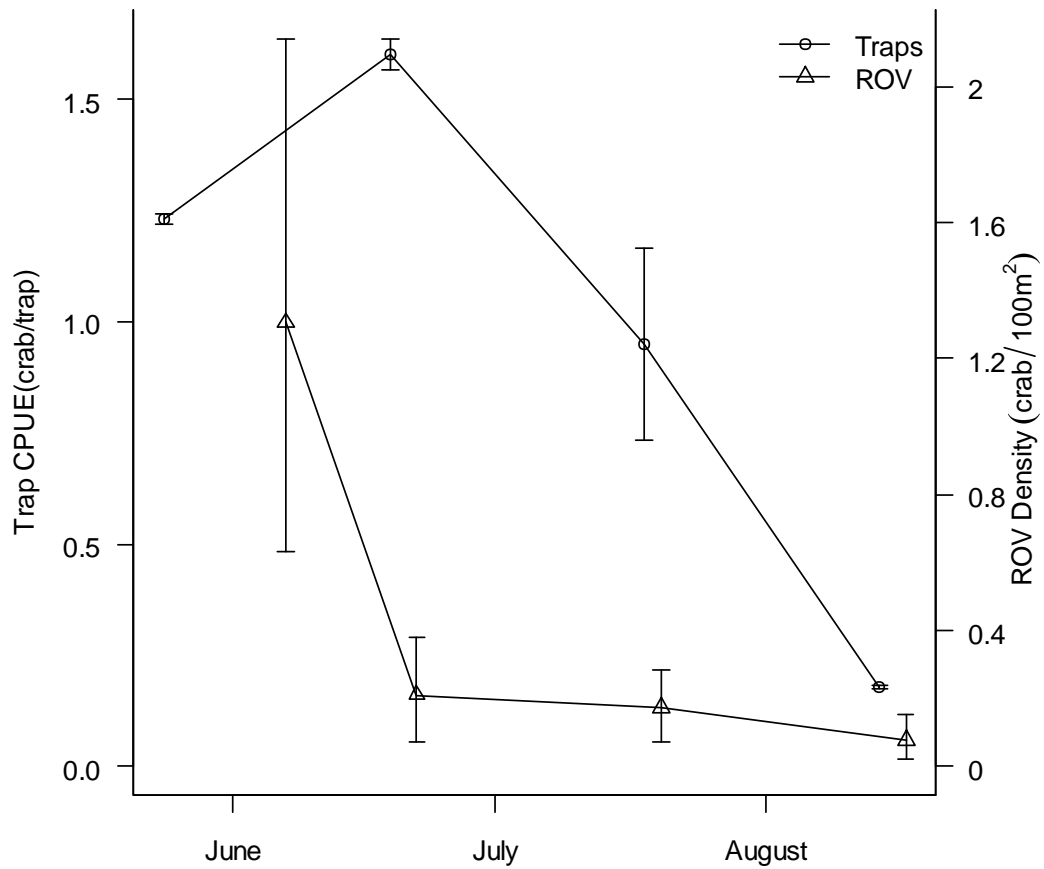


Figure 1.6: Mean trap CPUE and mean ROV density by sampling period, with 95% confidence intervals, of legal sized catch (≥ 155 mm CW) for the concurrent surveys at fixed trapping sites in Burrard Inlet, 2007.

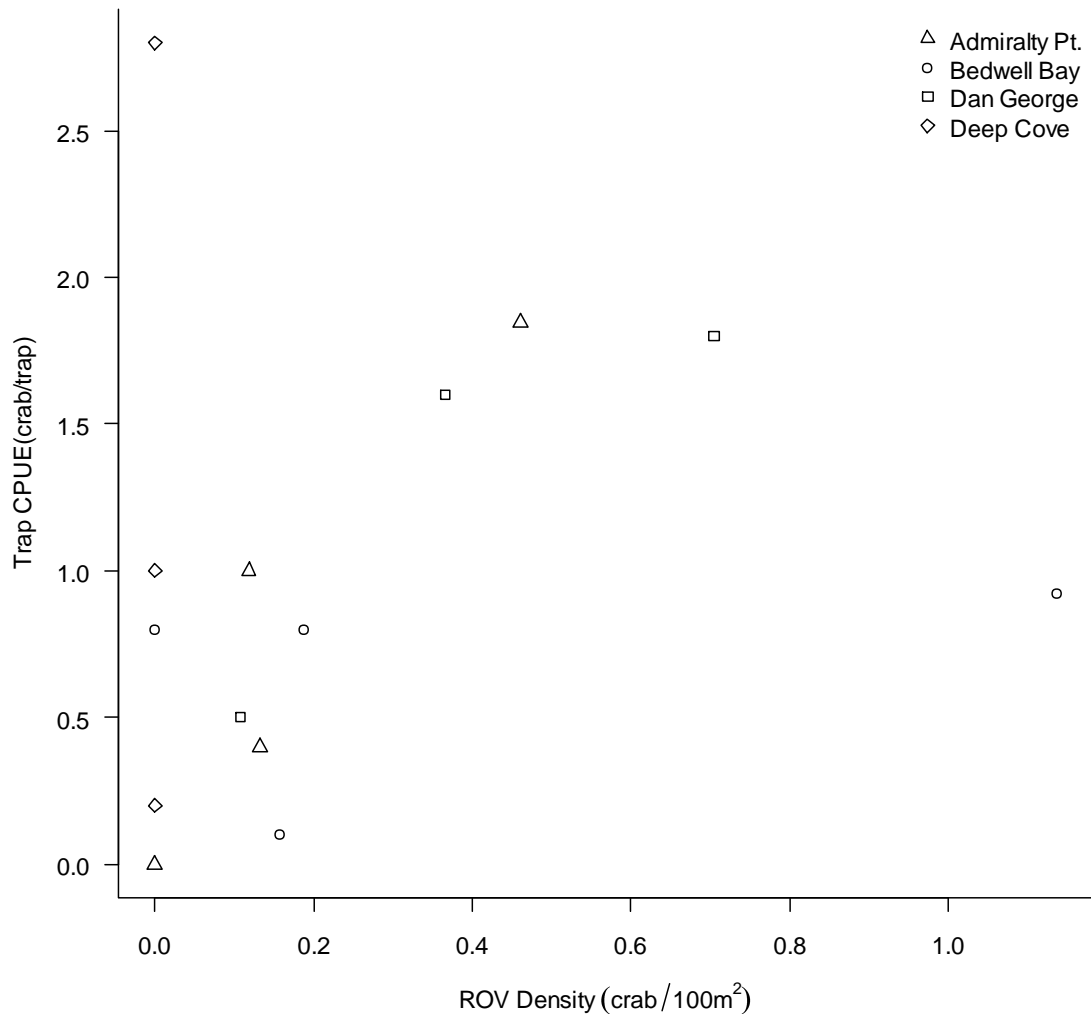


Figure 1.7: Relationship between mean trap CPUE and mean ROV density for the aggregated, concurrent surveys of legal sized Dungeness crab ($\geq 155\text{mm CW}$) in Burrard Inlet, 2007.

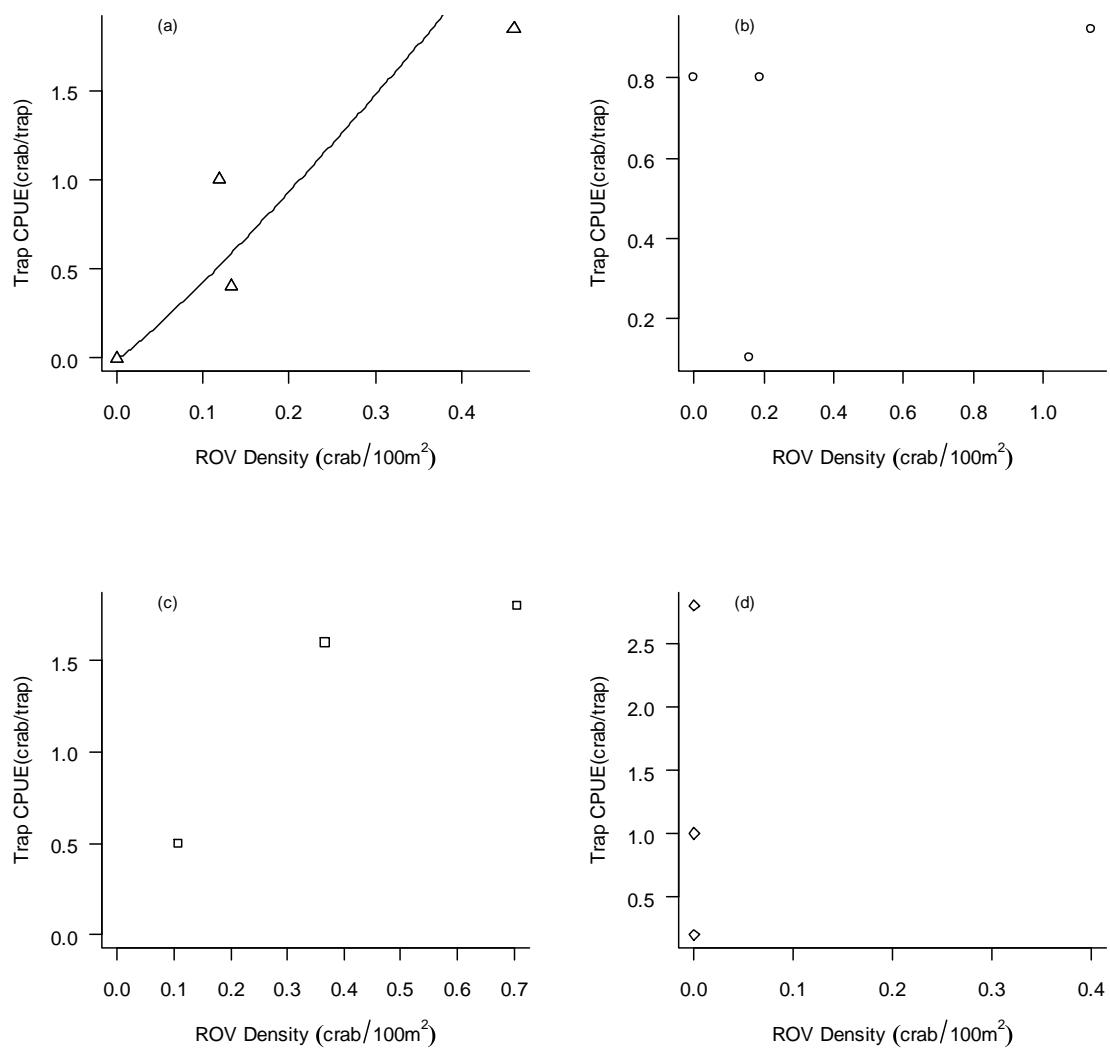


Figure 1.8: Relationship between mean trap CPUE and mean ROV density for the concurrent surveys of legal sized Dungeness crab ($\geq 155\text{mm}$ CW) at sites (a) Admiralty Point, (b) Bedwell Bay, (c) Dan George, and (d) Deep Cove in Burrard Inlet, 2007. The line of best fit is presented where a significant relationship was observed.

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CHAPTER 2: BUILDING THE LEGITIMACY OF THE RECREATIONAL FISHING SECTOR IN MIXED COMMERCIAL-RECREATIONAL FISHERIES

2.1 Recreational fishing within fisheries resource management

Marine fisheries management is rapidly diversifying to include a wider range of stakeholders and interests, including recreational fisheries. While recreational fisheries have a long history on the Pacific Coast, and have developed with commercial interests, many have increased in magnitude of both fishing effort and catch, and are thus encroaching on what were once considered commercial fishery resources (Coleman et al 2004).

In fisheries management, it is expected that stakeholders who support the rules and practices of sustainability should generally be rewarded with secure allocation rights. Recreational fishers have not been required to contribute to sustainable fisheries management (e.g. catch reporting, cost recovery) to the same degree as the commercial sector, but they also have not received many of the same benefits in terms of harvest rights. Management objectives for each sector are also often quite different as well (Walters and Cox 1999; Goodyear 2007). Such an apparent mismatch in harvest rights, management responsibilities, and objectives for these different stakeholders leads to misunderstanding and conflict between competing user groups (Goodyear 2007; Mitchell et al. 2008).

A common task for fisheries resource managers is to identify stakeholders and determine their salience, or the degree to which managers decide to give priority to their competing claims (Mitchell et al 1997). Salience depends on a combination of

characteristics that define the stakeholder. For instance, one key attribute of a stakeholder is the *legitimacy* of their relationship in the management process. Legitimacy in this context is defined as “a generalized perception or assumption that the actions of an entity are desirable, proper, or appropriate within some socially constructed system of norms, values, beliefs, definitions” (Mitchell et al 1997). Other defining attributes are the *power* of the stakeholder’s influence and the *urgency* of their claims. A stakeholder is defined as an individual or association having any combination of these three attributes (see Mitchell et al. 1997 for a broader description). Because of the diffuse nature of recreational fisheries, which typically consists of a diverse collection of individuals lacking in a cohesive vision, their power of influence in the management process is generally low. Similarly, due to the low economic impact compared to commercial fisheries, the urgency of recreational claims is also perceived as moderately low. Therefore, recreational fisheries frequently rely on the legitimacy of their relationship in the management process as the defining characteristic of their stake.

In this paper, I demonstrate how discrepancies in requirements and responsibilities between the recreational and commercial sectors generally contribute to a series of legitimacy gaps that limit the scope of harvest rights attainable by the recreational sector. I argue that by closing these gaps recreational fishers could improve their harvest rights and increase their recognition as stakeholders within the fisheries management processes. First and foremost, the recreational fishery sector must ensure that individual or collective groups of fishers meet acceptable standards of catch

reporting, assessment, and limits on total fishery catch if they are to become fully legitimate stakeholder in the resource allocation process. I critically examine the worldwide experience in recreational fisheries management to suggest mechanisms that will allow these legitimacy gaps to be plugged, and I apply these lessons to a competitive mixed-use fishery for Dungeness crab in Burrard Inlet, British Columbia.

2.2 Sustainable fisheries management

To become equitable stakeholders alongside government and commercial interests, recreational fisheries must operate in accordance with the rules, principles, or standards established for sustainable fisheries. These rules generally fall within the following fisheries management processes:

- (i) monitoring and assessment of stock status relative to targets and reference points,
- (ii) control of either total fishing effort or total harvest, and
- (iii) fair allocation of the harvest amongst stakeholders.

The degree to which these processes are successfully managed within the recreational fishery should determine the level of access to a particular fishery resource. Below, I give a detailed description of these 3 management processes, and briefly describe their general level of implementation in recreational fisheries.

2.2.1 *Monitoring and Assessment*

Knowledge of the state of a fishery, both in terms of fish stock status and supporting ecosystems, is crucial for sustainable fisheries management. Information on landings is commonly considered the minimum data required for managing a fishery (Vasconcellos and Cochrane 2005). However, lack of basic catch and effort reporting often impedes assessments of stock status and fisheries policy analysis. Where this information is available, statistics such as catch, fishing effort, as well as size- and age-composition of the catch can be compiled and interpreted through fisheries stock assessment models to give an indication of how stocks have responded to fishing impacts in the past. This provides some guidance on how to set harvest rates in the future so as to maintain a sustainable stock size and harvest.

Commercial fisheries worldwide have well-established reporting standards and procedures (Hilborn et al. 2005), but these protocols continue to be absent in most recreational fisheries. For example, self-reporting in a log book is common in commercial fisheries and may be verified by on-board monitoring with cameras or human observers, despite the fact that this level of monitoring creates significant challenges for the fishery. For individual self-reporting, (i) total landed catch needs to be verified via a dockside monitoring program; (ii) the fisher logbooks need to be digitized and compared to landed catch from the dockside report; (iii) at-sea discarding reports have to be monitored with at-sea observers or electronic systems. In the latter case, (iv) the many hours of video have to be audited to ensure that reported activity matches actual activity to within some reasonable tolerance. Therefore, fully established

monitoring programs involve large costs, that are typically funded by the harvesters. For example, in 2007 Pacific Halibut (*Hippoglossus stenolepis*) vessels paid an average of \$62,000 each for catch monitoring and licensing fees (DFO 2010a).

Self-reporting and compliance monitoring are generally absent for marine recreational fisheries, it is therefore not uncommon for catch data to be entirely absent in recreational fisheries. Where catch data does exist for recreational fisheries, it is generally collected through creel surveys that are funded from revenue generated through license sales. Design and frequency of surveys to estimate recreational catches and effort have improved, however this information can be costly to acquire for recreational fishing management agencies. Given the difference in structure and scale between sectors, recreational fisheries monitoring will likely have to continue with low cost and low precision solutions.

2.2.2 *Control of harvest*

Control over total harvest in recreational fisheries is increasingly important, especially given the growth in the recreational fleet. For example, marine recreational fishing in the United States increased by over 20% between 1996 and 2000 (Sutinen and Johnston 2003). While only 4% of total landings of finfish were attributable to recreational harvest in 2002, when focussing on populations of concern recreational fisheries accounted for 23% of all US landings, and in the Gulf of Mexico were as high as 64% (Coleman et al. 2004).

Limiting the total harvest of recreational fisheries is typically attempted using "input control" measures such as limits on fishing effort, season length, or area management, but can also be limited with "output controls" in the form of catch and possession limits. There are fundamental difficulties involved in establishing recreational fisher acceptance and compliance with both input and output controls. For example, although common approaches such as bag- and size-limits are accepted by the recreational community, they are often viewed by recreational managers as inefficient for controlling total harvest because these measures do not restrict the total fishing effort (Radomski et al. 2001; Lewin et al. 2006; Cox et al. 2003). Because open-access (i.e., no restrictions are placed on who can fish or how many anglers can access a particular resource) is a central tenet of North American recreational fishing, attempts to limit fishing effort through restrictive input controls usually face sustained opposition from the recreational community (Cox and Walters, 1999). Similarly, in the rare cases where output controls have been implemented for recreational fisheries, control has not been effective. For example, although total allowable catch limits have long been established for each sector in the mixed-use red snapper fisheries in the Gulf of Mexico recreational harvest often exceeded both commercial harvest and the recreational TAC throughout the 1990s (Sutinen and Johnston, 2003). Control of harvest therefore remains a fundamental problem for many recreational fisheries.

2.2.3 Allocation of harvest

The principles of resource allocation that are being applied to commercial fisheries worldwide are increasingly being applied to recreational fisheries. Harvest allocation for recreational fishers can be categorized into (as described for commercial fisheries in Hilborn et al. 2005) open, limited entry, individual quotas, and exclusive use access structures. Most of these have examples in management of marine recreational fisheries; however, open access is overwhelmingly the standard structure in recreational fisheries. Canada has a legacy of open access policies for recreational fisheries. The guiding principle behind this policy is that common property, including fisheries, require egalitarian access rights across fishing sectors.

Limited entry, in which a fixed number of licenses are issued (Hilborn et al. 2005), is often cited as a potential strategy for sustainable recreational fisheries management (Walters and Cox 1999; Lester et al. 2003; Cox et al. 2003). However, because this restrictive form of access is aimed at restricting fishing effort directly, it is rarely implemented anywhere in the world (Johnston et al. 2007) because it runs counter to the most common objection, which is to increase fishing effort. One notable exception is the limited entry fishery for pink snapper (*Pagrus auratus*) in Shark Bay, Australia (Mitchell et al. 2008), where fishing is only permitted with the purchase of a harvest tag. Modeled after limited entry hunting strategies, one tag permits a fisher to fish in Shark Bay and harvest a single snapper. This strategy, combined with seasonal spawning closure, has proven highly effective at controlling recreational pink snapper catches compared to conventional open access strategies (i.e. bag limits and size limits)

employed for pink snapper in other areas (Mitchell et al. 2008). However, the success of the novel tagging programs in Shark Bay is attributed largely to pink snapper ecology, involving a constrained spatial range and low release mortality. Applicability of this limited entry strategy to other species remains untested.

Individual quotas are an even more exclusive access structure, in which the license not only allows access, but also entitles the licensee (e.g. an individual or a vessel) to a proportion of either the total catch or total fishing effort (Parsons 1993, Hilborn et al. 2005). This quota entitlement is a means for economic rationalization of fisheries (Parsons 1993) and, when marketable and transferable between licensees, promotes flexibility in the fishery (Scott 1979). Individual quota strategies have not been implemented in recreational fisheries, but have been considered for a number of prominent mixed-use fisheries, particularly where recreational fishing is mainly comprised of commercial guiding and resort operations (Sutinen and Johnston 2003; Abbot et al. 2009). However, fundamental differences between commercial and non-commercial recreational fisheries make this strategy complicated in a recreational context. Commercial fisheries are characterized by high catchability, low effort, and fewer fishers, whereas recreational fisheries have low catchability, high effort, and many fishers. The costs and complexity of adequately monitoring individual quotas within such a large and diverse group of fishers does not appear to be feasible at present.

Finally, exclusive use rights can be applied in recreational fisheries management by closing certain fishing areas to the commercial sector. Exclusive use is the most

restrictive means of allocating harvest among sectors, and it is an increasingly common management strategy for mixed-use fisheries. For instance, in 2007, striped bass (*Morone saxatilis*) and red drum (*Sciaenops ocellatus*) were designated by US Executive Order 13449 as recreation-only species in all United States federal waters. More commonly, however, exclusive use rights are enacted on small spatial scales. For example, exclusive recreational harvest rights have been granted in Puget Sound, WA and Ucluelet Harbour, BC. In both of these cases, the benefits to the recreational Dungeness crab (*Metacarcinus magister*) fishers are high because of dense human populations and ease of access. Exclusive-use access structures are more feasible allocation strategies for the recreational sector than limited entry and individual quota strategies, and are therefore increasingly applied where an alternative to open-access is necessary.

Conflicts inevitably arise wherever there are overlaps in fishery resource use between commercial and recreational sectors (Jamieson 1993; Kearney 2002; Cooke and Cowx 2006; Goodyear 2007; Mitchell et al. 2008). Part of the reason for this conflict may be that access management for recreational fisheries generally follows the extremes of either open-access or exclusive use. Weak control over recreational harvest under open-access is often viewed as inequitable by commercial harvesters who are typically held to tight catch limits and compliance monitoring protocols (e.g. Mitchell and Baba 2006). At the other extreme, providing the recreational fleet with exclusive use of some fishing areas excludes commercial fishers. This increases their cost of fishing because the fleet must travel to new areas and develop new technologies to fish

these areas (Salas and Gaertner 2004; Branch et al. 2006) and increases competition and over-crowding of areas that remain open to commercial fishing. Although individual quotas used to allocate commercial harvest have exclusivity properties, such quotas cannot exclude the entire recreational sector from accessing the resource in the way that exclusive recreational fishing zones exclude the commercial sector (e.g., Florida inshore net ban, Atlantic striped bass). The first step toward alleviating these intra-sector conflicts is to ensure that recreational sector management processes are within the norms and standards established for commercial fisheries. Although this is a lofty goal, establishing legitimacy of the recreational sector would substantially improve the perception of equitability in the sharing of mixed-use resources. While, in part, this will be achieved by altering the regulatory structure of recreational fisheries, incentive structures (as described below) that encourage both rule compliance and an ethic of resource conservation will likely be necessary to creating recreational legitimacy.

2.3 Incentives for sustainable fishery management

Development of commercial fisheries management initially involved similar access structures to today's recreational fisheries. Much like recreational management, the two dominant types of commercial fishing controls in the late 1800s were gear restrictions and seasonal closures. However, by the mid-1900s it became evident that these approaches were not enough to prevent over-fishing as commercial fleets grew larger and more efficient. As governments unsuccessfully imposed limited entry in an effort to limit high fishing mortality output controls, mechanisms to target the causes of

over-exploitation for common property rather than the effects, were instituted (Parsons 1993).

Strong incentive structures for sustainable biological and economic fisheries management have been built into commercial fisheries management that are generally absent from or not applicable to recreational fishing. Such incentives fall into two main classes: direct, rights-based incentives and indirect, market-based incentives. Direct incentives associated with harvesting rights, such as individual quotas, are established as a means for economic rationalization of fisheries by eliminating overcapitalization in the fishing fleet (Moloney and Pearse 1979) and the incentive to “race-to-fish” (Parsons 1993; Hilborn 2005). Rights-based incentives create long-term asset value by appealing to commercial fishers desire to be economically efficient. When fishers are allocated harvest rights, improvements in income come not by catching more fish, but rather by reducing costs and increasing productivity and quality of the product (Hilborn, 2005). This tends to motivate the commercial fishing industry to contribute directly to all three fishery management functions. A good example of this is the BC groundfish trawl fishery, where the management functions are facilitated by an at-sea observer program (Branch and Hilborn 2008). In this fishery, all fishing activity is monitored by an on-board fishery observer, mostly funded by the individual vessel (Branch and Hilborn 2008). Despite the costs to the fishers, this oversight combined with an individual quota system adds value to the fishery by reducing inefficiencies (e.g. bycatch reduction) and reducing opposition from groups opposed to the bottom-trawl fishery by providing third-party auditing of fishing activities.

Indirect, market-based incentives are increasingly used to promote sustainability of capture fisheries. For example, non-governmental organizations such as the Marine Stewardship Council (MSC) attempt to influence seafood consumer choices by "...[using an] eco-label and fishery certification program to contribute to the health of the world's oceans by recognizing and rewarding sustainable fishing practices" (MSC 2010). Sustainable fishing practices are evaluated by MSC with voluntary standardized assessments of the entire fishery. The reward for participants is access to premium seafood markets all over the world. Although market-based incentives are likely to produce ecological benefits in the long-term, fisheries risk the immediate costs of losing markets if they do not obtain the MSC label. Thus, market-based incentives benefit long-term asset value as well as short-term value of the catch.

In contrast to the incentives built into commercial fisheries management described above, aside from personal conservation ethics, there are few incentives for recreational fishers to contribute to management. Recreational fishers do not respond to economic incentives, but are rather are motivated to maximise fishing opportunities and quality (e.g., catch-per-unit-effort). Management objectives for each sector generally reflect this difference; commercial management objectives generally seek to limit total harvest and fishing effort, whereas recreational management objectives are more likely to be structured to maximise fishing participation and quality, which may actually be incompatible (Walters and Cox 1999).

If direct and indirect incentives are improving sustainability of commercial fisheries, it seems sensible to consider what types of incentives might be applied to

recreational fisheries. Below I use an urban recreational-commercial crab fishery as a case study to explore:

- (i) management tools that could be used to plug the gaps that exist between recreational and commercial in each of the three management processes,
- (ii) incentives that improve the implementation success of these management tools, and
- (iii) obstacles to implementing each of the described tools.

A summary of this case study is provided in Table 2.1

2.4 Case study: An urban recreational-commercial Dungeness crab fishery in British Columbia

2.4.1 Overview of the fishery

Calm conditions, easy access and a dense human population make Burrard Inlet a popular destination for fishing. A relatively intense recreational crab fishery has developed in the Inlet, coincident with a small but efficient commercial fishery. Recreational fishing generally involves trapping either from shore or from boats, which allows a wide range of access and crabbing opportunities. Crabbers exploiting these stocks range from local residents trapping from piers, docks, or private boats to tourists utilizing the inlet as a yachting destination. Often recreational trapping occurs in the same spots as commercial fishing activity. Similar competitive, mixed-use fisheries occur throughout the Dungeness crab Pacific Coast range (e.g. the San Francisco Bay area and Puget Sound, USA) as well as for other invertebrate species across North

America (e.g. Blue crab in Chesapeake Bay and the Gulf of Mexico, Lobster resource in New England).

Equitable access to crab fishing is important in Canada, where management agencies are required to maintain recreational access even in areas of intensive commercial fishing pressure. One of DFO's guiding principles for managing recreational fisheries is that DFO is "...responsible for providing sustainable recreational harvesting opportunities as part of integrated management plans consistent with its policies" (DFO 2001). Under these principles, DFO will give consideration to increased or priority access for recreational use under the concept of "best use" of the resource, after obligations to conservation and First Nations are met (DFO 2001). To date, the management tools employed within the recreational sector to ensure equitable allocation of crab stocks have been open access policies coupled with input controls, described below.

Management tactics for recreational Dungeness crab harvest are fairly uniform along the Pacific Coast. Input control measures are applied that do not control fishing effort directly, but instead follow a "3-S" strategy, where harvest is limited by size, sex, and season regulations. Both commercial and recreational fisheries are managed in the same way by limiting harvest to legal-sized (minimum 165 mm carapace width, measured from tip to tip of the longest lateral spines) males only. However, in the lower mainland area of Vancouver, the recreational fishery is open year-round, while the commercial fishery is only open mid-June to late November. In most cases, recreational crab fisheries remain open-access with neither limits on total catch nor strict catch reporting requirements. While these conditions are supported by the recreational

crabbing community, I argue that they create unintended consequences that hinder their ability to attain broader stakeholder recognition and rights within the fisheries management framework.

2.4.2 *Recreational trapping within the Dungeness crab management processes*

Monitoring and assessment of crabbing activity are vastly different for the recreational and commercial sectors in Burrard Inlet. Commercial vessels are monitored electronically by continuous recording of: (1) vessel locations, speed, and direction; (2) hydraulic activity on the trap hauler; and (3) scanning of unique radio-frequency identification (RFID) tags on the traps to monitor trap limits. Total commercial harvest is also enumerated in mandatory logbooks and landed biomass is reported by management area with commercial sales slips, issued when they sell their harvest to buyers. The commercial fishery pays all costs associated with this monitoring. In contrast, recreational crabbing activities are not monitored and enforcement is low; therefore, total recreational crab harvest and fishing effort are unknown. This lack of data for the recreational fishery severely hinders attempts to assess the effect of recreational harvest on the crab population or determine appropriate allocation by means of historical use and makes it impossible for managers to determine if access to the resource is equitable across fishing sectors.

Lack of control over harvest and regulation compliance arises for the recreational Dungeness crab fishery in Burrard Inlet in part because harvest is not monitored and regulations are not easily enforced. Recently, there have been several convictions for recreational crabbers in Burrard Inlet highlighting that compliance with

the regulations (particularly input controls: size, sex, and bag restrictions) are likely very low for this sector. Poor controls over recreational harvest due to low enforcement of regulations have thus prompted increased restrictions on recreational crabbing by limiting trapping activity to daylight hours. However, proximity to the city of Vancouver and low enforcement of the regulations ensure that compliance problems will likely continue.

Because the greater efficiency of even a few commercial vessels shifts the outcome of competition heavily in favour of commercial harvest, an open-access policy cannot guarantee equitable harvest allocation between sectors. Previous studies have demonstrated that Burrard Inlet and surrounding crab fisheries have been heavily exploited at rates over 90% (Jamieson et al. 1998; Zhang et al. 2002). Our own surveys of legal and sub-legal crabs stocks in the adjacent Indian Arm crab fishery show a similar level of exploitation (Chapter 1). Following the commercial season opening, catch rate of legal-sized Dungeness crabs often declines swiftly, leaving much lower abundances for recreational trappers. Additionally, under the “3-S” management strategy, intensive fishing effort on these crab stocks has been identified as a factor that can threaten the long-term yield of crab stocks due to reduced recruitment and handling mortality of sub-legal crabs from frequent catch-and-release (Zhang et al. 2002), which is a consideration given DFOs mandate to prioritize conservation above all other considerations (DFO 2001).

High exploitation by commercial fisheries also creates conflict between the sectors and results in increasing calls for restrictions on commercial trapping and

greater harvest access for recreational trappers (DFO 2007). DFO has attempted to address these access issues in some mixed-use areas for First Nations and recreational harvesters through seasonal or permanent commercial closure of the relevant locations. Ideally, an equitable approach would involve an allocation of total catch between the competing sectors (i.e. commercial, recreational and First Nations). In practice, however, there are very few documented cases in which multi-sector harvest allocation of any kind has been attempted for marine fisheries, and almost never attempted for invertebrate stocks. The Washington State Dungeness crab fishery is a notable exception where crab harvest in some areas is divided equally between First Nations and non-First Nations users (US v. Washington 1994). Thus, the difficulty of allocating harvest to multiple sectors and a desire to change the *status quo* for the recreational fishery suggests that some form of exclusive harvest rights for the recreational sector may be the only feasible alternative.

There is merit to considering exclusive harvest rights for the recreational sector. Recreational crab landings are a small fraction of the commercial fishery (e.g. recreational Dungeness crab harvest has been estimated to be 1% of total harvest in California; Dewees et al. 2004), and allocating exclusive use rights to the recreational sector would likely reduce exploitation significantly and fulfill DFO's mandate to provide access to high quality recreational fishing. However, granting exclusive harvest rights to the recreational sector reduces diversity of employment in the resource harvesting sector as well as a sustainable source of seafood for general public consumption. From a management perspective, permanently closing the commercial fishery would involve a

replacement of a small, highly regulated fleet of commercial fishers with a large group of unmonitored and essentially unregulated fishers. The justification for such a shift from a mixed-use fishery to an exclusively recreational fishery is made even more difficult by the lack of recreational data and the compliance problems; where exclusive harvest rights have been attempted in other areas the catch and effort data to support the decisions has been limited (DFO 2007).

The problems with the Burrard Inlet recreational crab fishery defined above constitute a general failure to participate in the three management processes. Recreational crabbers have not been integrated into the management framework established for commercial crabbers in the area and despite the gaps in participation, exclusive use is increasingly contemplated for areas like Burrard Inlet for reasons of equitable allocation and feasibility. This represents a shift in the historical status of recreational fisheries. In being given exclusive use of the resource, recreational fishers shift from being a *dependent* stakeholder to a *dominant* stakeholder, defined by the power of their influence (i.e. they are the primary harvesters) and their legitimacy (Mitchell et al. 1997). Whereas the legitimacy of recreational fishing has historically been unquestioned in open-access mixed-use fisheries, by becoming the dominant stakeholders, their legitimacy is dependent on acting within the existing standards of sustainability for the fishery.

In the next section, I summarize ways in which other recreational fisheries have been integrated into the management processes and discuss ways in which they can or

cannot be applied to the Burrard Inlet crab fishery to close the legitimacy gaps between the commercial and recreational sectors.

2.5 Equitable mixed-use crab fishery: plugging the gaps

Commercial management protocols have generally proven beneficial for management of these important crab resources. However, what works for a small fleet of economically-motivated commercial crabbers will not necessarily work for a large group of recreationally-motivated crabbers. Both management cost sharing and the required incentives to participate in management can be different. Generally, low-cost management measures that motivate recreational crabbers are likely to be more successful at ensuring that the gaps between commercial and recreational management are diminished.

2.5.1 Closing the gaps in monitoring and assessment

2.5.1.1 Accountability in catch and effort reporting

Mandatory logbooks and crab vessel electronic monitoring systems provide high quality commercial catch and effort statistics for stock assessment and monitoring of fishing activities. Information provided in these datasets allows a manager to individually track fishing success and pinpoint the trapping locations of each vessel. This precision provides unprecedented monitoring of fishing activity, but comes at significant cost to DFO and the vessel. By comparison, there is no recreational crabbing data available. Moreover, while commercial fishers accept paying for biological sampling and

a portion of the management costs because of the economic payback, recreational fishers receive no economic gain from their fishing and therefore are unwilling to accept high fishing fees to enable harvest monitoring. Thus, relatively inexpensive options for obtaining reliable catch and effort data from the recreational sector will be needed to ensure compliance with this management process.

Studies have shown that data collected directly by the recreational community can be low cost and effective (Cooke et al. 2000). Where creel surveys are not realistic due to the effort and expense they require, two alternative management tools can be used to promote accountability in catch and effort reporting; (a) catch cards and (b) citizen science.

a. Catch cards

Recreational fishers are required to record their catch on catch cards in a number of freshwater and marine fisheries. For example, catch cards are used on the Pacific Coast to account for Salmon catch (DFO 2010b) and notably for recreational Dungeness crab catch and effort in Puget Sound, WA, USA. Washington Department of Fish and Wildlife (WDFW) began issuing catch cards to the recreational crabbers after the Fish and Wildlife commission made a significant policy change in 2000 granting exclusive use for the recreational community during the peak summer period in three management areas (WDFW 2000). The data on recreational harvest is used in estimating in-season catch relative to area harvest quotas and total harvest after the end of the season (Sonntag 2010). Because the state has this data, they are able to assess the harvest effects on the crab population from the recreational sector, evaluate tradeoffs,

and increase fishing rights for the recreational users. This information provides a valuable tool for managers and stakeholders that is not available for Burrard Inlet.

High rates of card return are critical to the success of this type of program, leading to issues with whether mandatory or voluntary reporting should be required. Voluntary reporting has been shown to be reliable for estimating basic fishery statistics (Gerdeaux and Janjua 2009) and cost effective, given appropriate program design and application (Cooke et al. 2000). However, one study evaluating voluntary catch card reporting in Mississippi lakes found that only 5% of catch cards were voluntarily completed and returned, increasing to 13% when agency personnel verbally requested participation (Walker et al. 2004). They also found a bias in reporting; older and more experienced anglers were more likely to participate. For these lakes, voluntarily completed catch cards were not found to be a viable substitute for creel surveys. Thus, effectiveness of these voluntary reporting programs can be dependent on the individual fisheries.

Catch card return rates are also an issue where it is mandatory. In the Puget Sound crab fishery, catch card returns have never been greater than 33%, despite incentive programs such as reminder post cards, free fishing licence draws, and Internet reporting. Therefore, in an effort to link catch reporting to an economic incentive, WDFW began fining crabbers \$10 in 2009 before re-issuing a license to improve participation (Sonntag 2010).

Catch cards are a promising tool for collecting data on recreational crabbing in Burrard Inlet, however ensuring accurate reporting is an important consideration.

Mandatory reporting was instituted in Puget Sound only after recreational crabbers were given exclusive use of the resource during the peak summer harvest period. Because recreational crabbing catch rates are a small fraction of commercial harvest in most mixed-use fisheries such as Burrard Inlet, it seems unlikely to be accepted by recreational crabbers without the incentive of exclusive use. Even where exclusive use is in place, making catch card compliance a requirement of license renewal may be necessary as an incentive for recreational participation in this management process.

b. Citizen Science

Engaging the public in scientific research has a long lineage in North America. “Citizen scientists” are individual volunteers or networks of volunteers, often untrained in scientific methods, who help monitor animal populations and ecosystems (Cohn 2008). The practice is very common in ornithology (e.g. the National Audubon Society’s annual Christmas bird count has been operating since 1900; Cohn 2008), but is also used in fisheries studies. Volunteers in these types of projects have been shown to be very capable at accurately collecting data on simple and straight forward tasks (Foster-Smith and Evans 2003). Because many recreational fishers are inherently interested in conservation and management, they constitute a social group with great potential for positively enhancing fisheries management (Arlinghaus 2006; Granek et al. 2008), and this conservation ethic acts as an incentive to participation.

Estimating fishing effort in a crab trap fishery is a simple operation that requires no equipment, aside from a boat, and a little experience. Because all recreational traps are connected to a buoy, estimating fishing effort is a simple matter of counting trap

buoys following an unbiased sampling design. This requires certain assumptions about the type of trap being fished (i.e. prawn trap or crab trap); however, this uncertainty can be evaluated with by pulling random traps, classifying traps by depth (i.e. prawn traps are generally set in deeper water), or simply requiring relevant labelling on the buoy. Just as with voluntary catch cards, where a small percentage of anglers were willing to provide accurate catch information, it does not seem unreasonable that reliable volunteers could be found who are willing to count traps. Tapping the recreational community's natural tendency and desire to get involved in management would produce benefits for the fishery and increase recreational accountability for the fishery.

2.5.1.2 Accountability in discard reporting

Handling mortality has been highlighted as an important issue for Dungeness crab fishery sustainability. In high exploitation fisheries, such as the Burrard Inlet, the discard rate of crabs is estimated to be greater than ten discarded sub-legal crabs to one retained legal sized crab during some periods (DFO 2007). Zhang et al. (2002) estimated that even moderate handling mortality under such conditions could negatively affect yield for Dungeness crab.

Estimating the lethal and sub-lethal effects of handling mortality is not easily accomplished (Zhang et al. 2002). However, similar to reporting catch, recording discards on catch cards would improve the data quality available to managers to ensure sustainability of the fishery. Given the experiences with catch card reporting in other fisheries, reporting discards would likely also need to be a condition of license renewal to ensure significant compliance.

2.5.2 *Closing the gaps harvest control*

2.5.2.1 Illegal harvest

There are no harvest targets for Dungeness crab under the current management framework, so control of the harvest is limited to ensuring compliance with input control fishing regulations. While fairly restrictive monitoring controls (e.g. electronic monitoring) have been placed on commercial Dungeness crab vessels to ensure compliance, controlling recreational harvest has proven difficult, demonstrated by persistent convictions in Burrard Inlet illegal for possession of female crabs, undersized crabs, and possessing more legal crabs than the daily limit.

While research into compliance with regulations in recreational fisheries is limited, a few examples from marine and freshwater fisheries have contributed to the field. Kuperan and Sutinen (1998) demonstrated that an individual fisher's decision to comply with regulations is influenced largely by their perceptions of deterrence (i.e. enforcement) and the regulatory legitimacy¹. Therefore, in theory managers can increase compliance to regulations by: (a) boosting the perception of deterrence; or (b) boosting the perceived legitimacy of regulations within the recreational community through co-management frameworks or increased education and outreach.

a. Increasing deterrence

A recent study conducted in Alberta lakes demonstrated an empirical relationship between compliance and deterrence (Walker et al. 2007). Illegal harvest

¹ Note that legitimacy in this context is a measure of the public support for the regulations; different from the key attribute of a stakeholder, defined above.

was reduced when anglers had a high perception of deterrence, which is defined as a product of the certainty of detection and severity of punishment. Walker et al. (2007) examined the effect of enforcement patrols and the use of signage. They found that signs had an effect on anglers' perceptions of the severity of the punishment while patrols had an effect on anglers' perception of certainty of detection. However, the effects of patrolling were limited, and reached an asymptotic maximum certainty of detection when 3% of anglers were approached by conservation officers. These results suggest that continued use of appropriate signage and modest enforcement in the Burrard Inlet can have considerable impact on illegal harvest. Because recreational fishers may respond to declining fish stocks by increasing illegal harvest (Sullivan 2002), ensuring compliance to regulations becomes increasingly important as stocks deplete. As the pool of legal-sized crabs is depleted in the course of the fishing season, increased enforcement patrols could help deter potential increases in illegal harvest. However, this has a limited effect with diminishing returns as enforcement costs increase.

b. Increasing regulatory legitimacy

Co-management framework

In the absence of a strong deterrent, authorities need to obtain legitimacy in the management process and regulations from their constituents to use power effectively and gain acceptance of and compliance with social policies and regulations (Pinkerton and John 2008). The recent series of regulation violations has demonstrated that the legitimacy of the regulations is low for the recreational crabbers in Burrard Inlet. The diverse and numerous users in the recreational fishery have little stake in the success of

the fishery and therefore few incentives to follow regulations, aside from personal conservation ethics. One mechanism to increase regulatory legitimacy is to give participants a stake in its management.

Pinkerton and John (2008) describe a process whereby the legitimacy of harvest regulations and management has been established for a clam fishery adjacent to a First Nations community in Kyuquot Sound, British Columbia. Regulatory legitimacy increased through establishing a co-management structure with the First Nations harvesters in the community. Traditional command-and-control style management was devolved to a more local management framework (Pinkerton and John 2008), which improved all of the management processes for the fishery. Prior to this policy change, similar concerns were described for this fishery as are present in the Burrard Inlet crab fishery, namely active poaching, a depleted resource (certainly true for Dungeness crab soon after the commercial opening) and an open access “race to fish” style of harvest management. The fishery was also structurally similar, with recreational fishing activity generally visible and confined to small areas close to access points. Although the similarities of these two fisheries suggest that co-management could be a useful management tool in Burrard Inlet, there are key differences in community characteristics that make co-management unlikely to be as successful as the Kyquot Sound clam fishery.

In Kyuquot Sound, the First Nations harvesting community is small and remote, with strong cultural ties and dependence on the clams as food source. This is almost completely opposite to the urban recreational community in Burrard Inlet (i.e. large, disparate group of recreational harvesters with weak cultural ties and no dependence as

a food source). A co-managed fishery necessitates some sort of representative association to put forward management issues.

DFO collaborates with the recreational community through the Sport Fish Advisory Board (SFAB), composed of representatives from recreational fishing and business/industry organizations and regional resource managers. The SFAB meets regularly with DFO resource managers to discuss issues with recreational fisheries, and it is through this process that many new policies are developed for the recreational community. Sutinen and Johnston (2003) describe how angling management organizations (AMOs) might be used to facilitate management devolution, and strengthen harvest rights and co-management for the Gulf of Mexico red snapper fishery, in a similar fashion to Kyuquot Sound. They argue that AMOs would improve recreational management because they meet seven principles of integrated management (Table 2.2; Sutinen and Johnston 2003). It is conceivable that the SFAB could act as a Sub-regional recreational management council (as described in Sutinen and Johnston 2003), should data and harvest control improve in the recreational crabbing community. However, not one of the principles of integrated management justifying such a paradigm shift are present in the Burrard Inlet recreational crab fishery. Therefore, co-management is an implausible option at present, and unlikely be a useful tool for increasing regulatory legitimacy.

Increasing education and outreach

Assessments of environmental education programs consistently show that environmentally knowledgeable people are more likely to behave in pro-environmental

ways (Angermeier, 2007 and references therein). Fisheries managers use education to disseminate knowledge and to attempt to change attitudes within the fishing community. Identified educational requirements include promotion of general environmental stewardship (Granek et al. 2008), improvements in regulation awareness (Page and Radomski 2007) and sponsorship of best fishing practice guidelines for reducing the impacts of fishing on ecosystems (Lewin et al .2006; Cooke and Suski 2005).

Educational tools for recreational fisheries include regulation summaries, public access signs, local media, word of mouth, and community-based participatory restoration projects (Page and Radomski 2006; Angermeier 2007). The effectiveness of these tools vary as do their application. For example, when considering ways to increase regulatory awareness in Minnesota fisheries, Page and Radomski (2006) found that not all types of angling groups were informed about the regulations and not all regulations required the same amount of promotion. Regulation awareness was maximized when groups that receive little exposure to regulation information (e.g., generalist anglers or non-residents) were targeted and managers concentrated on promoting new regulations.

Another similar option for increasing regulatory legitimacy is to increase outreach with the recreational community. Three important factors determining the type of involvement and the likelihood of recreational fisher interest in involvement (Granek et al. 2008). First, presence of an environmental stewardship ethic amongst anglers facilitates support for conservation measures and solidifies commitment from

anglers. Second, the scale of the fishery is important because the smaller the fishery, the more likely anglers are to perceive that their actions directly influence fishery sustainability, making involvement in management more likely. Finally, whether the threat to the fishery comes directly from recreational fishing will influence involvement. If it does, anglers are more likely to resist direct conservation involvement than if the source of the threat is external to fishing, such as commercial fishing or habitat destruction. When the combination of factors leads to high likelihood of angler involvement in a range of management activities, the result is a net positive benefit to conservation and sustainability.

2.5.3 *Closing the gaps in harvest allocation*

2.5.3.1 Status quo heavily favours commercial fishery

The “3-S” strategy prescribed for Dungeness crab management has been surprisingly resilient to very high exploitation rates and recruitment variability seen in Dungeness crab fisheries (Orensanz et al. 1998; Helliwell 2009). However, in a mixed-use fishery with open access, the recreational sector is outmatched by the effectiveness of the commercial vessels. Additionally, the infeasibility of applying limited entry or quota strategies to recreational fisheries leaves very few options for resource sharing amongst sectors. Therefore, commercial fishing exclusion areas are increasingly being granted to both recreational and First Nations fisheries to mitigate the competitive advantage of commercial harvesters.

Area-based exclusivity (i.e. commercial closure) is only one possible way of limiting commercial access. Two more moderate forms of exclusivity are possible: (a) differential size limits; and (b) temporal exclusion.

a. Differential size limits

Different size limits for BC commercial and recreational Dungeness crab fisheries have not been tried, but have been effective at providing access to recreational fishers in other important invertebrate fisheries (e.g. abalone stocks in Western Australia; Mitchell and Baba 2006). For BC crab fisheries, increasing the commercial size limit has been suggested as an equitable alternative to total commercial closures (Phillips and Zhang 2004). The current legal size class would remain the same for non-commercial harvesters, providing stable and equitable access to the resource. Phillips and Zhang (2004) estimated that increasing the commercial size limit by 5mm would produce a 22% loss in total harvest for commercial crabbers in the Vancouver area during the first season. However, they suggest that reducing exploitation on the previously harvested size classes may reduce future losses to the commercial fishery by increased recruitment and crab growth. Recreational crabbers also have a much lower exploitation rate than commercial crabbers, potentially producing future dividends for all sectors by increasing the abundance of larger crabs and increasing larval production by allowing a greater proportion of larger crabs to breed (Phillips and Zhang 2004).

The major obstacle to this management strategy is the initial losses to the commercial harvest. However, the current situation in competitive mixed-use fisheries (e.g. Ucluelet, Burrard Inlet) is not perceived to provide reasonable access for

recreational fishers. Differential size limits would likely provide reasonable allocation to the recreational community, without major structural changes to the management framework.

b. Temporal exclusion

Restricting commercial crabbing to off-peak periods of the year is another option for equitable resource sharing between sectors, as an alternative to total commercial closure. For example, restrictions on commercial fishing are lifted in the Puget Sound crab fishery in the fall, after the recreational and First Nations fishery have concluded. This policy is a way of ensuring that the legal sized crabs are fully utilized and provides access to the recreational and First Nations fisheries.

Similar to differential size limits, an argument against implementation of temporal exclusion is the economic losses to the commercial sector. However, unlike differential size limits, the exclusion would not be mitigated by potential future gains from increased growth and recruitment. Commercial crabbers with a harvest history in Burrard Inlet would have their activities dramatically curtailed.

2.6 Management recommendations

Dungeness crabs on the West Coast have been perceived to be resilient to high exploitation under the current “3-5” management strategy, despite a number of small-scale collapses (Orensanz et al. 1998). Therefore, any changes to management are not likely to be initiated because of sustainability, but rather allocation. Providing equitable access between sectors has become increasingly important, and the only feasible option

for allowing DFO to provide access to high quality fishing for the non-commercial sectors is some form of exclusivity, either area-based exclusion, differential size limits or temporal exclusion. This is unfortunate because there is little data on catch and effort within the recreational sector with which to justify such an approach (DFO 2007), in addition to poor control over recreational harvest. Deficiencies in monitoring, assessment and control over recreational harvest lead to deficiencies in allocation.

Issues with catch monitoring and control in any fishery are complicated, particularly when dealing with multiple competing sectors. Innovative management strategies, such as individual quotas, that have evolved to work well for a commercial fishery will not necessarily be work in a recreational context. Some of the incentives that recreational fishers respond to are quite new for crab fisheries, such as excluding commercial fishing. However, most of the alternatives discussed make use of incentives already entrenched in recreational fisheries management; regulations, enforcement and stewardship.

Management tools such as catch cards, citizen science and increasing deterrence are viable options for reducing the legitimacy gaps between the commercial and recreational sectors. In turn, the increased legitimacy of the recreational sector as a whole would help justify reducing the allocation gaps.

2.7 Tables

Table 2.1: Table of common deficiencies in recreational management relative to commercial crab fisheries management in Burrard Inlet. Differences constitute a legitimacy gap impeding crab fisheries allocation of full harvest rights seen in the commercial fishery. Included are potential management tools, obstacles, incentives, and examples for recreational fishers to close the gaps in the three functions of fisheries management.

	Management tool	Obstacles	Incentive for participation	Key Examples
i. Monitoring and assessment gaps				
Accountability in catch and effort reporting	Catch reporting cards	-Non-compliance	-Regulatory (mandatory condition of licence)	Puget Sound crab (WDFG) Yukon salmon (DFO)
	Citizen Science	-Lack of interest/resources	-Conservation ethic	Foster-Smith and Evans (2003) Granek et al. (2008)
Accountability for discard reporting	Discard reporting on catch cards	-Non-compliance -Not required for the commercial sector	-Regulatory (mandatory condition of licence)	Puget Sound crab (WDFG)
ii. Harvest control gaps				
Illegal Harvest	Deterrence - Signage and increased enforcement	-Limited effect	-Regulatory	Sullivan (2002) Walker et al. (2007)
	Increase regulatory legitimacy - co-management	-Requires definable associations between recreational users -None of the principles of integrated management have been met	-Conservation ethic -Increased contribution to management	Sutinen and Johnston (2003) Pinkerton and John (2008)
	Increase regulatory legitimacy - education/outreach	-Effectiveness varies by regulation and with experience of fisher	-Conservation ethic	Granek et al. (2008) Page and Radomski (2006)
iii. Allocation gaps				
Open access fishery heavily favours commercial fishery	Exclusive Use- -Commercial closure -Differential size limits -Temporal exclusion	- Gaps in assessment, monitoring and control of recreational harvest -Losses to commercial harvest	-Reduction in competition with commercial sector -Exclusive harvest rights	Hilborn et al. (2005) Phillips and Zhang (2004)

Table 2.2: Seven basic principles for improving the management of recreational fisheries developed by Sutinen and Johnston (2003). Each principle builds on the previous principle, and all are essential ingredients for fully realizing the benefits of co-management.

Seven principles of integrated management

Principle 1: Integrated recreational management is desirable only where the benefits of integration outweigh the costs of integration.

Principle 2: A workable mechanism must exist for allocating catches among recreational, commercial and other user groups as a precondition for integrated recreational management.

Principle 3: Managers must implement management measures that in practice provide a high degree of control over recreational fishing mortality.

Principle 4: Recreational fishery management should be based on a system of strong angling rights.

Principle 5: Recreational fishery managers should consider assigning angling rights to organizations or other groups as well as individuals in recreational fisheries.

Principle 6: Recreational fishery management should be decentralized with limited management authority devolved to and shared with local organizations and governing institutions.

Principle 7: Cost recovery should be applied to recreational fishery management since it will strengthen accountability and improves the overall performance of the management program.

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APPENDIX 1: SAMPLING SCHEDULE

Table A1.1: Sampling schedule for trap survey of Burrard Inlet at fixed sites in 2007.

Site (depth)	Sampling Period	Date	Number of traps
Admiralty Point (10-20m)	1	5/24/07	13
	2	6/17/07	5
	3	7/16/07	5
	4	8/12/07	10
	5	9/24/07	9
Bedwell Bay (10-50m)	1	5/22/07	13
	2	6/19/07	5
	3	7/18/07	5
	4	8/15/07	10
	5	9/17/07	9
Dan George (10-30m)	1	5/26/07	12
	2	6/18/07	5
	3	7/17/07	5
	4	8/17/07	10
	5	9/25/07	9
Deep Cove (50-80m)	1	5/28/07	11
	2	6/19/07	5
	3	7/18/07	5
	4	8/14/07	10
	5	9/18/07	9
Dollarton (10-30m)	1	5/23/07	13
	2	6/17/07	5
	3	7/16/07	5
	4	8/13/07	10
	5	9/16/07	9
ROCHE (10-30m)	1	5/25/07	13
	2	6/18/07	5
	3	7/17/07	5
	4	8/16/07	10
	5	9/26/07	9

Table A1.2: Sampling schedule for trap survey of Burrard Inlet at fixed sites in 2008.

Site (depth)	Sampling Period	Date	Number of traps
Admiralty Point (10-20m)	1	5/20/2008	12
	2	6/12/2008	12
	3	7/7/2008	12
	4	9/3/2008	12
Bedwell Bay (10-50m)	1	5/21/2008	13
	2	6/11/2008	12
	3	7/8/2008	12
	4	9/4/2008	11
Dan George (10-30m)	1	5/22/2008	12
	2	6/10/2008	12
	3	7/8/2008	11
	4	9/2/2008	12
Deep cove (50-80m)	1	5/21/2008	10
	2	6/10/2008	12
	3	7/7/2008	12
	4	9/4/2008	12
Dollarton (10-30m)	1	5/20/2008	11
	2	6/12/2008	12
	3	7/9/2008	12
	4	9/3/2008	11
ROCHE (10-30m)	1	5/22/2008	11
	2	6/11/2008	12
	3	7/9/2008	11
	4	9/2/2008	11

Table A1.3: Sampling schedule for trap survey of Burrard Inlet at random sites in 2007.

Date	Sampling Period	Depth stratum	Number of traps
6/6/07	1	1	5
6/7/07	1	3	6
	1	2	5
6/8/07	1	3	5
	1	1	5
7/3/07	2	3	5
	2	2	5
7/4/07	2	2	5
	2	1	5
7/5/07	2	1	5
	2	3	5
7/30/07	3	3	5
	3	2	5
7/31/07	3	1	5
	3	1	5
8/1/07	3	1	5
	3	3	5
8/2/07	3	2	5
	3	3	5
8/28/07	4	1	5
	4	2	5
8/29/07	4	3	5
	4	1	5
8/30/07	4	3	5
	4	1	5
10/1/07	5	1	4
	5	1	5
10/2/07	5	2	4
	5	2	5
10/3/07	5	3	4
	5	2	5

Table A1.4: Sampling schedule for trap survey of Burrard Inlet at random sites in 2008.

Date	Sampling Period	Depth stratum	Number of traps
5/28/2008	1	1	7
	1	2	22
	1	3	18
5/29/2008	1	1	7
	1	2	14
	1	3	27
7/15/2008	2	1	3
	2	2	18
	2	3	33
7/16/2008	2	1	5
	2	2	22
	2	3	21
9/8/2008	3	1	3
	3	2	30
	3	3	15
9/9/2008	3	1	6
	3	2	20
	3	3	21

APPENDIX 2: ROV OPERATIONS

ROV configuration

All ROV transects were executed with a Deep Ocean Engineering model DHD 2+2 ROV. The recorded ROV video displayed time, depth and compass heading, and transects were conducted using a Tritech Sea King® sector-scanning sonar to enhance transect quality in poor visibility. The main ROV camera is a Sony® low light, colour CCD camera capable of tilt and zoom functions. The angle of the view varied during transects dependant on factors such as visibility, angle of substrate and height off bottom. However, the mean width of the camera field of view was estimated using a pair of parallel red lasers mounted on the camera with a spacing of 10cm.

ROV position was tracked and recorded using a Trackpoint II® Ultrashort Baseline (USBL) acoustic transponder system and Hypack Max® navigation software. Vessel position, time and depth were recorded at the beginning and end of each transect in a separate log.

Transects

ROV time was limited to 2 hours per site for each sampling event. The objective for the ROV survey, therefore, was to randomly choose 3 half-hour transects within each site for each sampling period. Transects were selected by choosing two random points (start and end positions) within each site and following the prescribed linear track. Transects were automatically ended after 30 minutes on the bottom, however if

the defined transect was too short the heading was followed until the 30 minute mark or the area boundary was reached.

Successfully piloting the vehicle along the transect was dependent on tides, surface environmental conditions and obstructions or snags on the bottom. The pilot attempted to maintain constant heading and speed wherever possible. Generally, the benthic habitat was low gradient, mud sediments with low complexity; therefore environmental conditions on the surface and on the seabed were the primary reason for deviating from the track. Where maintaining the correct heading proved impossible, a new heading was chosen that could be maintained for the duration of the transect.

Our ability to track the ROV along the bottom varied. For the initial sampling period in June, 2007 I did not have the benefit of the Trackpoint II system, so I relied on the recorded vessel position at the start and end of the transect (Table A2.1). For subsequent transects where positional data was available, the quality of the data was variable. After removing tracking-induced outliers, the data was smoothed in R using Friedman's Super Smoother (R Development Core Team, 2008). The ROV track was inferred by linear interpolation between smoothed points.

To evaluate our use of logbook vessel position for some of the transects I calculated the linear distance between logged endpoints and measured the interpolated track lengths using ArcMap 9.0 GIS® software (Figure A2.1). Results indicate that logbook data underestimates that track length of the transect by a mean of 12% (Figure A2.2).

Transect area swept was calculated by taking the product of the measured transect length and the average transect width. Mean transect width (W) was estimated by measuring the distance between the lasers on the monitor screen (L) every 30 seconds while reviewing the video and calculated as follows:

$$W = \frac{M * 0.1}{L}$$

M is the width of the view on the monitor and 0.1 meters is the actual width of the lasers reflected on the substrate. The mean transect width was 1m; however the distribution had a positive skew towards larger values with a median of 0.9m (Figure A2.3).

Video viewing

All crab species were identified to species and enumerated for each transect. Crabs buried in sediments posed a particular concern; however buried crabs were identifiable from the depression left in the substrate for respiration and often from their visible rostrum and antennae. Observations of buried individuals were frequently verified when the crab exited out of the sediments as the ROV approached. Parry et al. (2002) have also demonstrated that ROVs are effective at detecting buried megafauna in un-vegetated substrates.

Dungeness crab size was also measured across the carapace wherever feasible while viewing the video. Carapace width (CW; millimeters) was calculated by measuring the carapace width of the crab on-screen (CW_{screen}) and the distance between the lasers on the monitor screen (L) as follows:

$$NW = \frac{NW_{screen} * 100}{L}$$

100mm is the distance between laser points on the substrate. The aspect of the crab in relation to the ROV was an important characteristic for measurement. When the crab was oriented at an oblique angle to the camera view, it was not always possible to obtain an accurate measurement across the carapace. In these cases, half the width of the carapace was often measured and doubled, when possible. I was able to obtain reliable estimates on 65% of the viewed crabs. I assumed that our ability to measure crabs was a random process and there was no systematic bias affecting our resulting size distributions.

Suggestions

This survey represents our initial attempt at operating the ROV during a scientific survey on the *CJ Walters* research vessel. The sampling protocol was set from the beginning; however, ROV operations were improved upon over the two field seasons. For future surveys of this type, a few recommendations may be useful.

The camera angle is important for making size measurements of individual crabs in relation to the substrate. Measuring crabs became more uncertain with distance from the ROV; therefore the camera should be angled just ahead of the ROV wherever possible.

At times, when operating in shallow depths, the lights mounted on the ROV were switched off and the ROV operated with the ambient light. While this enabled a greater

transect width, it became very difficult to distinguish individual crabs (which appear red under the lights) from the other features on the bottom. Buried crabs were near impossible to distinguish. I recommend that the lights remain illuminated throughout the transect.

For some of our transects, the Hypack was initialized while the ROV was diving prior to the transect and ended logging while the ROV surfaced after the transect. This tendency was easily recognizable as the course correction on either end of the transect (Figure A2.1), however it added needless ambiguity in the transect start and finish positions. Keeping a written log of vessel positions also proved valuable in post-dive video viewing and data analysis.

Table A2.1: Survey dates and transect lengths in meters. Transect length was estimated from the dive logbook (indicated by *) when Trackpoint II acoustical data was unavailable or unreliable. The number of useable tracking locations for each estimate of track length is also indicated.

Site	Date	Transect	Track Length	Trackpoint II
Bedwell Bay	6/4/2007	1	302*	-
		2	268*	-
		3	103*	-
Dollarton	6/5/2007	1	64*	-
		2	99*	-
		3	370*	-
Admiralty Pt.	6/7/2007	1	378*	-
		2	364*	-
Admiralty Pt.	6/19/2007	1	249	1469
		2	369	2308
		3	361	2131
Dan George	6/20/2007	1	470	2346
		2	521	2581
		3	216	1072
Bedwell Bay	6/21/2007	1	274	1449
		2	252	1615
Deep Cove	6/22/2007	1	373	2182
		2	273	2151
		3	203	1254
Bedwell Bay	7/17/2007	1	361	1975
		2	278	2054
		3	394	2117
Admiralty Pt.	7/18/2007	1	223	1804
		2	381	2117
		3	264	1925
Dan George	7/19/2007	1	288	1822
		2	298	1955
		3	434	2158
Deep Cove	7/20/2007	1	450	2461
		2	71*	-
		3	412	3067
Bedwell Bay	8/14/2007	1	353	2613
		2	449	1827
		3	370	1897
Admiralty Pt.	8/15/2007	1	375	2031
		2	307	2056
		3	377	2112
Dan George	8/16/2007	1	271	1270
		2	265	1372
		3	310	1790
Deep Cove	8/17/2007	1	352	1972
		2	304	1891
		3	405	2542

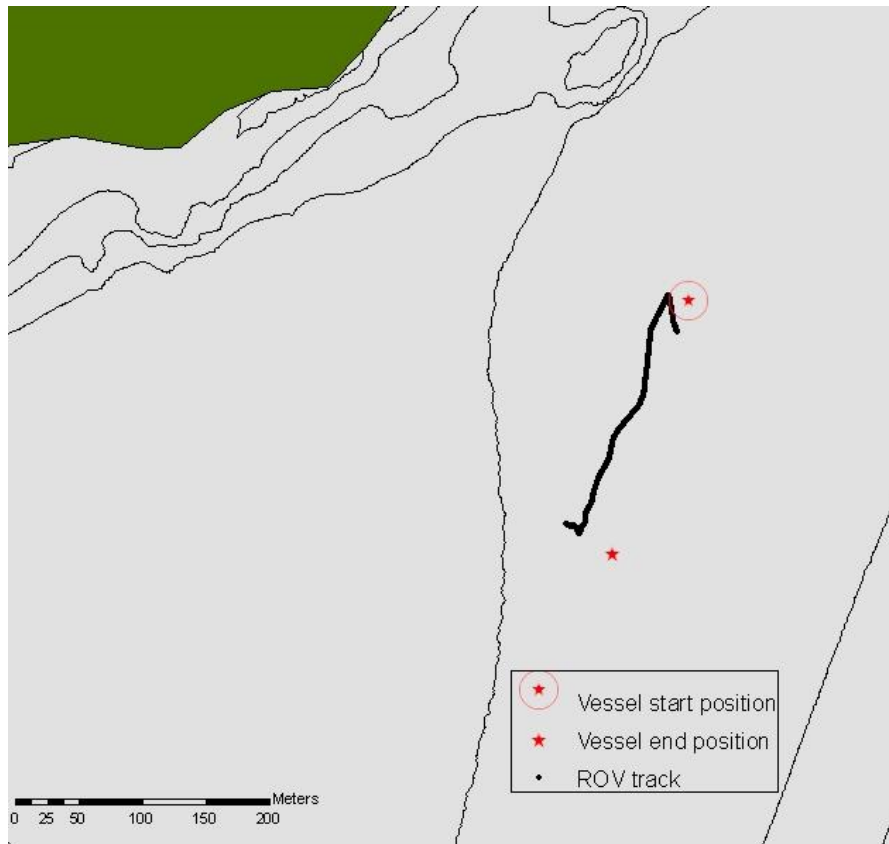


Figure A2.1: Example of two data types used for determining track length. The ROV track was interpolated from Trackpoint II data and the vessel positions were recorded in a log at the start and end of the transects. The course corrections at the beginning and end of the ROV track are from the ROV diving and surfacing. The lines are ten meter contour intervals.

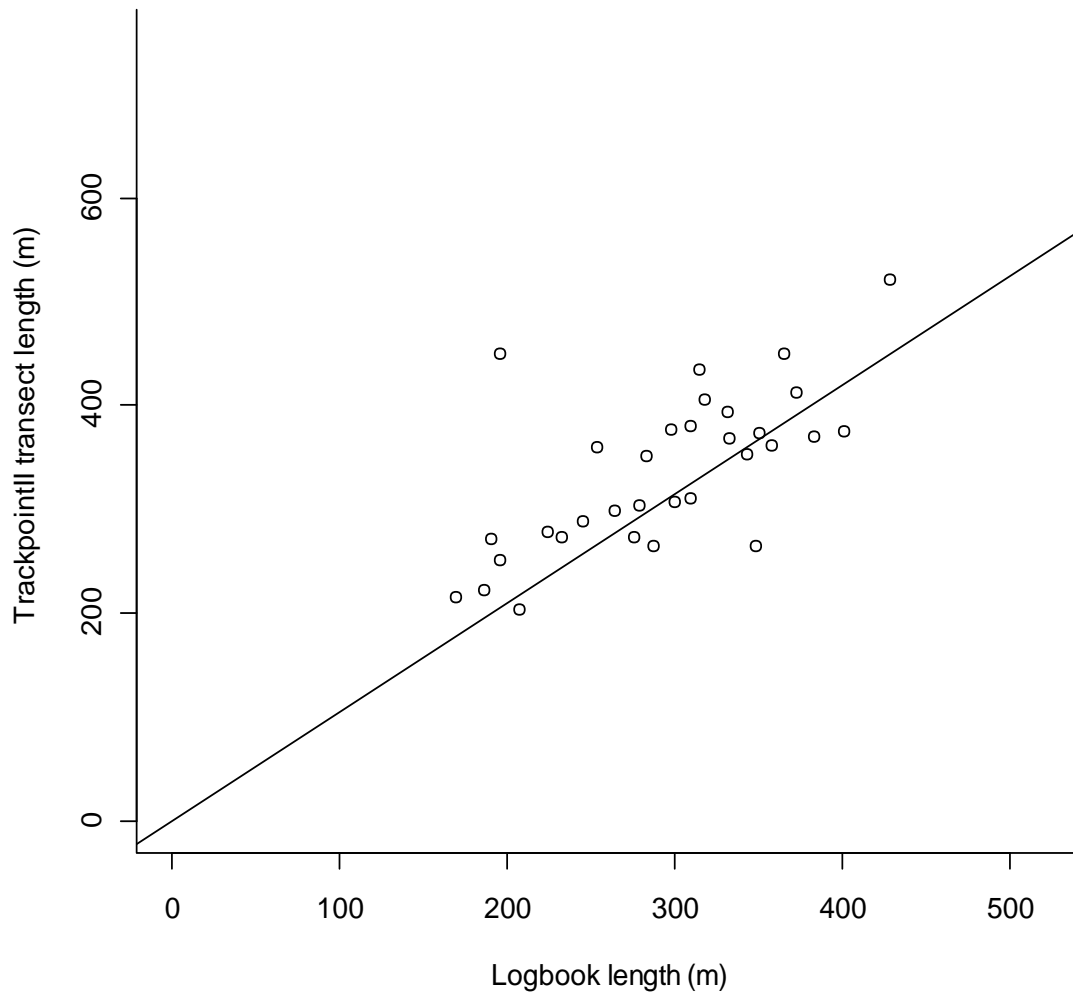


Figure A2.2: Relationship between measured track length from the Trackpoint II system and linear distance between vessel positions at the start and end of the transect. The solid line represents the linear regression predicted from the observed data (points).

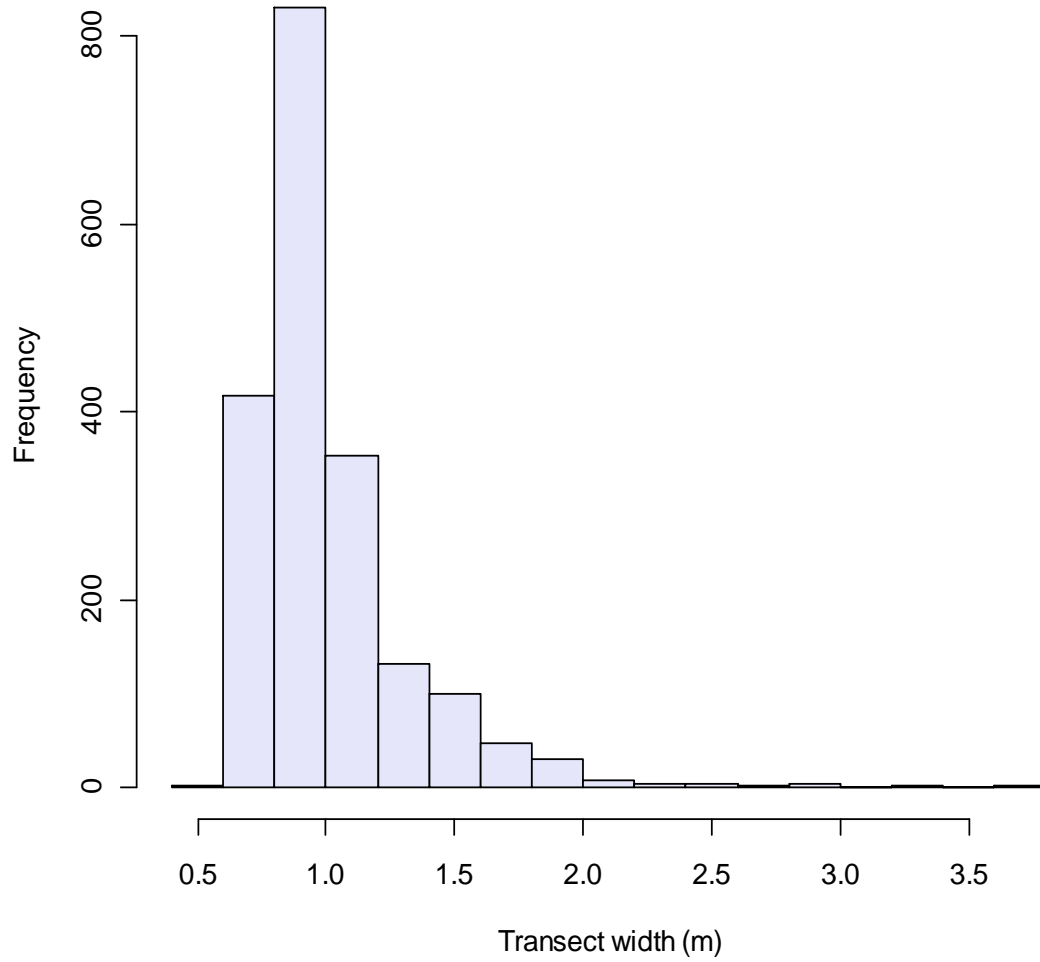


Figure A2.3: Pooled transect width estimates observed during the ROV surveys.