Exxon Valdez Oil Spill
Restoration Project Final Report

Harlequin Duck Population Dynamics: Measuring Recovery from the *Exxon Valdez* Oil Spill

Restoration Project 040407
Final Report

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Study History: Restoration Project /407 began a new phase of harlequin duck (*Histrionicus histrionicus)* studies in 2000, with a focus on measuring recovery from spill-induced injury. The original studies, assessing injury to the species, were initiated in 1991 by the Alaska Department of Fish and Game with Bird Study Number 11 (*Assessment of Injury to Sea Ducks from Hydrocarbon Uptake in Prince William Sound and the Kodiak Archipelago, Alaska, Following the Exxon Valdez Oil Spill*) and Restoration Study Number 71 (*Breeding Ecology of Harlequin Ducks in Prince William Sound, Alaska*). These earlier studies concluded that the number of harlequin ducks inhabiting oiled areas in western Prince William Sound (WPWS) declined as a result of the *Exxon Valdez* oil spill in 1989. The decline was attributed to direct mortality caused by oiling, and to subsequent low productivity of ducks that survived or avoided initial exposure. A Masters of Science thesis describing breeding habitat of harlequin ducks was also produced during the course of these initial studies (Crowley, D. W. 1994. *Breeding habitat of harlequin ducks in Prince William Sound,* Alaska. M. S. Thesis. Oregon St. Univ., Corvallis. 64pp.). Restoration Project (RP) 94427 (*Experimental Harlequin Duck Breeding Survey*) was initiated in 1994 in response to concerns that post-spill productivity by harlequin ducks in WPWS was not at a level necessary to maintain a viable population. The study developed criteria to differentiate harlequin ducks by age and sex to compare demographic characteristics of populations inhabiting oiled areas in WPWS with unoiled areas in eastern PWS (EPWS). Variation in population structure between areas would indicate dissimilar extrinsic influences affecting harlequin populations. A survey design was also developed to determine trends in harlequin abundance and production. Restoration Project /427 (*Distribution, Abundance and Composition of Harlequin Duck Populations in Prince William Sound, Alaska*), 1995-1997, utilized methods derived from RP 94427. Results from surveys conducted from 1995-1997 (Final Rept. 97427) found no major differences in population structure or timing of movements between WPWS and EPWS but did detect a decline in numbers of ducks in oiled areas of WPWS and no significant change in numbers in unoiled areas of EPWS. Winter surveys, which were utilized in RP /407, were originally initiated in March 1997 for RP 97427.

Abstract: We compared sex and age composition, and population trends of harlequin ducks *(Histrionicus histrionicus)* between oiled and unoiled treatments in Prince William Sound during six winters from 1997–2005. Sex ratios were skewed towards males in all treatments, consistent with other populations of Pacific harlequin ducks. Sex ratios were significantly different between treatments ($P = 0.022$) with the oiled treatment having a lower proportion of females. Recruitment varied annually but not by treatment ($P = 0.502$). Annually, we observed a slight increase in recruitment. We found no significant difference in the change in density (trends) between oiled and unoiled treatments ($P = 0.761$) and the mean rate of change for oiled areas (0.0125, $P = 0.138$) and unoiled areas (0.0186, $P = 0.304$) was not significantly different from zero. The lower proportions of females in oiled areas provided the only evidence for a possible lingering oil spill effect. Demographic data interpreted in concert with other biological

parameters leads us to conclude that harlequin duck populations are recovering from the *Exxon Valdez* oil spill.

<u>Key Words</u>: *Exxon Valdez* oil spill, harlequin duck, *Histrionicus histrionicus,* population monitoring, Prince William Sound, restoration, sea ducks.

<u>Project Data</u>: *Description of data* - Data on sex, age, and location were recorded for each flock of harlequin ducks observed in PWS. *Format* - These data are in Microsoft Excel spreadsheet format and DBASE IV format. GIS coverage of PWS showing the location of flocks, survey transects, broods, and streams are presented in ARC VIEW format. *Custodian* - Archived at ADF&G regional headquarters in Anchorage. Contact Dan Rosenberg at ADF&G, 525 West 67th Street, Anchorage, Alaska 99518 (907-267-2453) for information. E-mail: dan_rosenberg@fishgame.state.ak.us

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EXECUTIVE SUMMARY

This study was initiated to determine whether the harlequin duck population in oiled areas of Prince William Sound recovered or is in the process of recovering from the effects of the 1989 *T/V Exxon Valdez* oil spill. Continued oil exposure was the most likely mechanism constraining recovery through 1998 (Esler et al. 2002).

Harlequin ducks *(Histrionicus histrionicus)*, a sea duck *(Mergini*), occur year-round in PWS (Isleib and Kessel 1973) and were the most abundant waterfowl species in nearshore habitats prior to the *T/V Exxon Valdez* oil spill (Irons et al. 1988). On March 24, 1989, the *T/V Exxon Valdez* ran aground in northern PWS oiling 783 km of shoreline before spreading to the Gulf of Alaska (Galt et al. 1991, Piper 1993, Neff et al. 1995). Post-spill studies estimated that between 420 and 1838 harlequin ducks died in PWS as a direct result of the *T/V Exxon Valdez* oil spill (Ecological Consulting, Inc. 1991, Piatt and Ford 1996, J.F. Piatt, pers. comm.)

Harlequin ducks are particularly vulnerable to oil spills because of their fidelity to nearshore molting and wintering areas (Robertson et al. 1999, Robertson et al. 2000, Cooke et al. 2000) where they forage for invertebrates in intertidal and shallow subtidal zones (Dzinbal and Jarvis 1982). Further, sea ducks are sensitive to catastrophic causes of mortality because long-term population stability depends on high adult survival (Goudie et al. 1994). As a result, full recovery may be delayed until well after the absence of any spill effects (Esler et al. 2000b).

In 1997, we began winter surveys in western Prince William Sound (oiled) and eastern Prince William Sound (unoiled) study areas (Rosenberg and Petrula 1998) to monitor harlequin duck demographics (population trends and composition). In 2000, we added additional study areas on Montague Island (unoiled) and southwestern Prince William Sound (oiled). We hypothesized that the population structure and trend in oiled and unoiled areas of Prince William Sound would be similar if the harlequin population in oiled areas had recovered or was in the process of recovering from the effects of oil exposure.

In 1997, we counted 2,860 harlequin ducks along 550.3 km of shoreline in western and eastern Prince William Sound (Rosenberg and Petrula 1998). From 2000–2005, combining all areas, numbers ($\bar{x}$ = 4,964.8 (SD = 147.0) and densities ($\bar{x}$ = 6.6 ducks/km) were relatively stable along approximately 746 km of shoreline. Numbers of ducks ranged from 4,823 to 5,186. Densities ranged from 4.8 ducks/km in southwestern Prince William Sound in 2000 to 10.6 ducks/km at Montague Island in 2000.

Sex ratios were skewed towards males in both treatments, consistent with other populations of Pacific harlequin ducks (Smith et al. 2001, Rodway et al. 2003). Sex ratios were significantly different between treatments (P = 0 .022) but not among years or regions. The oiled treatment had lower proportions of females than the unoiled treatment. Recruitment varied annually but not with oiling history (P = 0.502). We observed a slight increase in recruitment over the course of the study.

We found no significant difference in the change in density (trends) between oiled and unoiled treatments (t = -0.30, P = 0.761, DF = 193.9) and the mean rate of change for oiled areas (0.0125, P = 0.138) and unoiled areas (0.0186, P = 0.304) was not significantly different from zero.

The *Exxon Valdez* Oil Spill Trustee Council (2002) defined recovery as a return to prespill demographics and similar levels of hydrocarbon exposure in ducks from oiled and unoiled areas (treatments). Without good pre-spill data we were limited to measuring recovery in relative terms by comparing changes in abundance and composition between treatments within PWS. Similar age and sex composition and numbers of ducks in oiled areas increasing at an equal or greater rate than in unoiled areas would indicate that the harlequin population has recovered from the effects of the *T/V Exxon Valdez* oil spill. We would interpret differences in these demographic parameters as evidence of continuing injury. Once recovered, demographic parameters would converge and our two treatments would exhibit parallel changes (Wiens and Parker 1995).

With an oil spill we would expect different survival rates between males and females if oil, directly or indirectly, had a more pronounced effect on either sex. We observed a lower proportion of females in the oiled treatment and this is consistent with lower female survival in these areas from 1995–1998 (Esler et al. 2000a). Differences in sex ratios may reflect long-term and not necessarily recent differences in survival. The differences we observed in sex ratios may reflect lack of complete recovery from past injury as we observed no differences in density change or recruitment between oiled and unoiled areas. Sex ratios we observed in unoiled treatments were consistent with observations for other populations of harlequin ducks in coastal British Columbia (Smith et al. 2001, Rodway et al. 2003).

Without immigration, recovery is dependent upon recruitment exceeding spill related mortality. We used age ratios as an index of recruitment. Differences in age ratios may indicate recent differences in breeding propensity, breeding success, or immature survival between oiled and unoiled populations. The recruitment pattern we observed appears consistent for K-selected species that have low rates of annual recruitment and relatively low and variable breeding propensity (Goudie et al. 1994) and was sufficient to maintain a stable population. We observed an annual increase in recruitment but not as a function of treatment. As oil was still present on Prince William Sound beaches during our study (Short et al. 2004), remaining oil does not appear to be affecting recruitment. The age ratios we observed were consistent with observations for other populations of harlequin ducks in British Columbia (Smith et al. 2001, Rodway et al. 2003).

From 1997–2005, differences in slopes between oiled and unoiled areas decreased from our 1995–1997 fall surveys (Rosenberg and Petrula 1998). We also observed a reversal in trends from these earlier surveys, going from a negative slope to a neutral slope in oiled areas. This smaller and non-significant difference in slopes between treatments and stable trend in oiled areas from 1997–2005 applies when comparing all oiled and unoiled areas or just western Prince William Sound with eastern Prince William Sound.

We failed to reject the null hypothesis of no significant difference in the mean rate of change in harlequin densities between treatments during the 1997–2005 survey period. This is a positive sign and provides evidence of recovery (lack of injury) although we still lack conclusive evidence for a population increase in oiled areas. Populations in both treatments were stable whether comparing our two original study areas only (eastern Prince William Sound and western Prince William Sound) or all four study areas. These results differ from our 1995–1997 surveys when we attributed a negative population trend for harlequin ducks in oiled areas as a continuing oil spill affect (Rosenberg and Petrula 1998).

Natural geographic variation within PWS is not affecting population change disproportionately at the spatial scale we used to define our regions, areas and treatments. The amount of variability in our surveys limits our ability to detect subtle differences.

The outlook for full recovery is good. Populations in the oiled area are stable, age ratios are similar between treatments, oil exposure rates have declined, and female survival has improved. However, the persistence of bioactive *EVO* in intertidal sediments, continued exposure to hydrocarbons, proportionately fewer females in oiled areas, and lack of evidence for a significant population increase in oiled areas provides a basis for continued population-level effects. More information is needed to address the relevancy of these observation to lingering spill effects or other biological mechanisms.

INTRODUCTION

This study was initiated to determine whether the harlequin duck population in oiled areas of Prince William Sound (PWS) recovered or is in the process of recovering from the effects of the 1989 *T/V Exxon Valdez* oil spill (EVOS). Continued oil exposure was the most likely mechanism constraining full population recovery through 1998 (Esler et al. 2002). We compared population structure (age and sex ratios) and trends between oiled and unoiled areas in PWS with data gathered during four annual winter (March) surveys beginning in 1997 and continuing with expanded geographic coverage in 2000–2002, and 2004–2005.

Harlequin ducks *(Histrionicus histrionicus)*, a sea duck *(Mergini*), occur year-round in PWS (Isleib and Kessel 1973) and were the most abundant waterfowl species in nearshore habitats prior to the EVOS (Irons et al. 1988). Winter population estimates (±95%CI) for the entire PWS have ranged from a high of 19,204 (±4,515) harlequin ducks in 1994 (Sullivan et al. 2006) to a low of 10,629 (± 2,544) ducks in March 1990, a year after the spill (Klosiewski and Laing 1994). The only winter surveys prior to EVOS were conducted in 1972 and 1973 (Klosiewski and Laing 1994). More recent winter population estimates are 14,876 (±3,288) and 13,174 (±2,994) ducks in 2000 and 2004 respectively (Sullivan et al. 2005).

On March 24, 1989, the *T/V Exxon Valdez* ran aground in northern PWS spilling approximately 42 million liters of crude oil. Oil drifted southwest, oiling hundreds of kilometers of PWS beaches (Galt et al. 1991, Piper 1993). Within oiled areas, the entire harlequin duck wintering population was at risk of exposure because the EVOS occurred prior to movements to breeding areas. Post-spill studies estimated that between 420 and 1838 harlequin ducks died in PWS as a direct result of the EVOS (Ecological Consulting, Inc. 1991, Piatt and Ford 1996, J.F. Piatt, pers. comm.).

Direct and indirect mortality to seabirds from oil spills is manifested in two primary ways: (1) changes in population size or structure (e.g., changes in sex and age ratios), which are in turn related to oil induced effects on reproductive rate and recruitment, and (2) less available habitat supporting fewer birds (Wiens 1995). A reduction in prey or indirect exposure (ingestion of contaminated foods) may further increase adult mortality or reduce productivity. Oil spills may affect demography (e.g., age structure, birth rates, individual growth rates) and alter population trajectories without affecting species abundance (Paine et al. 1996).

Harlequin ducks are particularly vulnerable to oil spills because (1) they exhibit strong philopatry to nearshore (marine) molting and wintering areas (Robertson et al. 1999, Robertson et al. 2000, Cooke et al. 2000), (2) they utilize intertidal and shallow subtidal zones exclusively for foraging for invertebrates (Dzinbal and Jarvis 1982), and (3) their nearshore habitats were subjected to the most severe and persistent effects of oiling (Highsmith et al. 1996, Short and Babcock 1996).

A significant decline in numbers potentially predisposes a population of sea ducks to a relatively long recovery period. Strong philopatry has adaptive advantages in relatively stable marine environments (Robertson et al. 2000), but following an oil spill it has the disadvantage of subjecting birds to lingering oil year after year. This potentially increases chronic and cumulative effects that may result from direct or indirect exposure (Esler et al. 2000b). Further, with limited dispersal to new wintering areas, recovery must occur primarily through production and recruitment (Esler et al. 2000b).

Relative to dabbling (*Anatini*) and diving (*Aythyini*) ducks, sea ducks are considered *K* selected species because: (1) they occupy relatively stable environments (2) first breeding occurs later than 1 year of age; and (3) their life history is characterized by (a) low rates of annual recruitment, (b) high adult survival, and (c) relatively low and variable breeding propensity. Sea ducks are sensitive to catastrophic causes of mortality because long-term population stability depends on high adult survival (see Goudie et al. 1994). Consequently, full recovery may be delayed beyond the period when spill effects are no longer detectable (Esler et al. 2000b).

Several post-spill surveys and damage assessment studies were designed to measure the extent and severity of injuries to the PWS harlequin duck population from the EVOS and assess recovery (see Esler et al. 2002 for a review) and oil spill effects were still evident through 1998 (Esler et al. 2002). Although injury to PWS harlequin ducks from the spill was well documented the extent and magnitude of the injury remains controversial (Wiens et al., in press).

Recently prior or coincidental to this study 1) invertebrate recovery in upper intertidal and subtidal areas remained incomplete for some taxa (Hooten and Highsmith 1996, Jewett et al. 1999, Peterson 2001); 2) oil persisted in mussel beds (Carls et al. 2001) where it had been identified as a source of contamination for benthic invertebrates (Harris et al. 1996); 3) lingering surface oil (Hayes and Michel 1999) maintained the possibility of external oiling of feathers and resultant metabolic consequences (Trust et al. 2000); 4) cytochrome P4501A (CYP1A) induction was greater in tissues of harlequin ducks captured in oiled areas than in reference areas (Trust et al. 2000); and 5) overwinter female survival was lower in oiled than reference areas (Esler et al. 2000a). Observed differences between treatments in winter survival (Esler et al. 2000a) and population trends (Rosenberg and Petrula 1998) were linked to observed differences in contaminant exposure (Esler et al. 2002). Collectively, these studies supported the conclusion that harlequin duck populations had not recovered from the spill as of 1998.

More recent studies indicated improving conditions. Measurements of CYP1A levels and female survival rates were converging between oiled and unoiled areas during the period from 2000–2002 (Bodkin et al. 2004). However, lingering oil still remained in the environment maintaining the possibility of continued exposure and chronic effects to wildlife (Short et al. 2004).

In 1997, we transitioned from summer and fall surveys (Rosenberg and Petrula 1998) to winter surveys because winter is a period of maximum population stability. Annually, the numbers of harlequin ducks in PWS declines from early to late spring as breeding pairs depart for nesting areas (Rosenberg and Petrula 1998). Numbers increase as non-breeding and post-breeding males return to the coast followed by non- or failed breeding females. Finally, successful females return with broods in late summer and early fall. Winter site-fidelity is prevalent (Robertson et al. 1999, Cooke et al. 2000, Robertson et al. 2000) and harlequin ducks on wintering areas are thought to constitute demographically independent subpopulations (Cooke et al. 2000). Once settling at the wintering area individual ducks rarely move more than a few kilometers from a given section of shoreline until the following spring (Robertson et al. 2000, D.H. Rosenberg and M.J. Petrula, unpubl. data, Iverson and Esler, in press).

OBJECTIVES

1. Compare annual changes in density and population structure (immatures, adult males, and females) between oiled and unoiled areas.

2. Compare annual changes in density and population structure *within* oiled and unoiled areas.

We hypothesized that the population structure would be similar between oiled and unoiled areas and trends in the oiled area would be positive and increasing at an equal or greater rate than in the unoiled area if the oiled population had recovered or was in the process of recovering. We used age and sex composition as parameters to test whether harlequin ducks in oiled and unoiled areas of PWS exhibited similar demographic characteristics. We used annual changes in density of harlequin ducks to compare population trends between oiled and unoiled areas.

STUDY AREA AND METHODS

The study was conducted in Prince William Sound (PWS) (ca. 60°30'N, 147°00'W), a marine water body located on the southcentral coast of Alaska (Fig. 1). PWS is a large estuarine embayment of the northern Gulf of Alaska characterized by fjord-like ports and bays surrounded by steeply rising mountains. Highly irregular in shape, it is approximately 160 km east to west and 140 km north to south. Tides can exceed 4.5m and water depth can reach 870m. Total shoreline (including islands) is approximately 5,000 km (Irons et al. 1988). A general description of the physiography, climate, oceanography, and avian habitats of PWS was described by Isleib and Kessel (1973).

After running aground on Bligh Reef in northern PWS, *T/V Exxon Valdez* oil spread southwest, oiling 783 km of shoreline in PWS before spreading to the Gulf of Alaska (Galt et al. 1991, Piper 1993, Neff et al. 1995) (Fig. 1).

Survey Design

In 1997 we surveyed harlequin ducks in areas of western Prince William Sound (WPWS) oiled by the EVOS and in unoiled areas of eastern Prince William Sound (EPWS). The two survey areas were geographically separate (Fig. 1). From 2000–2005 we included two more geographically separate survey areas: an oiled area in southwestern Prince William Sound (SWPWS) and an unoiled area in nearshore waters of northwestern Montague Island (MONT) (Fig. 1). The SWPWS and MONT survey areas were added in 2000 to broaden the geographic scope and sample size of the study. Oiled and unoiled transects were separated by a minimum of ca. 8 km to a maximum of ca. 150 km. We repeated surveys of the same transects in successive years in both the oiled and unoiled areas.

Conventional aerial and boat surveys do a poor job detecting harlequin ducks (Savard 1989, Breault and Savard 1999); therefore dedicated census techniques are necessary (Rodway et al. 2003). We chose a species specific approach that allowed us to design a survey with a large sample size over a broad geographic area, incorporating the unique life history, behavior and habitat utilization of harlequin ducks. This was in contrast to several post-spill bird studies that designed surveys using a multi-species approach (Day et al. 1997, Murphy et al. 1999, Irons et al. 2000, Lance et al. 2001).

We designed our sampling to include a large proportion of the ducks within the spill region and a comparable sample size in unoiled areas of PWS geographically separate from the spill region. Time-series baseline data (pre–spill) did not exist for harlequin duck populations in PWS. Only two winter surveys (counts only) were conducted 16–17 years before the EVOS. This lack of recent pre-spill data on population structure, numbers of wintering ducks, and concerns about the effects of natural variation on population estimates, precluded the use of a before-after-control-impact (BACI) design (Esler et al. 2002).

Neither harlequin ducks nor oil was uniformly distributed. We attempted to incorporate all aspects of harlequin duck habitat attributes (Esler at al. 2000, Rodway et al. 2003) within areas of low to high densities in both treatments (oiled and unoiled). Harlequin ducks respond to small-scale variations in habitat attributes (Esler at al. 2000b) resulting in a patchy rather than uniform distribution throughout PWS. We selected a large sample size of ducks distributed in a variety of habitats dispersed over a broad geographic area

We also attempted to sample over the full range of oil exposure. Oil was distributed in a mosaic pattern resulting in varying concentrations from none to heavy (Neff et al. 1995). However, a significant portion of each transect in the spill region had some degree of oiling history, ranging from light and patchy to heavy and continuous (Alaska Dept. of Environmental Conservation, Oil Spill Response Center 1990) and all transects supported ducks.

We assumed similar temporal changes in demographics in both oiled and unoiled areas in the absence of the oil spill and we assumed no movement of ducks between oiled and unoiled areas. Male and female harlequin ducks exhibit high within-year site-fidelity to relatively small geographic regions (see above) so ducks are unlikely to move great distances throughout the winter reducing the probability of interchange between study areas (Robertson et al. 2000). Both males (paired and unpaired) and females exhibit similar annual return rates to wintering areas (Robertson et al. 2000) reducing the probability of interchange between years.

The oil spill was concentrated in just a portion of PWS making it difficult to intersperse or randomize samples from oiled and unoiled areas (Paine et al. 1996). Thus, geographic variation between treatments may introduce variables that affect demographics unrelated to oil. We compared two geographically separate study areas within each treatment in an effort to segregate effects of oiling from spatial variance due to unique environmental factors (Wiens and Parker 1995).

Age ratios serve as an index of recruitment and can be used in the field for wintering harlequin ducks. Plumages of first winter males are distinct from older males and females (Smith et al. 2001). This allowed us to measure recruitment by counting immatures.

We assumed all birds in the spill region would not be equally affected by oil, but the population structure and growth rates we observed would be different from a reference area if injury persisted. With time, as oil weathers and exposure decreases we should observe a convergence in population trends and structure between oiled and unoiled areas.

Survey Methods

All transects were surveyed within a 2–week period beginning in late March (Table 1). We counted all ducks on the water or hauled out on rocks within 200m of the tide line. Transects included shoreline and nearshore habitats including beaches and concomitant offshore rocks, islets, and islands. Birds flying towards the boat (opposite the direction of our travel) were counted but birds flying from behind the boat (in direction of our travel) were not counted. Surveys were conducted from open skiffs (ca. 6m long) traveling at 2-10 km/hr within 100 meters of shore (tide line) at a pace, course, and distance that assured complete coverage of the survey area. To improve observations boats often came to a complete stop.

Two skiffs worked simultaneously on different transects or on few occasions different portions of the same transect. This included circling all exposed rocks, and scanning shallow lagoons from shore when boat travel was not possible. Boating distance from shore depended on habitat, light, weather, and tide conditions. One full-time observer and an observer/boat operator continuously surveyed nearshore habitats using 10X binoculars. When possible large flocks of resting ducks were observed from vantage points on shore using a 20X-60X spotting scope.

Surveys were not conducted when wave height, precipitation, or light conditions compromised accuracy. When intermittent weather changes during the course of a survey affected visibility it was often impractical to abort the entire transect. In these conditions birds were counted but classified as unknown sex and age. We assumed equal proportions of ducks (sex, age) in unclassified flocks as we documented in classified flocks. Most unclassified birds consisted of entire flocks that flushed prior to sampling or could not be classified due to poor visibility. In a minority of cases we randomly sub-sampled flocks prior to flushing.

During all surveys, we recorded the number, sex, and age of all harlequin ducks observed in each flock, and the location of the flock (GPS coordinates). We also marked flock locations on nautical charts (National Ocean and Atmospheric Administration).

We classified birds as female, adult male, or immature male based on plumage patterns (Smith et al. 1998). Immature referred to birds in their first year of life but in their second calendar year (e.g., hatched in July 2000 and observed in March 2001). The alternate I plumage of immatures varies among individuals from mostly female-like to mostly adult male-like but each individual retains a similar pattern throughout the winter (Smith et al. 1998, Smith et al. 2001, Rodway and Regehr, unpubl. data). We used “immature” synonymously with “sub-adult” (Rosenberg and Petrula 1998) and “first winter” birds (Rodway et al. 2003). Adult birds are at minimum, in their “second winter” or third calendar year. Immature females could not be visually differentiated from adults in the field. Harlequin ducks not identified to sex were recorded as unclassified.

Survey Coverage

Shoreline length (km) of transects was calculated from the Alaska Department of Natural Resources PWS_ESI ARC/INFO GIS database. Shoreline length of small islands not included in the PWS_ESI ARC/INFO GIS database was calculated using the U.S. Forest Service CNFSHORE ARC/INFO GIS database (Figs. 2–3, Appendix A).

We surveyed 2 study areas (EPWS, WPWS) in 1997 and 4 study areas (EPWS, WPWS, SWPWS, MONT) in 2000, 2001, 2002, 2004 and 2005; consequently we surveyed less shoreline in 1997 (550.3 km) than in later years (ca. 745.9 km) (Table 1). Variation in survey coverage within study areas existed among years because, on occasion, poor weather precluded the completion of some (or portions of) transects (Table 1, Appendix B).

In WPWS, transects were established in selected areas extending from the north end of Culross Island, south to Dangerous Passage, southeast to Squire Island, and east to Green Island. Additional surveys in oiled portions of SWPWS were established along the shorelines of Bainbridge, Evans, and LaTouche islands (Fig. 2). We surveyed ca. 444 km of the approximately 783 km of western and southwestern PWS oiled by the spill (Neff et al. 1995). The actual length of oiled shoreline we surveyed was slightly less because some transects contained unoiled portions. In the oil spill region more transects with low densities were selected than in unoiled regions. This was necessary to get a sufficient number and distribution of ducks.

Transects located in the EPWS study area included portions of Hinchinbrook Island, Sheep Bay, Port Gravina, Landlocked Bay, Bligh and Busby islands, Galena Bay and Valdez Arm in northeastern PWS (Fig. 3). In 2000, we added a study area (one transect) along the shoreline of northwestern Montague Island (MONT) (Fig. 2). In 2004 we added two transects to our MONT study area (Fig. 2).

We selected more transect locations in EPWS (n = 22) than WPWS (n = 18), SWPWS (n = 4) and MONT (n = 3), but total shoreline length was greatest in WPWS (Table 1). Transect length varied (range = 1 to >70 km) (Appendix A) and averaged 16.7 km (SD = 19.6) in WPWS; 10.0 km (SD = 7.5) in EPWS; 37.2 km (SD = 23.5) in SWPWS and 33.4 km in MONT (SD = 35.1).

Statistical Methods

Sex and Age Structure

We used a generalized logit model (natural logarithm of ratios) (Agresti 1990) to test for annual differences among study areas (WPWS, SWPWS, EPWS, MONT) and between treatment (oiled) and reference (unoiled) areas for the following sex and age ratios: (1) males to females; (2) adult males to immature males and 3) adult females to immatures (both sexes). Model fit was assessed using AIC and a backward elimination process. At each step a reduced model was used to test for significant year, area, or treatment effect (Agresti 1990). Such a criterion allows for optimal fitting of the data without over-parameterizing the model. The SAS model used the GLIMMIX Procedure with a binomial distribution and a logit link function.

The full model was (using the sex ratio as an example):

$$\ln\left(\frac{m_{1jkl}}{m_{2jkl}}\right)=\alpha+\beta*year+\tau_k+\tau_{k(l)}+\gamma_{k(l)}*year$$

Where m is the expected number of birds counted;

sex is indexed by number (1 = male, 2 = female);

j indexes year (1 = 1997, 2 = 2000, 3 = 2001, 4 = 2002, 5 = 2004, 6 = 2005);
k indexes treatment (1 = oiled, 2 = unoiled); and
l indexes regions (EPWS1-3, MONT, SWPWS, WPWS1-4) within a treatment.

Proc GLIMMIX also allowed us to create a more complex covariance structure that accounted for the correlation found in measuring the same transects over multiple years. This reduces the occurrence of Type I errors since the variance is more appropriately modeled and not underestimated.

We doubled the number of immature males to estimate the total number of immature birds in the population. We assumed the number of immature males equals the number of immature females because (1) juvenile sex ratios are similar on the breeding grounds (Ashley 1998); (2) fledged broods migrate with adults to the wintering areas (Smith 2000, Regehr et al. 2001); and (3) adult males and females exhibit similar winter survival rates (Cooke et al. 2000). The number of adult females was calculated by subtracting the number of immature males we observed (which equal immature females) from total females.

Harlequin ducks not identified by age and sex (unknowns) were not included in the ratio analysis. We did not adjust our counts to compensate for variation in survey coverage among years because we used relative measures of abundance.

Trend Analysis

Transect observations were modeled as Poisson counts weighted by the length of the transect. We standardized all counts of birds to linear densities (birds/km of shoreline surveyed) to facilitate comparisons in change in densities among regions and between treatments. Proc GLIMMIX was used, this time using a Poisson distribution with a log link function.
The full model was :

$$n_{jkl} = \alpha_k + \beta_k * year + \tau_{k(l)} + \gamma_{k(l)} * year$$

Where n is the expected number of birds counted;
j indexes year (1 = 1997, 2 = 2000, 3 = 2001, 4 = 2002, 5 = 2004, 6 = 2005);
k indexes treatment (1 = oiled, 2 = unoiled); and
l indexes regions (EPWS1-3, MONT, SWPWS, WPWS1-4) within a treatment.

Because the sampling scheme was not appropriate for comparing overall measures of abundance among regions we wanted to essentially model the two treatments separately, including estimating difference variance components for each treatment. As in the ratio analyses, proc GLIMMIX also allowed us to account for the correlation found in measuring the same transects over multiple years. Using this model eliminates the need for a power analysis because we are directly modeling a slope instead of evaluating a sample of slopes from each transect as was previously done (Rosenberg and Petrula 1998).

We analyzed our data in a nested model at four spatial scales: (1) transect, (2) region, (3) area and (4) treatment. Our four areas, also referred to as ‘study areas’ were composed of two oiled areas, WPWS and SWPWS, which collectively formed the oiled treatment and two unoiled reference areas EPWS and MONT which collectively formed the unoiled treatment (Figs. 1-3). Southwestern

PWS and MONT were composed of one region each, and thus region and area were synonymous. EPWS was composed of three regions and WPWS was composed of four regions. Regions consisted of groups of transects in close geographic proximity (Appendix A).

RESULTS

Abundance and Distribution

In 1997 we counted 2,860 harlequin ducks over 528.9km of shoreline in WPWS and EPWS (Table 1). In the following years we counted an average of 4,964.8 (SD = 147.0) ducks annually over an average of 749 km of shoreline with an average density of 6.6 ducks/km (Table 2). Since 2000, numbers ranged from a low of 4,823 ducks in 2000 to a high of 5,186 in 2005. In addition we recorded an average of 409 ducks along 26.3 km (15.6 ducks/km) of additional transects surveyed on MONT in 2004 and 2005 (Tables 1, 2).

Within each study area birds were not uniformly distributed among regions or transects. Some transects consistently supported more birds than others (Appendix B, D-F). Birds in SWPWS and MONT were more evenly distributed than WPWS or EPWS, in large part, due to smaller survey areas. Annually, numbers, densities, and distribution were relatively consistent.

Population Structure

Sex Ratios

Sex ratios were significantly different between treatments ($t = 2.35$, $P = .022$, $DF = 62$), but not among years or areas (Table 3.). Nonsignificant main effects and interaction terms were dropped from the model in a stepwise descending order. We observed a greater proportion of males in oiled treatments than unoiled treatments (Table 3, Fig. 4). Differences in sex ratios were not attributed to regional effects ($f = 0.88$, $P = .529$, $DF = 7/55$). We found similar results when we compared WPWS with EPWS only. We found a significant difference between treatments and would reject the null hypothesis ($t = 2.21$, $P = .031$, $DF = 55$).

Sex ratios were skewed towards males in all years, areas, and treatments (Tables 2 and 3, Fig. 4). Combining all years, we observed 40.2 (95% CI: 39.5 – 41.0) females per 100 birds in oiled areas and 41.8 (95% CI: 40.9 – 42.8) females per 100 birds in unoiled areas. By area, the lowest percentage of females was 38.4 recorded in WPWS in 2004 and the highest was 44.2 % recorded in SWPWS in 2001 (Fig. 4).

Age Composition

Differences in ratios of adult males to immature males were best explained by a year effect and by regional variation and not by a treatment effect ($t = -0.68$, $P = 0.502$, $DF = 55$) (Table 4). The variability between areas within treatments (WPWS vs. SWPWS and EPWS vs. MONT) made it difficult to detect a treatment effect. Annually, for PWS, the ratio (adult male/immature male) declined slightly (increase in immatures relative to adults) by 5.38% (95% CI: 2.26 to 8.41).

We observed 13.66 adult males for each immature male (7.3% immature males) over the entire study area (Table 3). Immature males comprised the greatest proportion (lowest ratios) of the male population in 1997 (8.8%) and the lowest proportion in 2001 (4.7%) (Table 3, Fig. 5).

Plumage patterns of immatures do not change as winter progresses (Smith et al. 2001). This makes it equally likely to distinguish an individual immature from a female or adult male in early or late winter allowing us to compare our data with data collected throughout the winter in coastal British Columbia (BC) (Smith et al. 2001). To enable comparisons we adjusted our ratios to reflect the number of immature males to *total* males. For the six years of surveys, the mean ratio of immature males (to *total* males) was 6.8% or 14.7 males: 1 immature.

We also compared the ratio of adult females to immatures (both sexes) (Table 3, Fig. 6). The patterns we observed were similar to those for male age ratios (above). Ratios differed significantly among years (year effect) and by regions within treatments but not by treatment ($t = -1.28$, $P = 0.205$, $DF = 55$) (Table 4). We also observed a positive trend in recruitment as the ratio of adult females/immature declined annually by 6.31% (95% CI: 3.00 to 9.51).

We observed 4.59 adult females for each immature (21.8% immatures) over the entire study (Table 3). Immatures comprised the greatest proportion (lowest ratios) of the adult female population in 1997 (26.5%) and the least proportion in 2001 (13.4%) (Table 3, Fig. 6).

We found no differences between treatments in age ratios when we compared our two original study areas, WPWS and EPWS, only. Differences in ratios of adult males to immature males were best explained by a year effect and by regional variation and not by a treatment effect ($t = -0.27$, $P = 0.791$, $DF = 55$). The ratio of adult females to immatures ($t = -0.37$, $P = 0.710$, $DF = 55$) exhibited a similar pattern.

Population Trends

We found no significant difference in the change in density (trends) between oiled and unoiled treatments ($t = -0.30$, $P = 0.761$, $DF = 193.9$) and the mean rate of change for oiled areas (0.0125, $P = 0.138$) and unoiled areas (0.0186, $P = 0.304$) was not significantly different from zero. We found similar results when comparing WPWS and EPWS only ($t = -0.68$, $P = 0.497$, $DF = 185.4$). Change in density followed similar patterns when comparing males and females separately. We found no significant differences in change in density for males ($t = -0.53$, $P = 0.598$, $DF = 210.9$) or females ($t = -0.82$, $P = 0.411$, $DF = 222.2$) between treatments.

Within treatments we found no significant differences in the change in densities when comparing regions within SWPWS and WPWS ($t = 0.17$, $P = 0.867$, $DF = 136.5$) or MONT with EPWS ($t = -1.02$, $P = 0.311$, $DF = 134.8$). Among regions, only region 3 in WPWS (Appendix A), which contains 46.1 % of the ducks within our WPWS study area (Appendix B) was significantly different from zero (mean rate of change = 0.031, $P = 0.029$).

DISCUSSION

The EVOS Trustee Council (2002) defined recovery as a return to prespill demographics and similar levels of hydrocarbon exposure in ducks from oiled and unoiled areas (treatments). Without good pre-spill data we were limited to measuring recovery in relative terms by comparing changes in abundance and composition between treatments within PWS. Similar age and sex composition between treatments and numbers of ducks in oiled areas increasing at an equal or greater rate than in unoiled areas would indicate that the harlequin population is recovering from the effects of the EVOS. We would interpret differences in these demographic parameters as evidence of continuing injury. Once recovered, demographic parameters would converge and our two treatments would exhibit parallel changes (Wiens and Parker 1995).

We did not conduct winter surveys until 1997 (8 years post-spill) and population demographics are expected to change in value from year to year due to annual changes in the environment. We assumed these temporal changes would affect all birds similarly in the absence of oil, regardless of location. We tested this by comparing population demographics at separate geographic areas within each treatment. The more temporally distant from the spill, the more likely our results may include a mixture of impact and recovery, as well as environmental changes that may mimic an impact or recovery or mask impacts that did occur or are still occurring (Wiens and Parker 1995).

Assessing injury and recovery in natural systems following an oil spill is fraught with complexities (Paine et al. 1996). By choosing a null hypothesis that represents a liberal definition of recovery any evidence of injury is a concern from a resource management perspective. Regardless, population trend and structure data is best viewed in context with physiological, behavioral, habitat, and life history data. This provides a more thorough assessment of recovery than using any one study or parameter independently. Thus, the demographic data we present serves as just one component, albeit an important one, of the recovery objectives (*Exxon Valdez* Oil Spill Trustee Council 2002).

Abundance and Distribution

In general, we observed similarities in distribution and habitat use from year to year. No regions or transects were abandoned and we were not aware of flocks inhabiting new areas with any consistency. Annually, harlequin ducks observed during our surveys likely represent many of the same birds as many individuals exhibit winter philopatry to relatively small geographic areas (Robertson et al. 2000, D. Esler, pers. comm.). We also believe little regional movement or interchange occurred during each of our 2–week survey periods. March is a period when pair bonds are well formed (Robertson et al. 1998) and there is relative geographic stability in both numbers and distribution (Breault and Savard 1999, Cooke et al. 2000). Specifically in PWS, no regional movements or interchange between our study areas occurred in March or early April by males (D.H. Rosenberg and M.J. Petrula, unpubl. data) or females (Iverson and Esler, in press) that would have biased our surveys.

Population Structure

Sex Ratios

Sex ratios skewed toward males are typical for sea ducks (Goudie et al. 1994). The sex ratios we observed in unoiled areas, 41.8 females per 100 birds were identical with sex ratios reported for harlequin ducks wintering in BC (Smith et al. 2001), (Fig. 7) and slightly greater than those reported by Rodway et al. (2003), who observed 39.9 females per 100 harlequin ducks wintering in BC. The sex ratios we observed in unoiled areas (40.2 females per 100 birds) were lower than reported by Smith et al. (2001), but slightly greater than reported by Rodway et al. (2003). We used harlequin ducks wintering in BC for comparison because they represent a geographically separate population on the Pacific coast not affected by a major oil spill.

With an oil spill we would expect different survival rates between males and females if oil, directly or indirectly, had a more pronounced effect on either sex. Harlequin ducks may live near the edge of an energetic threshold during winter (Goudie and Ankney 1986). Altered physiological processes resulting from oil exposure may be more detrimental to smaller females, which in winter spend more time feeding than males (Fischer and Griffin 2000). This would likely be more pronounced at the northern edge of their range due to a harsher climate and less hours of daylight. Consequently, we would expect sex ratios to be even more biased towards males in oiled areas due to greater female mortality.

We observed a lower proportion of females in the oiled treatment. This is consistent with lower female survival in these same areas of PWS from 1995–1998 (Esler et al. 2000a). More recent data from PWS indicate female survival rates in oiled areas are improving relative to unoiled areas (Bodkin et al. 2004). Differences in sex ratios may reflect long-term and not necessarily recent differences in survival. The differences we observed in sex ratios may reflect lack of complete recovery from past injury as we observed no differences in density change or recruitment between oiled and unoiled areas. Rodway et al. (2003) observed higher sex ratios (more males) further offshore. While we did not observe similar differences in male and female distribution, other factors affecting distribution could have an influence on sex ratios (see below *Variation in Counts*). We do not know the biological significance, if any, of the difference we observed.

Age Composition

Without immigration, recovery is dependent upon recruitment exceeding spill related mortality. The number of recruits is the product of the breeding population and the recruitment rate (Cowardin and Johnson 1979) both of which can be affected by many factors. We used age ratios as an index of recruitment. Differences in age ratios may indicate recent differences in breeding propensity, breeding success, or immature survival between oiled and unoiled populations.

Age ratios in PWS varied by year and location (Figs. 5, 6). We observed no significant difference in age ratios between treatments so we compared PWS age ratios collectively with those reported from BC. The mean age ratio (immature males:total males) we observed for PWS was identical with observations for wintering harlequin ducks in BC (Smith et al. 2001), (Fig. 8). Rodway et al. (2003) reported a male age ratio (immature males:adult males, unadjusted for survey bias) of 8.4 %, which is greater than our mean (7.3%) but within our annual range (4.7%–8.8%). The authors (Smith et al. 2001, Rodway et al. 2003) may have identified more immatures because they observed birds primarily from land while we used boats primarily (Rodway et al. 2003, see below).

Annual variation in recruitment is usual in waterfowl. The recruitment pattern we observed appears consistent for K-selected species that have low rates of annual recruitment and relatively low and variable breeding propensity (Goudie et al. 1994) and was sufficient to maintain a stable population.

We would expect oil exposure rates to decrease as oil weathers and becomes less available following a spill (see Wiens and Parker 1995). If oil exposure caused a decline in recruitment we would expect recruitment to increase as exposure rates declined. We observed an annual increase in recruitment but not as a function of treatment. As oil was still present on PWS beaches during our study (Short et al. 2004), remaining oil does not appear to be affecting recruitment.

We do not know the reasons for the within–year geographic variation we observed, although it may be a function of bird mobility and fewer transects and a smaller survey area in SWPWS and MONT. Habitat segregation by age or sex does not appear to occur in wintering areas in BC (Rodway et al. 2003) and probably is not a factor. Additional surveys will help confirm if this is a persistent pattern resulting from unequal geographic distribution, annual variation, or sampling bias (see below). We do not know if immigration or emigration is a factor.

While age ratios serve as an index of recruitment, they may not be an accurate measure of recruitment. Rodway et al. (2003) found boat surveys, when compared with land based surveys, slightly underestimated recruitment. It is easier to confuse an immature male with an adult male or female than vice-versa and correct identification becomes increasingly difficult from a boat as weather or light conditions deteriorate. Observer experience is also a factor in correctly identifying immatures. Further, the actual rate of recruitment into the breeding population is affected in years prior to breeding by attrition in each cohort. Not all female harlequin ducks breed in their second year and males may delay breeding for several years (Robertson and Goudie 1999). We believe our observations approximate recruitment rates of a cohort at the end of the first winter, but regardless, we are most concerned with comparisons of relative values between treatments.

Female age ratios reveal more about productivity and recruitment than male age ratios and ideally we would use these ratios to compare recruitment. With male biased sex ratios population growth rates are female limited (Goudie et al. 1994) and in general, dispersal rates of females may be less variable than unpaired males which may travel in search of mates. The drawback to using this procedure is that we cannot directly measure female age ratios in the field and must assume equal proportions of immature males and females (see Statistical Methods above).

Population Trends

Interpreting the results of trend data is often difficult due to natural temporal and spatial variation inherent in wildlife populations (Wiens and Parker 1995, Paine et al. 1996). One advantage with interpreting temporal change for harlequin ducks may be the lack of evidence for population change following recent climate change (Agler et al. 1999).

We failed to reject the null hypothesis of no significant difference in the mean rate of change in harlequin densities between treatments during the 1997–2005 survey period. This is a positive sign and provides evidence of recovery (lack of injury). Populations in both treatments were stable whether comparing our two original study areas only (EPWS and WPWS) or all four areas. This demonstrates a divergence from our 1995–1997 surveys when we attributed a negative population trend for harlequin ducks in oiled areas as a continuing oil spill effect (Rosenberg and Petrula 1998).

To declare recovery in oiled treatments, Lance et al. (2001) required both a positive trend and densities increasing at a significantly greater rate than unoiled areas. Although populations in the oiled treatment are no longer declining, we have not measured a positive growth rate nor are growth rates in oiled areas exceeding growth rates in unoiled areas. This is consistent with USFWS observations for wintering harlequin ducks in PWS from 1990–2000. In the USFWS surveys, oiled areas relative to unoiled areas of PWS did not show any evidence of a recovering population using a homogeneity of slopes test on winter densities. However when using a regression analysis of density data they did observe an annual increase (Stephensen et al. 2001). This pattern continued through 2004 (D. Irons, pers. comm.).

Comparisons of our late-summer surveys (1995–1997) with winter surveys (1997–2005) are not fully comparable due to seasonal differences in distribution and abundance (Rosenberg and Petrula 1998). We switched to winter surveys because winter is a period of minimum mobility and thus maximum stability in both numbers and composition of harlequin ducks. During late-summer and early fall (July to September) there is differential temporal movement from breeding and molting areas to wintering areas by sex, age, and breeding status. Further, from 1995–1997 we had just three years of surveys.

Our negative population trend from 1995–1997 was supported by lower winter survival rates (1996–1998) and elevated levels of CYP1A among female harlequin ducks (1998) in oiled treatments of PWS (Esler et al. 2000, Trust et al. 2000). Our present results are consistent with improving female survival and declining levels of CYP1A in oiled treatments (Ballachey et al. 2006) and demonstrate improving conditions for harlequin ducks. However, oil exposure is still a concern. Oil is still bioavailable in intertidal habitats (Short et al. 2005), benthic invertebrates may be affected by exposure (Day 2005), female ducks from oiled areas have significantly greater exposure rates than those from unoiled areas (Ballachey et al. 2006), and we observed a lower proportion of females in the oiled treatment. In addition, we have not observed an increase in abundance in oiled areas. Chronic oil exposure may be suppressing population growth and the ability of populations to return to historical levels.

Variation in Counts

Actual differences between years in abundance and composition are related to variation in productivity, mortality, and rates of immigration and emigration. We believe the variability we observed is a combination of these plus avian mobility and sampling errors.

Measurement error may have contributed to some variation in our counts. Generally, we believe any variability from measurement error is minimal and unbiased as it pertains equally to all transects. The same observers participated in surveys, surveys were conducted at the same time each year and transects were thoroughly searched in an effort to minimize measurement error. Weather (including light) conditions likely caused some error because our ability to correctly identify immature birds improved with better weather and it was impossible to survey all transects in identical conditions.

More unclassified birds occurred in unoiled areas than oiled areas primarily due to higher densities and intermittent weather conditions. Unclassified birds represented a small proportion of the total sample size and due to our methodology (see Methods) we do not believe it biased sex and age ratios.

We do not believe small areas of herring spawn ("spot spawn") in EPWS biased our survey results. Annually, in the EPWS study area we encountered herring spawn. Spawn always occurred in our survey area (on a transect) except for one occasion when it was adjacent to a transect. This increased transect variability within EPWS but we saw no evidence that it attracted large numbers of harlequin ducks from outside the EPWS study area. Herring spawn has been recorded as early as 1 March in PWS (S. Moffitt, ADF&G, pers. comm.) but telemetry studies indicated no movements of males or females to spawning events in March or early April (see Abundance and Distribution).

Birds are mobile and although telemetry data shows little interchange between oiled and unoiled areas and winter is a relatively stable period, birds may still move on and off transects frequently. On a daily basis during non-breeding seasons the number of males varies more than females at a given location as males range over a larger area than females (Robertson et al. 2000) and counts of males on any particular day may represent only a portion of the males using that habitat (Robertson et al. 1999). If short-term movements did not affect all regions, geographic areas, and treatments equally these could have contributed to the differences we observed in sex ratios between treatments.

Geographic Variation

We tested whether temporal change would affect birds in all four areas equally in the absence of oil. If we observed significant differences in trends or composition between regions within the same treatment we would attribute these differences to geographic (spatial) variation. Natural geographic differences between oiled and unoiled sites resulting from differences in currents, physiography, freshwater runoff, local climate, and nutrients may effect population change disproportionately (Spies et al. 1996, Patten et al. 2000). Further, differences in beach morphology affect the weathering and persistence of oil differently (Hayes and Michel, 1999). We attempted to randomize these possible effects by choosing multiple, widely scattered oiled and non-oiled study sites balanced between mainland and island habitats (Spies et al. 1996).

Region 3 in WPWS (Appendix A) was the only geographic unit we tested that differed from zero in its mean rate of change in density. This region contains almost half the ducks in WPWS. The large number of ducks in that region contributed to more relative stability and less variation in our counts, increasing the likelihood of detecting a trend. We did not observe any differences in population growth rates (rate of density change) at larger spatial scales (areas or treatments). Natural geographic variation within PWS is not affecting population change disproportionately at the spatial scale we used to define our regions, areas and treatments. The amount of variability in our surveys limits our ability to detect subtle differences.

CONCLUSIONS

The EVOS Trustee Council (2002) defined recovery as a return to prespill demographics and similar levels of hydrocarbon exposure in ducks from oiled and unoiled areas (treatments). The lack of prespill demographic data prevents direct comparison. However, a comparison between oiled and unoiled treatments in PWS and BC populations indicate demographics in oiled areas of PWS are similar to those elsewhere in their range. The lower proportion of females in the oiled treatment may be a lingering effect from lower female survival in oiled areas (Esler et al. 2002) and remains a concern. A stable population in the oiled treatment, similar age ratios, and recent increases in over–

winter female survival rates in concert with overall reductions in hydrocarbon exposure rates from 1998 to 2005 (Ballachey et al. 2006) are positive signs for recovery. The USFWS marine bird surveys, although equivocal, indicate a possible population increase in the oiled areas since 1990 (Lance et al. 2001).

However, harlequin ducks in oiled areas were still being exposed to lingering *Exxon Valdez* oil at a significantly greater rate than birds in unoiled areas (Ballachey et al. 2006). The persistence of bioactive *EVO* in intertidal sediments (Short et al. 2005), continued exposure to hydrocarbons, and proportionately fewer females provided a basis for continued population-level effects. Barrow's goldeneyes, another sea duck occupying similar habitats, also exhibited greater exposure to hydrocarbons in oiled than unoiled areas (Ballachey et al. 2006) and this was coincidental to population declines in oiled areas relative to unoiled areas from 1990-2005 (Sullivan et al. 2005).

Detecting population change requires numerous samples, distributed through time, focusing on long-lived species (Paine et al. 1996). While our surveys do not provide evidence of an absolute increase in the oiled population or evidence of oiled populations increasing relative to unoiled populations, we lacked sufficient sampling through time to detect small differences in population change between treatments given the amount of variability in our sample and the relatively few years of surveys. However, survey data does not provide evidence that oiled populations have increased sufficiently to account for losses from initial and chronic spill mortality.

The rate of recovery for a species depends upon natural environmental processes, the species biology and ecology, and the interaction of those processes with any effects of the original disturbance. The persistence of moderately weathered subsurface *Exxon Valdez* oil in the intertidal provides a basis for potentially long-term biological effects (Short et al. 2004) but we do not have good knowledge of how or if, chronic low-level oil exposure is currently affecting harlequin ducks (Trust et al. 2000, Rizzolo et al. 2004). Assuming chronic effects from persistent oil in the intertidal, then declining levels of hydrocarbon exposure to background levels, coupled with converging values in female survival, sex and age composition, and increased population growth rates are the best indication of recovery and must be viewed collectively.

There are no precedents for recovery from oil spills for harlequin ducks. Harlequin duck populations have relatively low intrinsic growth rates (Goudie et al. 1994) so full recovery from initial and chronic mortality may be delayed until long after all spill effects have abated (Esler et al. 2002). The outlook for full recovery is good. Populations in the oiled area are stable, age ratios are similar between treatments, oil exposure rates have declined, and female survival improved. The lower proportion of females in oiled areas, evidence of continued exposure, and no solid evidence for a population increase in oiled areas remains a concern.

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Table 1. Survey dates, kilometers of shoreline surveyed, and numbers of harlequin ducks counted in oiled areas of western (WPWS) and southwestern (SWPWS), and unoiled areas of eastern (EPWS) Prince William Sound and Montague Island, Alaska during March in 1997, 2000, 2001, 2002, 2004, and 2005.

	Year					
Survey Dates	1997 March 13-19	2000 March 20-31	2001 March 21-April 1	2002 March 18-31	2004 March 22 – April 5	2005 March 29 – April 9
Shoreline Surveyed (km)						
WPWS (oiled)	301.1	301.1	301.1	301.1	301.1	301.1
EPWS	227.8	227.8	227.8	219.0	227.8	227.8
SWPWS (oiled)	DNS[a]	143.3	148.9	148.9	148.9	148.9
Montague Island	DNS[a]	73.7	73.7	73.7	100.0	100.0
Total	528.9	745.9	751.5	742.7	777.8	777.8
No. of Harlequin Ducks						
WPWS (oiled)	1677	1814	1861	1651	1913	1863
EPWS	1183	1535	1664	1507	1401	1630
SWPWS (oiled)	DNS[a]	691	771	1029	916	941
Montague Island	DNS[a]	783	742	688	672[b]/1081[c]	752[b]/1133[c]
Total	2860	4823	5038	4875	4902[b]/5311[c]	5186[b]/5567[c]

[a] Did Not Survey
[b] Comparable to transects surveyed in 2000, 2001, and 2002
[c] Includes two new Montague Island transects added in 2004

Table 2. Number and composition of harlequin ducks counted in oiled areas of western (WPWS) and southwestern (SWPWS) and unoiled areas of eastern (EPWS) Prince William Sound and Montague Island (MONT), Alaska during March surveys in 1997, 2000, 2001, 2002, 2004, and 2005. Numbers in parenthesis indicate proportion for each category based on total birds classified except for 'Unclassified', which is a percent of total birds.

Study Area	Year	Adult Males	Immature Males	Unk.[a] Males	Total Males	Females	Un-classified[b]	Total	Density Birds/km
WPWS	1997	892 (54.8)	79 (4.8)	3 (0.2)	974 (59.8)	655 (40.2)	48 (2.9)	1677	5.6
WPWS	2000	986 (55.8)	70 (4.0)	2 (0.1)	1058 (59.9)	709 (40.1)	47 (2.6)	1814	6.0
WPWS	2001	958 (55.5)	55 (3.2)	2 (0.1)	1015 (58.8)	710 (41.2)	136 (7.3)	1861	6.2
WPWS	2002	871 (55.2)	67 (4.2)	1 (0.1)	939 (59.5)	640 (40.5)	72 (4.4)	1651	5.5
WPWS	2004	1052 (57.0)	83 (4.5)	2 (0.1)	1137 (61.6)	709 (38.4)	67 (3.5)	1913	6.4
WPWS	2005	1018 (55.4)	103 (5.6)	0 (0.0)	1121 (61.0)	716 (39.0)	26 (1.4)	1863	6.2
EPWS	1997	511 (52.8)	45 (4.7)	5 (0.5)	561 (58.0)	406 (42.0)	216 (18.3)	1183	5.2
EPWS	2000	706 (54.5)	44 (3.4)	0 (0.0)	750 (58.0)	545 (42.1)	240 (15.6)	1535	6.7
EPWS	2001	884 (56.1)	45 (2.9)	1 (0.1)	930 (59.0)	645 (41.0)	89 (5.3)	1664	7.3
EPWS	2002	683 (51.8)	59 (4.5)	6 (0.5)	748 (56.8)	570 (43.2)	189 (12.5)	1507	6.9
EPWS	2004	651 (53.2)	72 (5.9)	0 (0.0)	723 (59.1)	500 (40.9)	178 (12.7)	1401	6.2
EPWS	2005	730 (51.9)	97 (6.9)	0 (0.0)	827 (58.8)	580 (41.2)	223 (13.7)	1630	7.2
SWPWS	2000	373 (55.5)	13 (1.9)	6 (0.9)	392 (58.3)	280 (41.7)	19 (2.8)	691	4.8
SWPWS	2001	379 (54.4)	10 (1.4)	0 (0.0)	389 (55.8)	308 (44.2)	74 (9.6)	771	5.2
SWPWS	2002	565 (55.7)	42 (4.1)	1 (0.1)	608 (60.0)	406 (40.0)	15 (1.5)	1029	6.9
SWPWS	2004	495 (55.5)	40 (4.5)	1 (0.1)	536 (60.1)	356 (39.9)	24 (2.6)	916	6.2
SWPWS	2005	502 (55.5)	29 (3.2)	1 (0.1)	532 (58.8)	372 (41.2)	37 (3.9)	941	6.3
MONT	2000	334 (54.6)	12 (2.0)	0 (0.0)	346 (56.5)	266 (43.5)	171 (27.9)	783	10.6
MONT	2001	371 (54.6)	13 (1.9)	0 (0.0)	384 (56.6)	295 (43.4)	63 (8.5)	742	10.1
MONT	2002	373 (56.2)	21 (3.2)	0 (0.0)	394 (59.3)	270 (40.7)	24 (3.5)	688	9.3
MONT[c]	2004	348 (58.5)	11 (1.8)	4 (0.7)	363 (61.0)	232 (39.0)	77 (11.5)	672	9.1
MONT[c]	2005	388 (53.2)	20 (2.7)	0 (0.0)	408 (56.0)	321 (44.0)	39 (3.1)	752	10.2

[a] Age of males unknown.
[b] Not included in ratio analysis.
[c] Does not include two new transects added in 2004

Table 3. Ratios of demographic parameters for harlequin ducks in oiled areas of western (WPWS) and southwestern (SWPWS) Prince William Sound, and unoiled areas of eastern (EPWS) Prince William Sound and Montague Island (MONT), Alaska during March surveys in 1997, 2000, 2001, 2002, 2004, and 2005.

		Ratios		
Study Area	Year	Males: Females	Adult Males: Immature Males	Adult Females: Immatures
WPWS[a]	1997	1.49	11.29	3.65
WPWS	2000	1.49	14.09	4.56
WPWS	2001	1.43	17.42	5.95
WPWS	2002	1.47	13.00	4.28
WPWS	2004	1.60	12.67	3.77
WPWS	2005	1.57	9.88	2.98
WPWS	Mean	1.51	12.64	4.03
EPWS[b]	1997	1.38	11.36	4.01
EPWS	2000	1.38	16.05	5.69
EPWS	2001	1.44	19.64	6.67
EPWS	2002	1.31	11.58	4.33
EPWS	2004	1.45	9.04	2.97
EPWS	2005	1.42	7.53	2.49
EPWS	Mean	1.40	11.51	3.98
SWPWS[a]	2000	1.40	28.69	10.27
SWPWS	2001	1.26	37.9	14.9
SWPWS	2002	1.50	13.45	4.33
SWPWS	2004	1.51	12.38	3.95
SWPWS	2005	1.43	17.31	5.91
SWPWS	Mean	1.43	17.27	5.93
MONT[b,c]	2000	1.30	27.83	10.58
MONT	2001	1.30	28.54	10.85
MONT	2002	1.46	17.76	5.93
MONT	2004	1.56	31.64	10.05
MONT	2005	1.27	19.4	7.53
MONT	Mean	1.37	23.56	8.49

Continued

[a] Oiled
[b] Unoiled
[c] Original MONT transects, does not include transects added in 2004.

Table 3 (Cont).

Study Area	Year	Males: Females	Adult Males: Immature Males	Adult Females: Immatures
All Unoiled Areas	2000	1.35	18.57	6.74
All Unoiled Areas	2001	1.40	21.64	7.60
All Unoiled Areas	2002	1.36	13.20	4.75
All Unoiled Areas[c]	2004	1.48	12.04	3.91
All Unoiled Areas	2005	1.37	9.56	3.35
All Unoiled Areas	Mean	1.39	13.62	4.77
All Oiled Areas	2000	1.47	16.37	5.46
All Oiled Areas	2001	1.38	20.57	7.33
All Oiled Areas	2002	1.48	13.17	4.30
All Oiled Areas	2004	1.57	12.58	3.83
All Oiled Areas	2005	1.52	11.52	3.62
All Oiled Areas	Mean	1.48	13.69	4.46
All Areas[d]	1997	1.45	11.31	3.78
All Areas	2000	1.41	17.26	5.97
All Areas	2001	1.39	21.07	7.46
All Areas	2002	1.43	13.19	4.49
All Areas[c]	2004	1.54	12.36	3.86
All Areas[c]	2005	1.45	10.59	3.49
Total All Years and Areas		1.44	13.66	4.59

[a] Oiled
[b] Unoiled
[c] Original MONT transects, does not include transects added in 2004.
[d] EPWS and WPWS only

Table 4. Logit analysis used to test for differences in demographic parameters of harlequin duck populations in oiled areas of western (WPWS) and southwestern (SWPWS) Prince William Sound, and unoiled areas of eastern (EPWS) Prince William Sound and Montague Island (MONT) Alaska during March surveys in 1997, 2000, 2001, 2002, 2004 and 2005.

Sex Ratio Model (Males: Females)

Model	AIC	Parameter	DF[1]	F Value	Prob. > F
Full Model	18.20	Year	1/255	0.18	0.6710
		Treatment	1/55	3.42	0.0698
		Region (Treatment)	7/55	0.74	0.6402
		Year*Region (Treatment)	8/255	1.54	0.1423
Reduced Model	-18.14	Year	1/262	0.11	0.7450
		Treatment	1/55	2.23	0.1407
		Region (Treatment)	7/55	0.99	0.4493
		Year*Treatment)	1/262	3.19	0.0754
Reduced Model	-22.87	Year	1/263	0.00	0.9996
		Treatment	1/55	4.13	0.0471
		Region (Treatment)	7/55	0.88	0.5284
Reduced Model	-32.13	Treatment	1/55	4.13	0.0470
		Region (Treatment)	7/55	0.88	0.5286
Reduced Model[2]	-55.86	Treatment	2/62	146.58	0.0218

Male Age Ratio Model (Adult Males: Immature Males)

Model	AIC	Parameter	DF[1]	F Value	Prob. > F
Full Model	902.82	Year	1/261	13.04	0.0004
		Treatment	1/55	1.60	0.2112
		Region (Treatment)	7/55	3.79	0.0020
		Year*Treatment	1/261	2.98	0.0857
Reduced Model	908.34	Year	1/262	8.28	0.0043
		Treatment	1/62	0.01	0.9152
Reduced Model	896.12	Treatment	1/55	0.37	0.5478
		Region (Treatment)	7/55	3.39	0.0044
Reduced Model[2]	895.67	Year	1/262	11.22	0.0009
		Region (Treatment)	9/55	303.31	<.0001

Table 4 (Cont).

Recruitment Ratio Model (Adult Females:Immatures)

Model	AIC	Parameter	DF[1]	F Value	Prob. > F
Full Model	961.00	Year	1/253	11.13	0.0010
		Treatment	1/55	2.16	0.1473
		Region (Treatment)	7/55	3.19	0.0066
		Year* Region (Treatment)	8/253	0.83	0.5807
Reduced Model	942.55	Year	1/260	15.09	0.0001
		Treatment	1/55	2.88	0.0955
		Region (Treatment)	7/55	4.21	0.0009
		Year* Treatment	1/260	2.11	0.1479
Reduced Model[2]	936.02	Year	1/261	13.66	0.0003
		Region (Treatment)	9/55	93.28	<.0001
Reduced Model	937.63	Treatment	1/55	1.42	0.2382
		Region (Treatment)	7/55	3.73	0.0023

[1] Degrees of Freedom, Numerator/Denominator
[2] Best model fit

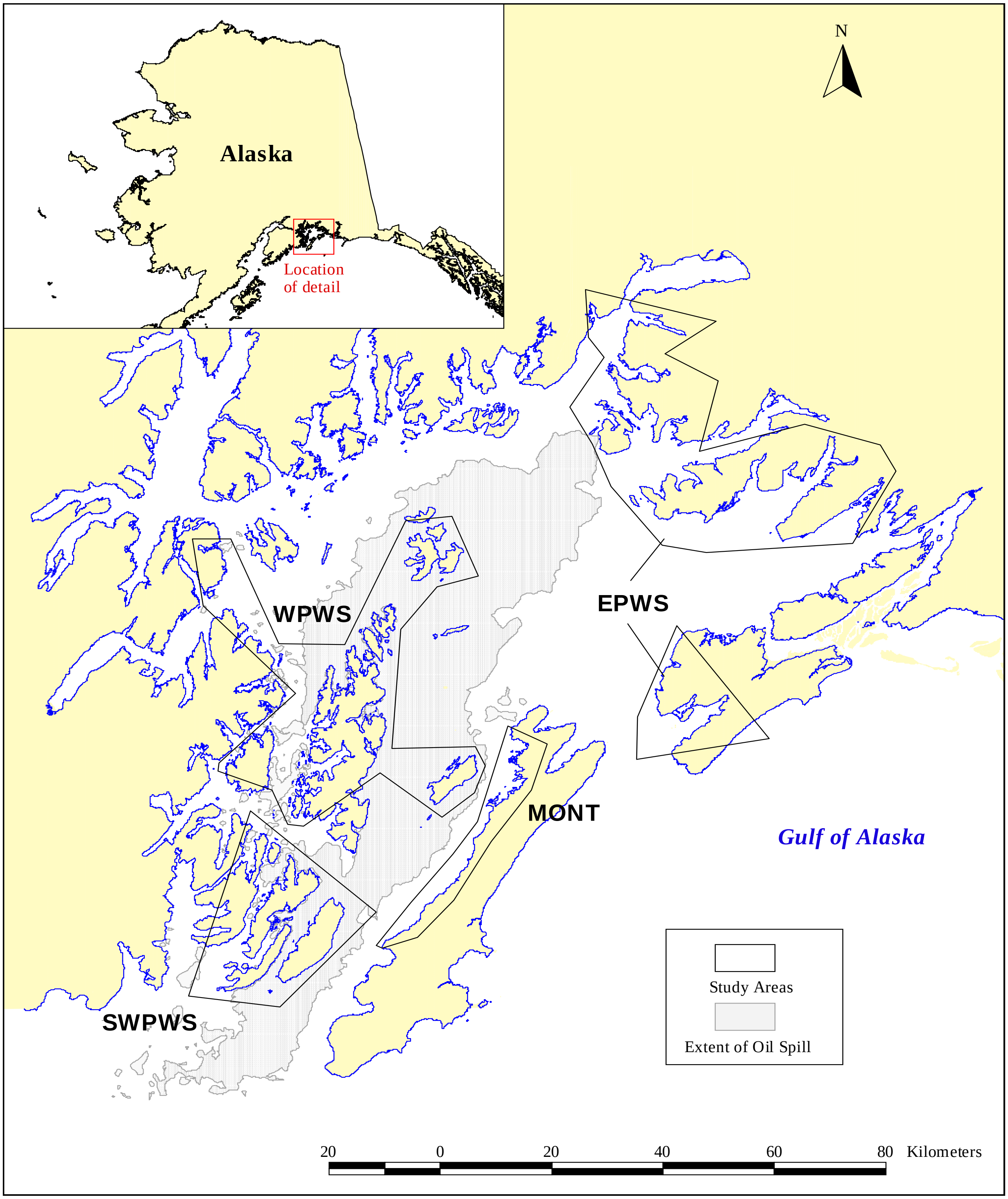


Figure 1. Map showing the extent of the *T/V Exxon Valdez* oil spill in Prince William Sound, Alaska, and the general location of study areas (western [WPWS], southwestern [SWPWS], eastern [EPWS], and Montague Island [MONT]) used to survey harlequin ducks.

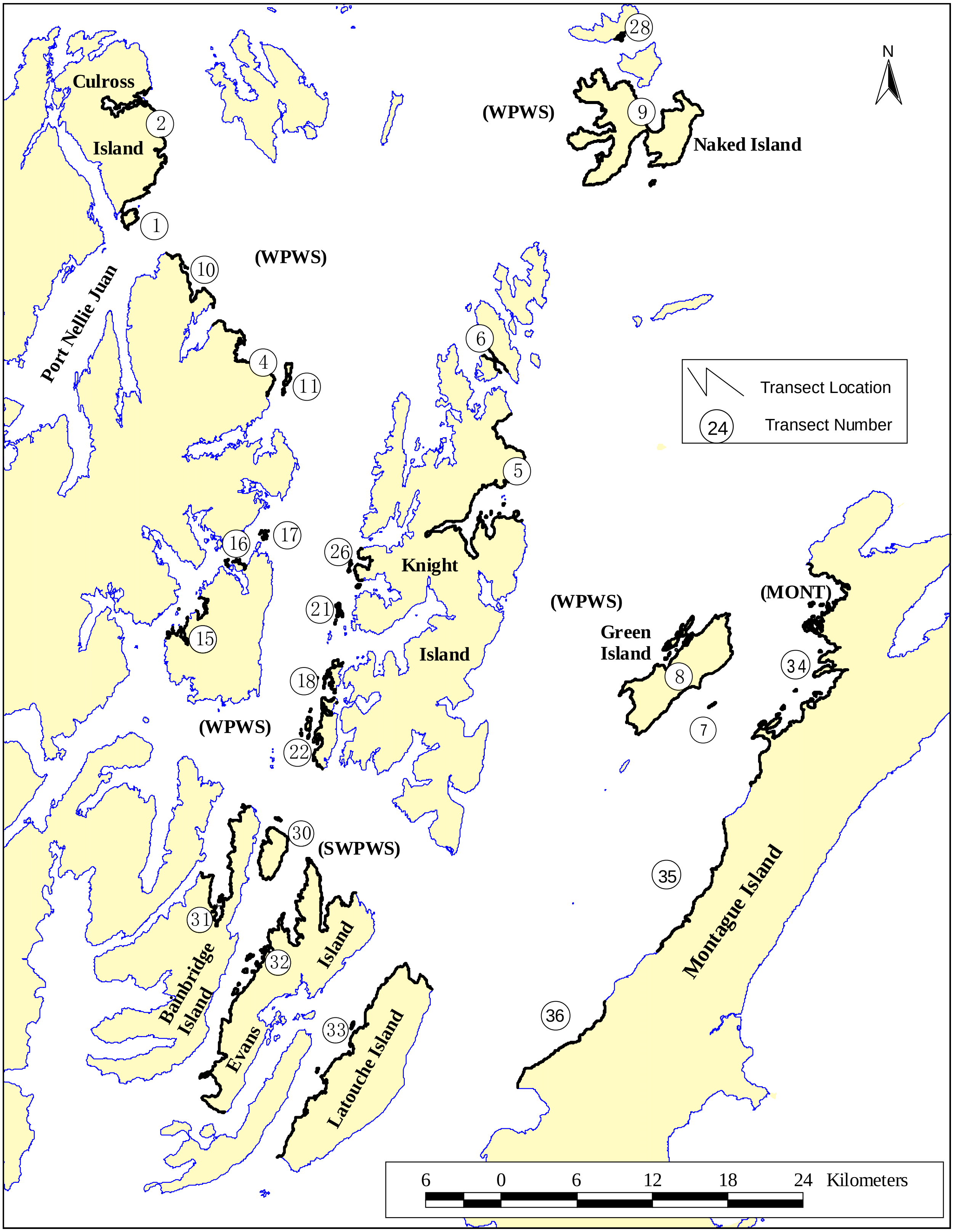


Figure 2. Location of transects in oiled (western [WPWS] and southwestern [SWPWS]) and unoiled (Montague Island [MONT]) areas of Prince William Sound, Alaska surveyed for harlequin ducks during March in 1999, 2000, 2001, 2002, 2004 and 2005.

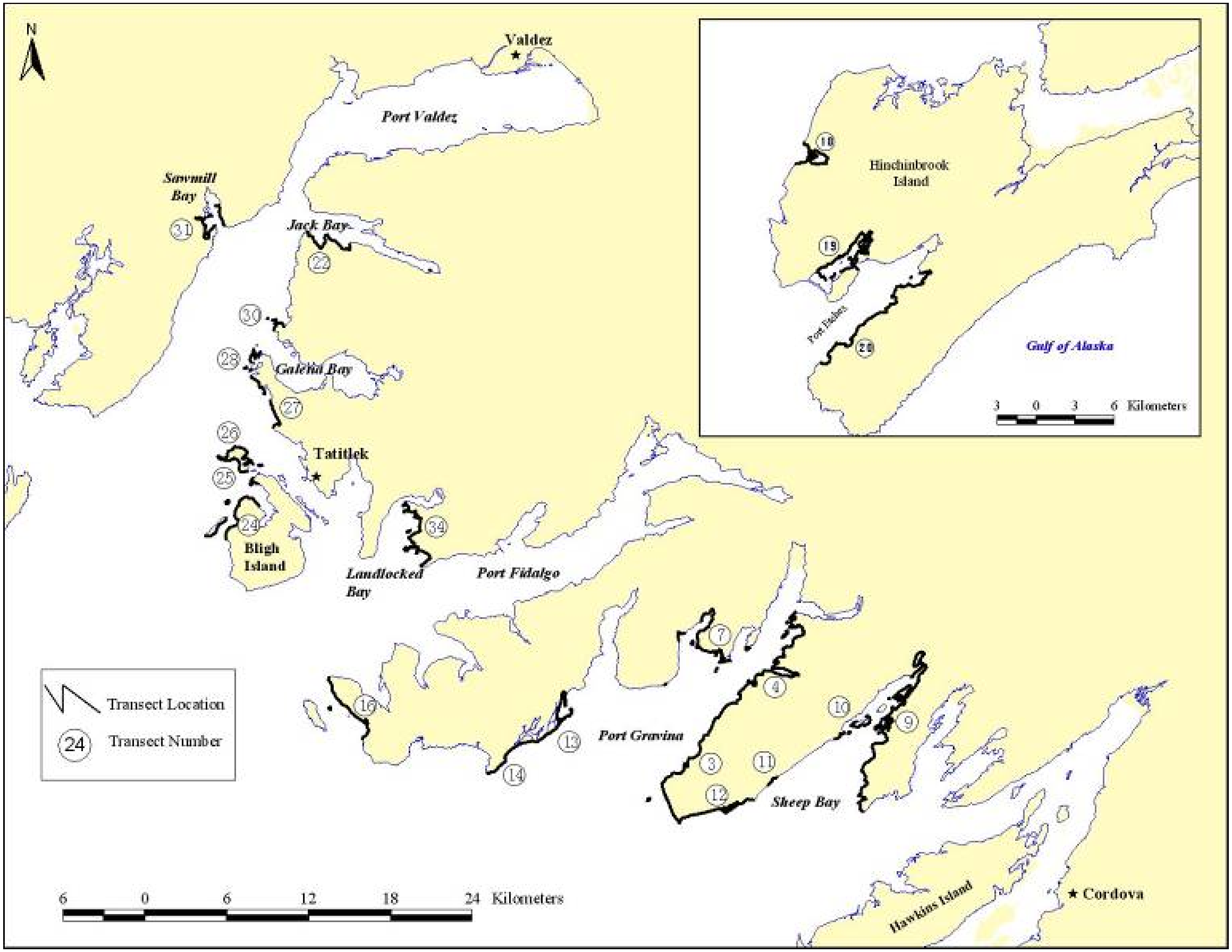


Figure 3. Location of transects in unoiled areas of eastern Prince William Sound (EPWS), Alaska surveyed for harlequin ducks during March in 1999, 2000, 2001, 2002, 2004 and 2005.

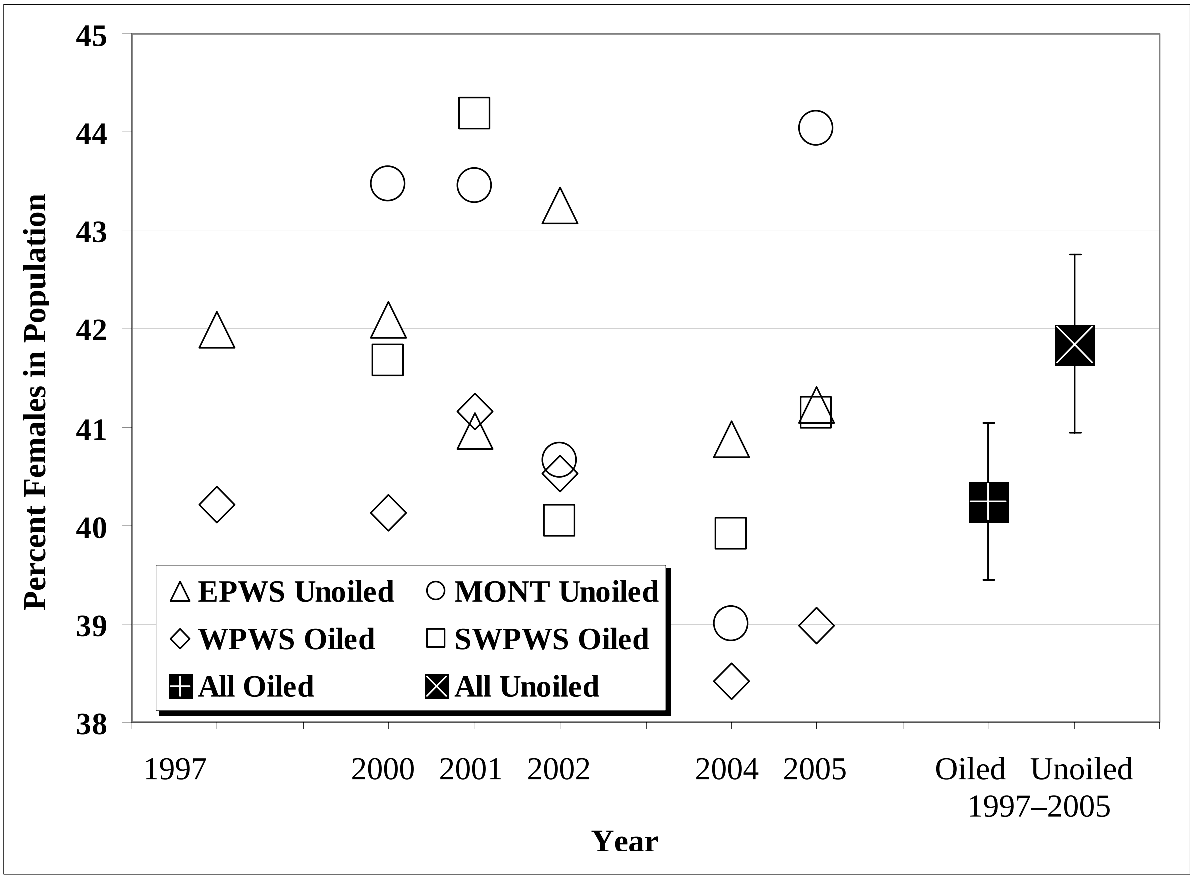


Figure 4. Percent of female harlequin ducks in Prince William Sound, Alaska during March shown by year (1997, 2000, 2001, 2002, 2004 and 2005), area (western [WPWS], southwestern [SWPWS], eastern [EPWS] and Montague Island [MONT]) and by treatment (oiled and unoiled).

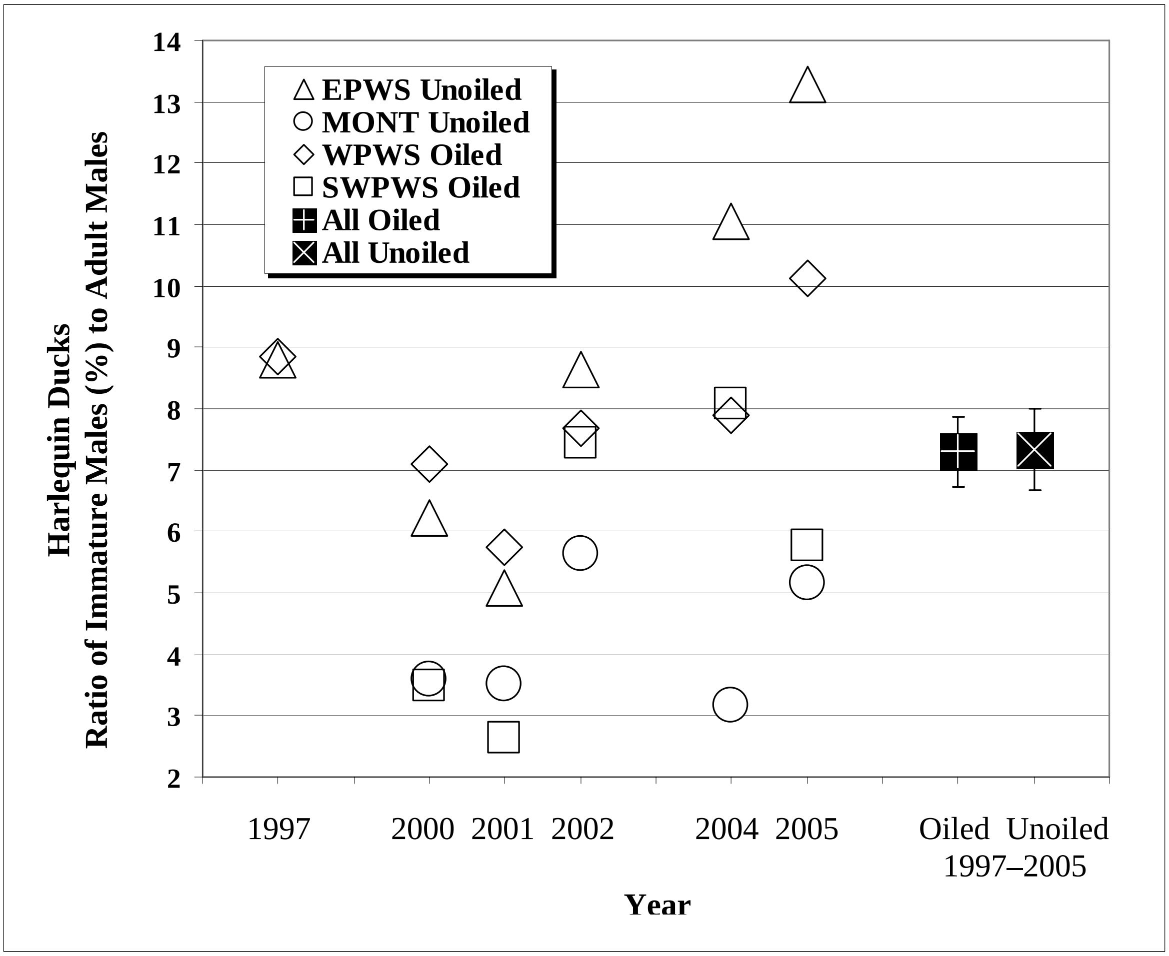


Figure 5. Ratio of immature to adult male harlequin ducks (expressed as a percent) shown by year (1997, 2000, 2001, 2002, 2004 and 2005), area (western [WPWS], southwestern [SWPWS], eastern [EPWS] and Montague Island [MONT]) and by treatment (oiled and unoiled).

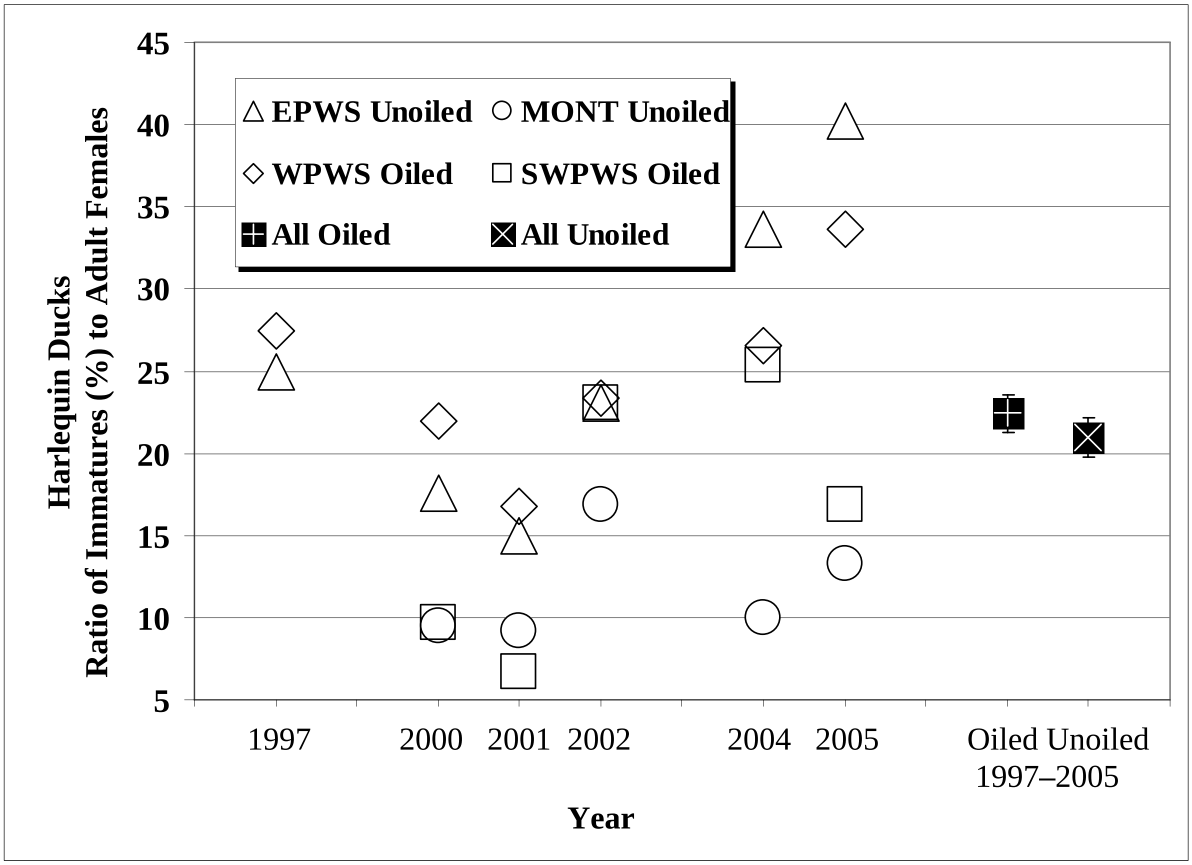


Figure 6. Ratio of immature (male and female) to adult female harlequin ducks (expressed as a percent) shown by year (1997, 2000–2002, 2004 and 2005), area (western [WPWS], southwestern [SWPWS], eastern [EPWS] and Montague Island [MONT]) and by treatment (oiled and unoiled).

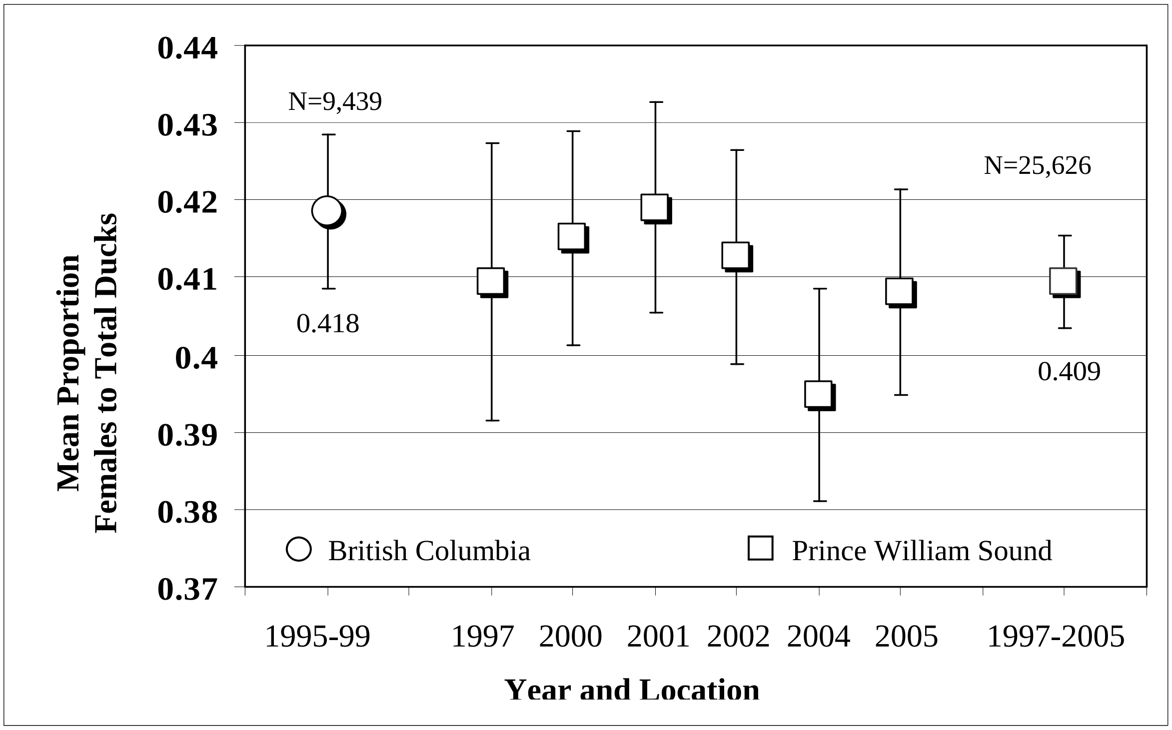


Figure 7. Comparison between Prince William Sound, Alaska (this study) and Strait of Georgia, British Columbia (Smith et al. 2001) in the proportion of females in the total harlequin duck population during winter. Prince William Sound data is presented for each survey year and all years combined.

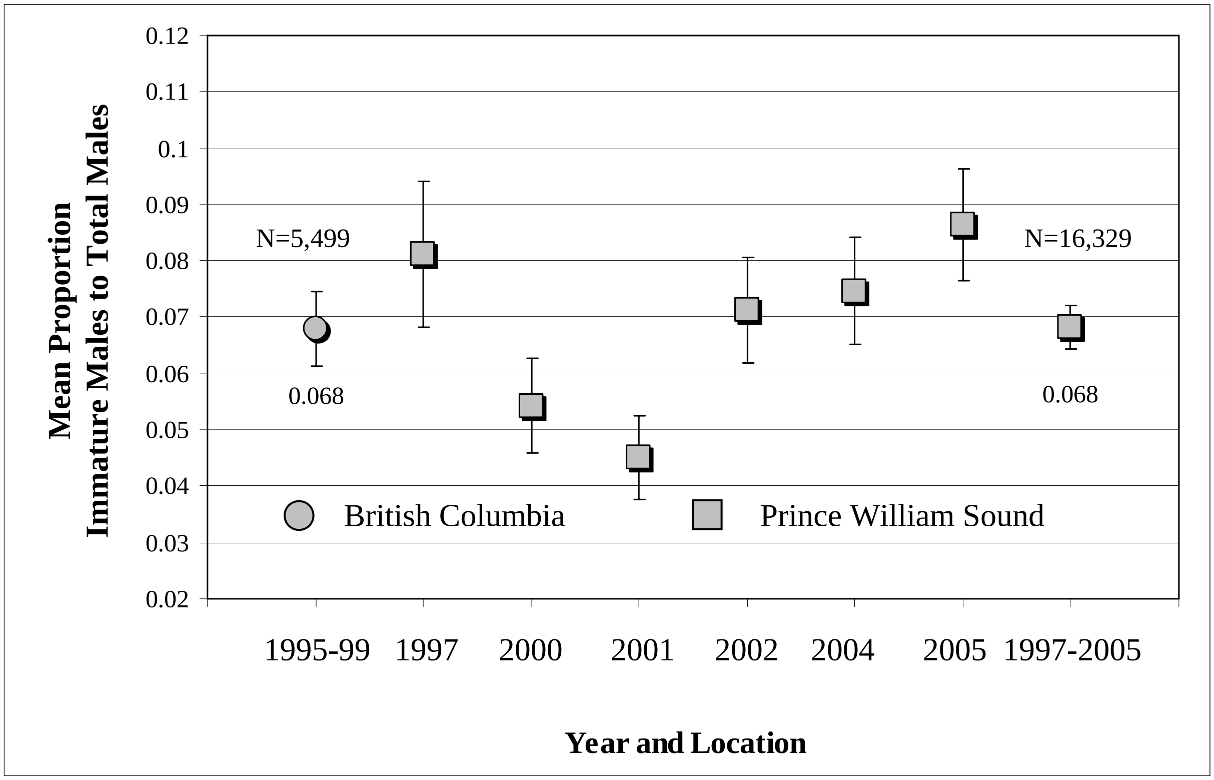


Figure 8. Comparison between Prince William Sound, Alaska (this study) with Strait of Georgia, British Columbia (Smith et al. 2001) in the ratio of immature males to total male harlequin ducks during winter.

Appendix A. Transect, region, and study area spatial scales (see Methods) used to compare trends in harlequin ducks observed in oiled areas of western (WPWS) and southwestern (SWPWS) and unoiled areas of eastern (EPWS) Prince William Sound and Montague Island (MONT), Alaska in March 1997–2005. SWPWS and MONT transects were not surveyed in 1997. Individual transects in SWPWS and MONT were divided into segments to facilitate analysis. Two new MONT transects were added in 2004.

Study Area	Location	Transect number[a]	Transect length (km)	Region[b]	Study Area	Location	Transect number	Transect length (km)	Region
WPWS	Aguliak Island	26	9.0	2	WPWS	Green Island	8	51.5	3
WPWS	Applegate Island	1	5.9	1	WPWS	Junction Island	17	2.7	2
WPWS	Bay of Isles	5	41.9	3	WPWS	Masked Bay	16	2.6	2
WPWS	Channel Island	7	1.6	3	WPWS	Mummy Island	18	10.8	2
WPWS	Crafton Island	11	6.8	1	WPWS	Naked Island	9	73.2	4
WPWS	Culross Island	2	21.0	1	WPWS	Squire Island	22	21.3	2
WPWS	Falls Bay	4	15.1	1	WPWS	Squirrel Island	21	4.5	2
WPWS	Foul Bay	10	11.7	1	WPWS	Storey Island	28	2.8	4
WPWS	Foul Pass	6	5.5	3	WPWS	Totemoff Creek	15	13.2	2
SWPWS	Bainbridge Bay[c]	31a	13.2	1	SWPWS	Latouche Is. (N)	33a	18.5	1
SWPWS	Bainbridge Pt. [c]	31b	13.1	1	SWPWS	Latouche Is. (S)	33c	2.8[e]	1
SWPWS	Danger Island	33d	2.9[e]	1	SWPWS	Latouche Is. (SW)	33b	16.1	1
SWPWS	Flemming Island	30a	12.6	1	SWPWS	Prince of Wales[d]	32c	20.2	1
SWPWS	Gage Island	30b	1.2	1	SWPWS	Shelter Bay[d]	32a	17.7	1
SWPWS	Iktua Bay[d]	32b	15.9	1	SWPWS	Squirrel Bay[d]	32d	14.7	1
EPWS	Black Creek	27	2.6	2	EPWS	Port Etches	20	17.0	3
EPWS	Busby Island (N)	26	6.2	2	EPWS	Port Gravina (NE)	4	20.6	1
EPWS	Busby Island (S)	25	6.2	2	EPWS	Port Gravina (SE)	3	17.3	1
EPWS	Close Island	10	4.8	1	EPWS	Redhead	14	8.8	1
EPWS	Constantine Harbor	19	19.7	3	EPWS	Reef/Bligh Islands	24	7.1	2
EPWS	Galena Rocks	30	2.5	2	EPWS	Rocky Pt./Galena Is.	28	6.1	2
EPWS	Hell's Hole	13	6.4	1	EPWS	Sawmill Bay	31	7.4	2
EPWS	Jack Bay	22	5.7	2	EPWS	Sheep Bay (E)	9	35.0	1
EPWS	Landlocked Bay	34	13.3	2	EPWS	Sheep Bay (SW)	12	8.8	1
EPWS	Olsen Bay	7	14.9	1	EPWS	Shelter Bay	18	9.0	3
EPWS	Porcupine Bay	16	7.4	2	EPWS	Surf Creek	11	1.0	1
MONT	Central Montague[f]	35	14.9	1	MONT	Port Chalmers (N)	34c	7.8	1
MONT	Gilmour Point	34d	9.9	1	MONT	Port Chalmers (S)	34f	10.7	1
MONT	Graveyard Point	34a	11.2	1	MONT	Stockdale Harbor	34b	14.8	1
MONT	Moose Lips	34g	10.4	1	MONT	Wilby Island	34e	8.9	1
MONT	Point Bazil[f]	36	11.4	1					

[a] Transect numbers referenced in Fig. 2 and Fig. 3.
[b] Regions are discreet for each study area
[c] Bainbridge Island
[d] Evans Island
[e] Did not survey in 2000
[f] New transect added in 2004

Appendix B. Number of harlequin ducks counted on transects surveyed in oiled areas of western (WPWS) and southwestern Prince William Sound (SWPWS), unoiled areas of eastern Prince William Sound (EPWS) and Montague Island (MONT), Alaska in March 1997, 2000–2002, 2004, and 2005.

	Transect		Year						Mean	Percent of	Density
Location	Number	Dist. (km)	1997	2000	2001	2002	2004	2005	No.	Total	Birds/km
WPWS											
Aguliak Island	26	9.0	37	67	58	46	59	81	58	3.2	6.4
Applegate Is.	1	5.9	40	33	54	43	33	44	41	2.3	7.0
Bay of Isles	5	41.9	86	81	157	124	114	113	113	6.3	2.7
Channel Island	7	1.6	33	31	67	65	24	67	48	2.7	29.9
Crafton Island	11	6.8	79	71	27	22	80	53	55	3.0	8.1
Culross Island	2	21.0	96	62	84	75	157	86	93	5.2	4.4
Falls Bay	4	15.1	154	167	86	102	89	86	114	6.4	7.6
Foul Bay	10	11.7	146	193	213	169	174	180	179	10.0	15.3
Foul Pass	6	5.5	6	13	20	24	15	22	17	0.9	3.0
Green Island	8	51.5	559	644	682	597	711	708	650	36.2	12.6
Junction Island	17	2.7	20	18	24	27	14	41	24	1.3	8.9
Masked Bay	16	2.6	3	6	0	2	10	4	4	0.2	1.6
Mummy Island	18	10.8	51	48	44	44	49	40	46	2.6	4.3
Naked Island	9	73.2	168	221	233	169	258	202	209	11.6	2.9
Squire Island	22	21.3	105	79	63	88	63	63	77	4.3	3.6
Squirrel Island	21	4.5	59	34	24	23	17	30	31	1.7	6.9
Storey Island	28	2.8	6	15	10	14	17	11	12	0.7	4.4
Totemoff Creek	15	13.2	29	31	15	17	29	32	25.5	1.4	1.9
Total		301.1	1677	1814	1861	1651	1913	1863	1797	100	6.0

Appendix B (Cont).

Location	Transect Number	Dist. (km)	1997	2000	2001	2002	2004	2005	Mean No.	Percent of Total	Density Birds/km
SWPWS											
Bainbridge Bay[b]	31a	13.2	DNS[a]	9	13	17	23	24	17	2.0	1.3
Bainbridge Pt.[b]	31b	13.1	DNS[a]	72	27	107	100	67	75	8.6	5.7
Danger Island	33d	2.9	DNS[a]	DNS[a]	120	127	119	148	129	14.8	44.3
Flemming Island	30a	12.6	DNS[a]	55	31	80	47	37	50	5.8	4.0
Gage Island	30b	1.2	DNS[a]	7	0	5	4	2	4	0.4	3.0
Iktua Bay[c]	32b	15.9	DNS[a]	64	23	60	60	58	53	6.1	3.3
Latouche Is. (N)	33a	18.5	DNS[a]	151	104	149	129	132	133	15.3	7.2
Latouche Is. (S)	33c	2.8	DNS[a]	DNS[a]	116	123	135	94	117	13.5	41.8
Latouche Is. (SW)	33b	16.1	DNS[a]	123	144	138	141	168	143	16.4	8.9
Prince of Wales[c]	32c	20.2	DNS[a]	56	75	98	53	77	71.8	8.3	3.6
Shelter Bay[c]	32a	17.7	DNS[a]	64	58	84	57	90	71	8.1	4.0
Squirrel Bay[c]	32d	14.7	DNS[a]	90	60	41	48	44	57	6.5	3.9
Total		148.9	------	691	771	1029	916	941	870	100	5.8

[a] Did not survey
[b] Bainbridge Island
[c] Evans Island

Appendix B (Cont).

Location	Transect Number	Transect Dist. (km)	Year 1997	2000	2001	2002	2004	2005	Mean No.	Percent of Total	Density Birds/km
EPWS											
Black Creek	27	2.6	4	6	9	4	14	16	9	0.6	3.4
Busby Island(N)	26	6.2	44	74	47	66	59	67	60	4.0	9.6
Busby Island(S)	25	6.2	35	81	74	78	76	58	67	4.5	10.8
Close Island	10	4.8	107	107	51	51	56	49	70	4.7	14.6
Constantine Harbor	19	19.7	27	49	44	64	54	52	48	3.3	2.5
Galena Rocks	30	2.5	0	18	35	9	4	19	14	1.0	5.7
Hell’s Hole	13	6.4	65	68	163	265	9	42	102	6.9	15.9
Jack Bay	22	5.7	21	31	28	25	20	54	30	2.0	5.2
Landlocked Bay	34	13.3	42	82	96	136	72	60	81	5.5	6.1
Olsen Bay	7	14.9	95	67	110	95	100	260	121	8.2	8.1
Porcupine Bay	16	7.4	30	83	101	66	50	60	65	4.4	8.8
Port Etches	20	17.0	55	86	82	67	84	77	75	5.0	4.4
Port Gravina (NE)	4	17.3	39	34	72	18	59	24	41	2.8	2.4
Port Gravina (SE)	3	20.6	189	149	273	189	192	226	203	13.7	9.9
Redhead	14	8.8	59	185	64	DNS[a]	22	29	60	4.0	6.8
Reef/Bligh Islands	24	7.1	23	9	81	73	30	47	44	3.0	6.2
Rocky Point/Galena	28	6.1	54	16	46	17	25	36	32	2.2	5.3
Sawmill Bay	31	7.4	0	8	18	8	12	22	11	0.8	1.5
Sheep Bay (E)	9	35.0	181	148	133	163	238	131	166	11.1	4.7
Sheep Bay (SW)	12	8.8	55	152	73	35	143	228	114	7.7	13.0
Shelter Bay	18	9.0	34	52	13	42	73	72	47.7	3.2	5.3
Surf Creek	11	1.0	24	30	51	33	9	1	24.7	11.7	24.7
Total		227.8	1183	1535	1664	1504	1401	1630	1486	100	6.5

[a] Did not survey

Appendix B (Cont).

Location	Transect Number	Dist. (km)	Year 1997	2000	2001	2002	2004	2005	Mean No.	Percent of Total	Density Birds/km
MONT											
Gilmour Point	34d	9.9	DNS[a]	23	89	47	69	130	72	9.8	7.2
Graveyard Point	34a	11.2	DNS[a]	135	99	124	106	81	109	15.0	9.7
Moose Lips	34g	10.4	DNS[a]	144	155	206	183	144	166	22.9	16.0
Port Chalmers (N)	34c	7.8	DNS[a]	32	72	62	93	68	65	9.0	8.4
Port Chalmers (S)	34f	10.7	DNS[a]	95	104	37	97	116	90	12.4	8.4
Stockdale Harbor	34b	14.8	DNS[a]	86	91	97	51	180	101	13.9	6.8
Wilby Island	34e	8.9	DNS[a]	268	132	115	73	33	124	17.1	14.0
Total		73.8	------	783	742	688	672	752	727	100	9.9
Central Montague	35	14.9	DNS[a]	DNS[a]	DNS[a]	DNS[a]	147	130	139	35.1	9.3
Point Basil	36	11.4	DNS[a]	DNS[a]	DNS[a]	DNS[a]	262	251	257	65.0	22.5
Total		26.3	------	------	------	------	409	381	395	100	15.0

[a] Did not survey

Appendix C. Number and composition of harlequin ducks in oiled areas of western (WPWS) and southwestern (SWPWS) and unoiled areas of eastern (EPWS) Prince William Sound and Montague Island (MONT), Alaska after unknown birds were partitioned among the appropriate age, sex, and breeding categories based on observed proportions. Numbers are presented for March 1997, 2000–2002, 2004, and 2005 surveys.

WPWS

Number of Harlequin Ducks by Sex and Age Classifications

Original Count/ Corrected Count

Year	Adult Males	Immature Males	Unknown Males[a]	Females	Unclassified[b]	Total
1997	892/918	79/81	3/4	655/674	48	1677
2000	986/1012	70/72	2/2	709/728	47	1814
2001	958/1033	55/59	2/2	710/767	136	1861
2002	871/911	67/70	1/1	640/669	72	1651
2004	1052/1090	83/86	2/2	709/735	67	1913
2005	1018/1032	103/104	0/0	716/726	26	1863

EPWS

Number of Harlequin Ducks by Sex and Age Classifications

Original Count/ Corrected Count

Year	Adult Males	Immature Males	Unknown Males[a]	Females	Unclassified[b]	Total
1997	511/625	45/55	5/6	406/497	216	1183
2000	706/837	44/52	0/0	545/646	240	1535
2001	884/934	45/48	1/1	645/681	89	1664
2002	683/781	59/68	6/7	570/651	189	1507
2004	651/746	72/82	0/0	500/573	178	1401
2005	730/846	97/112	0/0	580/672	223	1630

[a] Age of males unknown.
[b] Distributed among other categories based on relative percent.

Appendix C (Cont).

SWPWS

Number of Harlequin Ducks by Sex and Age Classifications

Original Count/ Corrected Count

Year	Adult Males	Immature Males	Unknown Males[a]	Females	Unclassified [b]	Total
2000	373/384	13/13	6/6	280/288	19	691
2001	379/419	10/11	0/0	308/341	74	771
2002	565/572	42/43	1/0	406/412	15	1029
2004	495/508	40/41	1/1	356/366	24	916
2005	502/524	29/30	1/0	372/387	37	941

MONT

Number of Harlequin Ducks by Sex and Age Classifications

Original Count/ Corrected Count

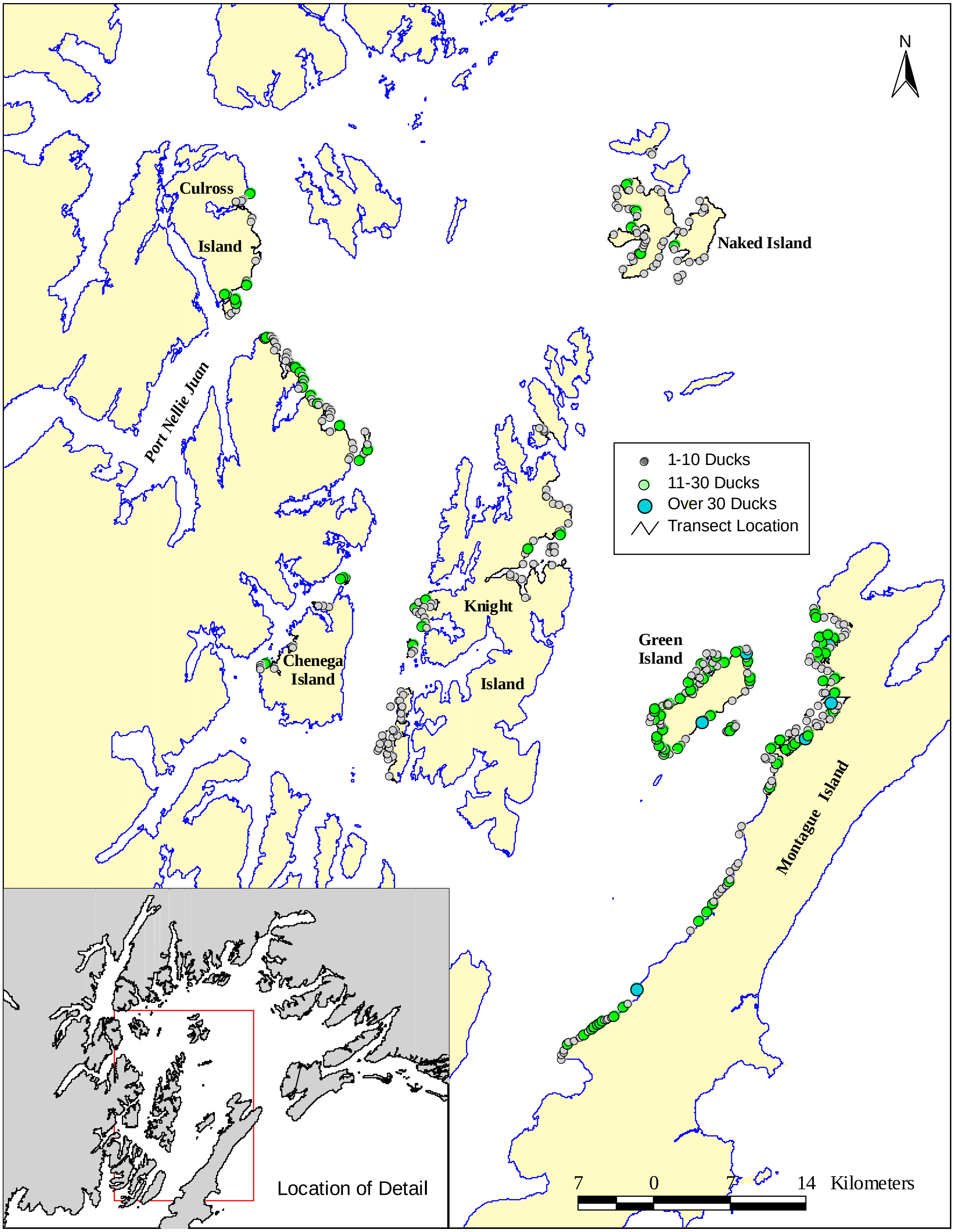
Year	Adult Males	Immature Males	Unknown Males[a]	Females	Unclassified [b]	Total
2000	334/427	12/16	0/0	266/340	171	783
2001	371/406	13/14	0/0	295/322	63	742
2002	373/386	21/22	0/0	270/280	24	688
2004[c]	516/628	20/24	5/6	347/423	193	1081
2004[d]	348/394	11/12	4/4	232/262	77	672
2005[c]	590/611	37/38	0/0	467/484	39	1133
2005[d]	388/400	20/21	0/0	321/331	23	752

[a] Age of males unknown.

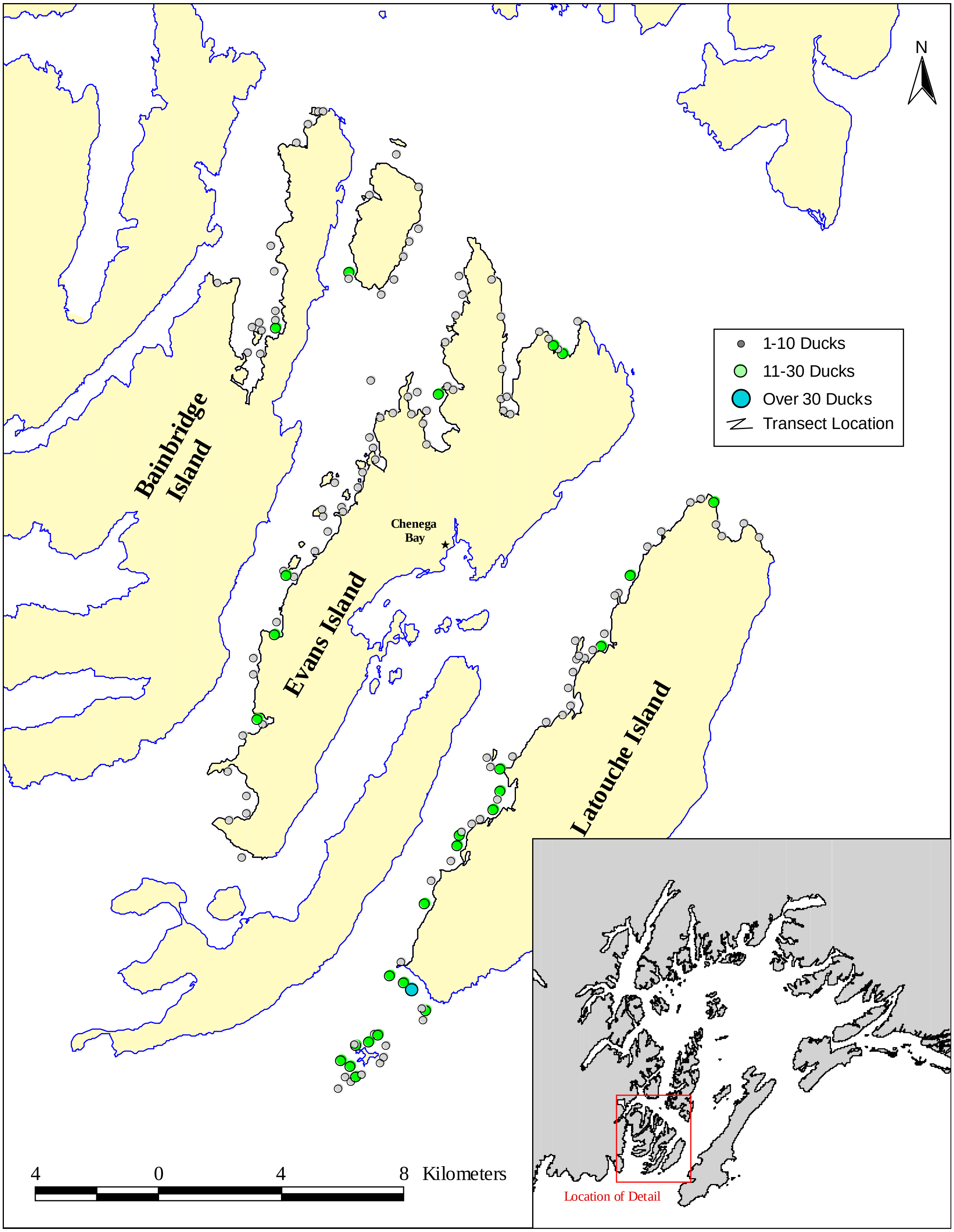
[b] Distributed among other categories based on relative percent.

[c] Includes two new transects added in 2004

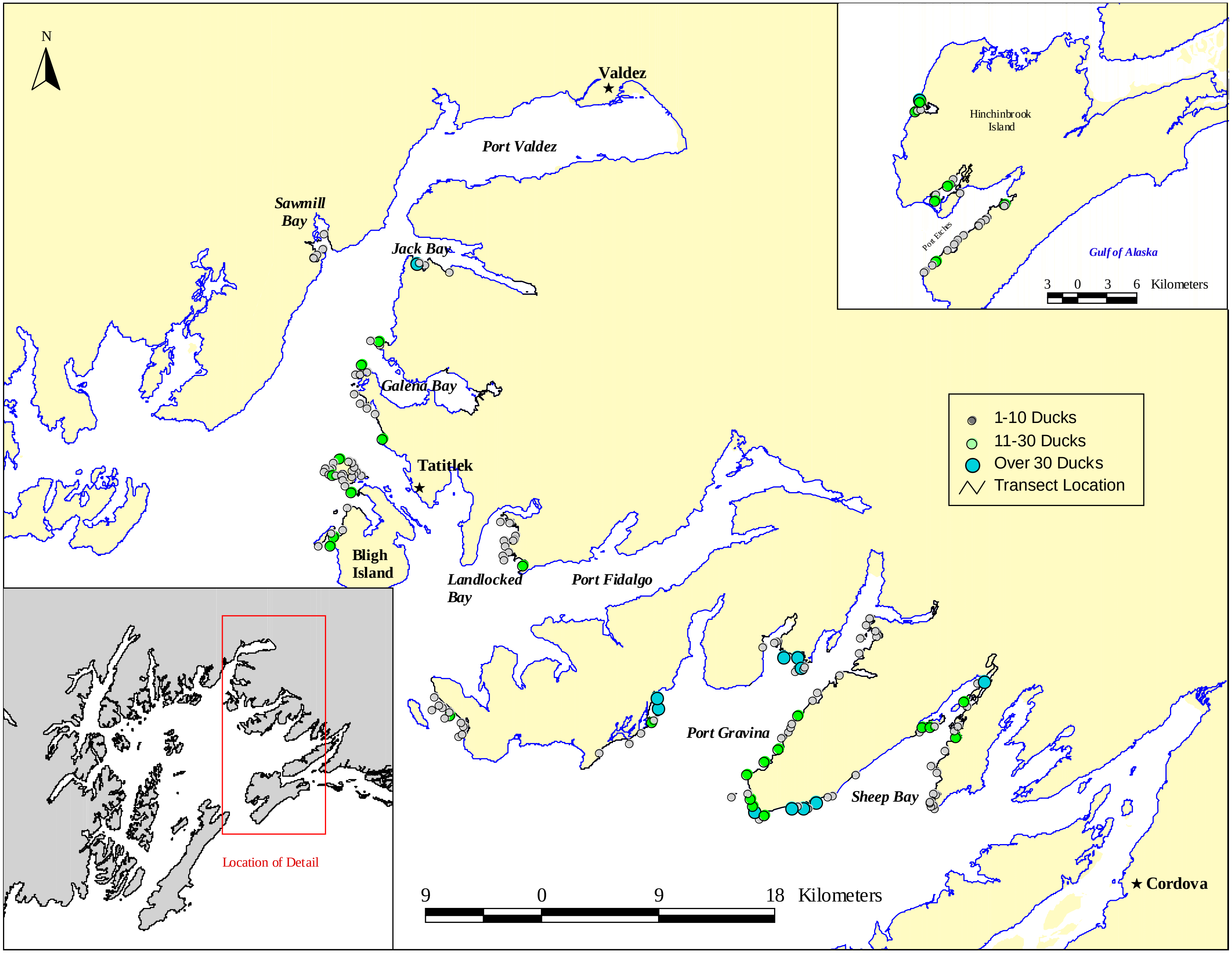
d Original transects established in 2000 only



Appendix D. Distribution and relative flock sizes of harlequin ducks observed during March surveys in western (WPWS) Prince William Sound, AK and Montague Island (MONT) in 2005, and considered typical of annual distribution throughout the course of the study.



Appendix E. Distribution and relative flock sizes of harlequin ducks observed during March surveys in southwestern (SWPWS) Prince William Sound, AK in 2005 and considered typical of annual distribution throughout the course of the study.



Appendix F. Distribution and relative flock sizes of harlequin ducks observed during March surveys in eastern (EPWS) Prince William Sound, AK in 2005 and considered typical of annual distribution throughout the course of the study.