Exxon Valdez Oil Spill
Restoration Project Final Report

Long-term Monitoring: Lingering Oil
Evaluating Chronic Exposure of Harlequin Ducks and Sea Otters to Lingering *Exxon Valdez* Oil in Western Prince William Sound

Project 12120114-Q

Daniel Esler
Pacific Wildlife Foundation and
Centre for Wildlife Ecology
Department of Biological Sciences
Simon Fraser University
5421 Robertson Road
Delta, British Columbia V4K 3N2 Canada

July 2013

Exxon Valdez Oil Spill
Restoration Project Final Report

Long-term Monitoring: Lingering Oil
Evaluating Chronic Exposure of Harlequin Ducks and Sea Otters to Lingering *Exxon Valdez* Oil in Western Prince William Sound

Project 12120114-Q

Daniel Esler
Pacific Wildlife Foundation and
Centre for Wildlife Ecology
Department of Biological Sciences
Simon Fraser University
5421 Robertson Road
Delta, British Columbia V4K 3N2 Canada

July 2013

Long-term Monitoring: Lingering Oil
Evaluating Chronic Exposure of Harlequin Ducks and Sea Otters
to Lingering *Exxon Valdez* Oil in Western Prince William Sound

Project 12120114-Q
Final Report

Study History: Harlequin ducks have been studied extensively in Prince William Sound following the *Exxon Valdez* oil spill, leading to one of the most thorough considerations of wildlife population injury and recovery following a major oil spill ever undertaken. These efforts have included population monitoring by the U.S. Fish and Wildlife Service and the Alaska Department of Fish and Game, as well as a series of directed research projects designed to elucidate the process of, and constraints to, population recovery. These studies demonstrated that harlequin ducks were exposed to lingering oil over a much longer time frame (i.e., through at least 2011, 22 years following the spill) than expected at the time of the spill, based on elevated levels of cytochrome P4501A induction in birds from oiled areas. In addition, several lines of evidence suggested that direct population injury persisted through at least 1998. Specifically, female winter survival probabilities were found to differ between oiled and unoiled areas, and densities were shown to be lower in oiled than unoiled areas after accounting for habitat-related effects. More recent data have indicated that female winter survival did not differ between oiled and unoiled sites during 2000-03, suggesting that direct effects of oil exposure on demographic properties had abated. Using demographic data, a population model was constructed to estimate timeline until recovery of numbers to pre-spill levels, which was projected to be 24 years post-spill or 2013. However, persistence of oil in the environment and evidence of exposure of harlequin ducks to that oil through 2011 has led to continued monitoring to evaluate the timeline of exposure. The current work was designed as another data point in that time series for 2013.

Abstract: For the first time since the 1989 *Exxon Valdez* oil spill, we found that average cytochrome P4501A induction (as measured by EROD activity) during March 2013 was not elevated in wintering harlequin ducks captured in areas of Prince William Sound oiled by the spill, relative to those captured in unoiled areas. Another metric of oil exposure (the incidence of individuals with elevated cytochrome P4501A induction) was consistent with this result, as it also showed similar values between oiled and unoiled areas. We interpret these findings to indicate that exposure of harlequin ducks to residual *Exxon Valdez* oil abated within 24 years after the original spill. These findings follow results from 2011, which showed reductions since 2009 in these metrics on oiled areas relative to unoiled and hence progress towards abatement of exposure, despite continued differences between areas in 2011,. The data presented in this report add to a growing body of literature indicating that persistence of oil in the environment, and exposure of wildlife to that oil, can occur over much longer time frames than previously assumed. These data may be used to define the duration of exposure (nearly two and a half decades) for one of the wildlife species most likely to suffer extended exposure.

Key Words: biomarker, cytochrome P4501A, *Exxon Valdez* oil spill, harlequin ducks, *Histrionicus histrionicus*, oil exposure, Prince William Sound, recovery.

Project Data: Data will be kept in digital format (MS Excel) at the Alaska Science Center, U.S. Geological Survey, Anchorage, Alaska.

Citation: Esler, D. 2013. Long-term Monitoring: Lingering Oil Evaluating Chronic Exposure of Harlequin Ducks and Sea Otters to Lingering *Exxon Valdez* Oil in Western Prince William Sound. *Exxon Valdez* Oil Spill Trustee Council Restoration Project Final Report (Project 12120114-Q), Pacific Wildlife Foundation and Centre for Wildlife Ecology, Simon Fraser University, Delta, British Columbia, Canada.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

EXECUTIVE SUMMARY

Extensive research and monitoring supported by the *Exxon Valdez* Oil Spill Trustee Council has led to a thorough understanding of the response of harlequin duck populations to the 1989 *Exxon Valdez* spill, and the process of (and constraints to) recovery. The information presented in this report adds to that body of work.

Induction of cytochrome P4501A (CYP1A) in vertebrates occurs in response to exposure to a limited number of compounds, including polycyclic aromatic hydrocarbons such as those found in crude oil. Because CYP1A induction is both specific and sensitive, it has been used to evaluate exposure to inducing compounds in many cases of environmental contamination, including that of the *Exxon Va*ldez oil spill. Elevated CYP1A has been demonstrated in several species in areas of Prince William Sound oiled by the *Exxon Valdez* spill relative to unoiled areas, including harlequin ducks.

In this study, CYP1A induction was determined by measuring hepatic 7-ethoxyresorufin-O-deethylase (EROD) activity, which is a well-established method and is the same approach used in earlier *Exxon Valdez* studies and in similar studies of harlequin ducks and other sea ducks elsewhere. During March 2013, we captured 25 harlequin ducks in oiled areas of Prince William Sound and 25 in unoiled areas. Small liver biopsies were surgically removed from each individual, frozen immediately in liquid nitrogen, and subsequently shipped to the University of California Davis for EROD analysis.

We found that CYP1A induction was not related to area, with average (pmol/min/mg ± SE) EROD activity of 17.8 (± 3.0) in oiled areas and 27.7 (± 5.9) in unoiled areas. This represents the first occasion since sampling was initiated in 1998 that CYP1A induction was not statistically higher in oiled areas than unoiled areas. This critical result follows the observation during 2011 that, although CYP1A induction was higher on oiled areas, the magnitude of the difference was reduced relative to previous years (1998 to 2009). We also considered the incidence of elevated exposure (defined as the number of individuals with EROD activity ≥ 2 times the average on unoiled areas for that year); for 2013 samples, we found that 4% of individuals captured in oiled areas had elevated EROD, compared to 12% in unoiled areas. As in previous years, we found that attributes of individuals (age, sex, and mass) were not related to variation in EROD.

We interpret these results to indicate that harlequin ducks were no longer exposed to residual *Exxon Valdez* oil as of March 2013, 24 years after the spill. Additional sampling in 2014 to confirm this finding is recommended.

This work adds to the body of literature evaluating cytochrome P4501A induction in several nearshore vertebrates in Prince William Sound, and defines the timeline over which exposure to lingering oil was evident for a species particularly vulnerable to long-term exposure.

INTRODUCTION

Effects of the 1989 *Exxon Valdez* oil spill on wildlife populations and communities in Prince William Sound, Alaska have been intensively studied, to document the process and timeline of population and ecosystem recovery. As part of that research, spatial and temporal extents of wildlife exposure to lingering *Exxon Valdez* oil have been inferred from indicators of induction of certain members of the cytochrome P450 1 gene subfamily (CYP1A). Vertebrate CYP1A genes are induced by larger polycyclic aromatic hydrocarbons (PAHs), including those found in crude oil, and halogenated aromatic hydrocarbons, including planar polychlorinated biphenyls (PCBs) and polychlorinated dibenzo-*p*-dioxins and difurans (Payne et al. 1987, Goksøyr 1995, Whitlock 1999). Because CYP1A is strongly induced by a limited number of compounds, it can be a particularly useful biomarker for evaluating exposure to those chemicals (Whyte et al. 2000). Although CYP1A induction does not necessarily indicate deleterious effects on individuals or populations (Lee and Anderson 2005), elevated CYP1A levels indicate exposure to inducing compounds and, hence, at least the potential for associated toxic consequences, including subtle effects that may be difficult to detect in nature (Carls et al. 2005). Therefore, indicators of CYP1A have been part of many considerations of environmental effects of contamination, including those associated with the *Exxon Valdez* oil spill.

Indicators of induction of CYP1A mRNA, protein or activity have been used routinely to evaluate exposure to PAHs, PCBs, and dioxins in fish (Stegeman et al. 1986, Gooch et al. 1989, Goksøyr 1995, Spies et al. 1996, Marty et al. 1997, Woodin et al. 1997, Collier et al. 1996, Wiedmer et al. 1996, Jewett et al. 2002, Carls et al. 2005). Although such studies are less common for birds and mammals, indicators of CYP1A levels have been used successfully as biomarkers of exposure of these taxa to inducing compounds, including PAHs (Lee et al. 1985, Peakall et al. 1989, Rattner et al. 1994, Trust et al. 1994; Ben-David et al. 2001; Miles et al. 2007; Esler et al. 2010; Esler et al. 2011; Flint et al. 2012).

In the case of the *Exxon Valdez* oil spill, indicators of CYP1A induction have been used to examine exposure to lingering oil for a number of vertebrates (e.g., Trust et al. 2000, Jewett et al. 2002; Esler et al. 2010; Esler et al. 2011). These studies demonstrated that, within Prince William Sound, CYP1A expression levels in many species were higher in areas oiled by the *Exxon Valdez* spill relative to unoiled areas nearly a decade after the spill. The authors of these studies concluded that oil remaining in the environment, particularly in intertidal areas, was encountered and ingested by some nearshore vertebrates. This conclusion is consistent with confirmation of the occurrence of residual *Exxon Valdez* oil in intertidal sediments of Prince William Sound during the same period in which elevated CYP1A was indicated (Short et al. 2004), as well as calculations that intertidal-foraging vertebrates would be likely to encounter lingering oil repeatedly through the course of a year (Short et al. 2006, Bodkin et al. 2012).

Harlequin ducks (*Histrionicus histrionicus*) were one of the species showing indication of elevated CYP1A induction in oiled areas of Prince William Sound relative to unoiled

areas (Trust et al. 2000; Esler et al. 2010). Harlequin ducks are marine birds that spend most of their annual cycle in intertidal and shallow subtidal zones of temperate and subarctic areas of the Pacific coast of North America (Robertson and Goudie 1999). They are common in Prince William Sound during the nonbreeding season (average of 14,500 individuals between 1990 and 2005; McKnight et al. 2006), and are at higher risk of exposure to residual *Exxon Valdez* oil than many other seabirds, given their exclusive occurrence in nearshore habitats where a disproportionate amount of oil was deposited (Galt et al. 1991, Wolfe et al. 1994) and where lingering oil has remained (Hayes and Michel 1999, Short et al. 2004).

In addition to higher likelihood of exposure, a number of natural history and life history characteristics make harlequin duck individuals and populations particularly sensitive to oil pollution (Esler et al. 2002). These include a diet consisting of invertebrates that live on or in nearshore sediments, a life history strategy predicated on high survival rates, and a small body size, relative to other sea ducks, that may limit their flexibility when faced with increased energetic demands. Consistent with these sensitivities to effects of oil contamination, demographic problems were observed in oiled areas of Prince William Sound during the same period in which elevated CYP1A was indicated, including reductions in population trends (Rosenberg and Petrula 1998), densities (Esler et al. 2000a), and female survival (Esler et al. 2000b) relative to unoiled areas. It was concluded that continued exposure to lingering oil was likely a constraint on population recovery (Esler et al. 2002). A population model built with available demographic information was used to estimate the timeline to numeric population recovery, which was estimated to be 24 years after the *Exxon Valdez* spill, or the year 2013 (Iverson and Esler 2010).

Because of the history of elevated indicators of CYP1A induction (Trust et al. 2000; Esler et al. 2010), continued occurrence of lingering oil in intertidal habitats where harlequin ducks occur (Short et al. 2004), and vulnerability of harlequin ducks to effects of oil exposure (Esler et al. 2002), the present study was conducted to follow up on research describing elevated biomarkers of CYP1A in this species. In previous studies, Trust et al. (2000) and Esler et al. 2010 found that average CYP1A expression levels, measured by hepatic 7-ethoxyresorufin-*o*-deethylase (EROD) activity, were significantly higher in wintering harlequin ducks captured in areas oiled by the *Exxon Valdez* spill than those captured in nearby unoiled areas through 2009. In 2011, average EROD activity was higher in harlequin ducks from oiled areas than those from unoiled, although the magnitude of the difference was smaller than during previous sample years (Esler 2011). The primary objective for the present study was to add to the monitoring timeline during 2013, 24 years after the *Exxon Valdez* oil spill, to evaluate whether differences in EROD activity persisted.

In addition to assessment of temporal variation, potential effects of individual attributes (age, sex, and body mass) on variation in CYP1A induction also were considered. Age, sex, and season have been shown to affect CYP1A induction in some fish (Sleiderink et al. 1995, Goksøyr and Larsen 1991, Lindstrom-Seppa and Stegeman 1995, Whyte et al.

2000, Kammann et al. 2005), and thus these factors should be accounted for when evaluating sources of variation in CYP1A induction (Lee and Anderson 2005).

METHODS

Capture and Sample Collection

To facilitate comparisons, the present study closely followed the design and procedures of previous work (Trust et al. 2000; Esler et al. 2010). We captured wintering harlequin ducks using a modified floating mist net (Kaiser et al. 1995) during March 2013. Birds were captured in a number of areas oiled during the *Exxon Valdez* spill, including Crafton Island (60.5º N, 147.9º W), Green Island (60.3º N, 147.4º W), Foul Pass (60.5º N, 147.6º W), and Herring Bay (60.5º N, 147.7º W). Also, birds were captured on nearby northwestern Montague Island (60.3º N, 147.3º W), which was not oiled and thus was considered a reference site. Harlequin ducks in Prince William Sound exhibit high site fidelity during winter, with 94% remaining all winter on the same island or coastline region where they were originally captured and only 2% moving between oiled and unoiled areas (Iverson and Esler 2006). We assume that this level of movement had little influence on our ability to draw inferences about differences in EROD activity between areas. Captured birds were placed in portable pet carriers and transported by skiff to a chartered research vessel for processing. Each individual was marked with a uniquely-numbered, U.S. Fish and Wildlife metal tarsus band; the band number was used to identify the data and samples for that individual. Sex of each bird was determined by plumage and cloacal characteristics, and age class was determined by the depth of the bursa of Fabricius for females and bursal depth and plumage characteristics for males (Mather and Esler 1999, Smith et al. 1998). Age class was summarized as either hatch-year (HY), i.e., hatched the previous breeding season, or after-hatch-year (AHY). Numbers of individuals used in analyses of CYP1A induction are indicated in Table 1, by age class, sex, and area (oiled versus unoiled).

Small (< 0.5 g) liver biopsies were surgically removed by a veterinarian from each harlequin duck while they were under general anesthesia using vaporized and inhaled Isoflurane. Once removed, liver samples were immediately placed into a labeled cryovial and frozen in liquid nitrogen. All samples were maintained in liquid nitrogen or a -80° C freezer until they were shipped to the lab in liquid nitrogen.

Laboratory Analyses

CYP1A induction was determined by measuring hepatic 7-ethoxyresorufin-*o*-deethylase activity, which is a catalytic function principally of hydrocarbon-inducible CYP1A enzymes. In studies of captive harlequin ducks, EROD activity was confirmed to be significantly higher in birds chronically ingesting weathered Prudhoe Bay crude oil, compared to controls (Esler 2008). Similarly, oil-dosed Steller's eiders (*Polysticta stelleri*), another sea duck, had roughly 4-fold increased EROD activity compared to controls (Miles et al. 2007). EROD activity analysis procedures followed standard

methods used in previous studies, described in detail by Miles et al. (2007). The measure of EROD activity is expressed in picomoles per minute per milligram of protein, i.e., pmol/min/mg protein.

Statistical Analyses

Variation in EROD activity was analyzed in relation to capture location and individual attributes for birds captured during March 2013. Our primary interest was to determine whether area (oiled versus unoiled) explained variation in EROD activity, after accounting for any effects of age class, sex, and body mass. Least squares general linear models (GLM) were used to estimate variation explained by each of a candidate set of models that included different combinations of variables of interest, and an information-theoretic approach was used for model selection and inference (Burnham and Anderson 2002) in which support for various model configurations is contrasted using Akaike's Information Criterion (AIC). Age, sex, and body mass variables (which we termed *individual attributes*) were included or excluded as a group, i.e., models either included all of these variables or none of them. We used singular and additive combinations of area and individual attribute effects, resulting in a candidate model set including: (1) EROD = area; (2) EROD = individual attributes; and (3) EROD = area + individual attributes. We also included a null model, which consisted of estimates of a mean and variance across all of the data; support for the null model would indicate that variables considered in other candidate models did not explain important variation in the response.

The model with the lowest AIC value corrected for small sample size (AIC_c) was considered to have the strongest support from the data among the models considered. Another metric, AIC_c weight (w), was calculated for each model; these sum to 1.0 across the entire model set and provide a measure of relative support for candidate models. The variables included in the models with highest support are considered to explain important variation in the response. Parameter likelihoods, which are the sums of w for all models including a given parameter, indicate the relative support for that variable, taking into account model uncertainty. Parameter likelihoods close to 1 indicate strong support. Finally, weighted parameter estimates and associated unconditional standard errors were calculated, which are estimates of the size, direction, and associated variation of effects of variables after accounting for model uncertainty.

RESULTS

Variation in EROD activity of harlequin ducks captured in March 2013 was not strongly associated with any of the explanatory variables. The best supported model included only the parameter indicating whether harlequin ducks were captured from oiled or unoiled areas (w = 0.43; Table 2). However, support for that model was virtually indistinguishable from the null model (w = 0.43), which indicated that none of the explanatory variables was strongly supported. In addition, average EROD activity was lower on oiled areas than on unoiled (Table 3; Figure 1); therefore, the moderate support for an area effect was in the opposite direction than expected under a hypothesis of

continued oil exposure. As in previous years (Esler et al. 2010, Esler 2011), the group of individual attribute variables did not explain meaningful variation in EROD, as both models including individual attributes had small *w* and received less support than the null model (i.e., had larger AIC_c values; Table 2).

Parameter likelihood values also supported the inference that none of the variables had strong value for explaining variation in March 2013 EROD activity. The area parameter was moderately supported, with a parameter likelihood of 0.49 (Table 3). However, the weighted parameter estimate indicated that EROD activity was slightly higher on unoiled areas than on oiled areas (Figure 1), by an average of 4.8 pmol/min/mg protein (Table 3). The corresponding unconditional standard error for the area variable (6.5; Table 3) was larger than the parameter estimate, further indicating the lack of strong support for an area effect. Parameter likelihood values for individual attributes were small, and the weighted parameter estimates were smaller than the corresponding unconditional standard errors (Table 3), indicating that they did not have strong explanatory value.

Several measures of CYP1A induction suggested that the degree and incidence of oil exposure on oiled areas was indistinguishable from, or lower than, that on unoiled areas in 2013, which is in stark contrast to previous years. First, average (pmol/min/mg ± SE) EROD activity on oiled areas was 17.8 (± 3.0) in 2013, compared to point estimates > 40 pmol/min/mg in the previous 4 sampling periods, in contrast to consistent estimates of EROD activity in unoiled areas over that same period (Figure 1). Similarly, when data were scaled relative to the reference values from birds captured on unoiled areas, findings from 2013 stand out as being the first time since sampling was initiated that EROD activity on oiled areas was similar to or lower than that on unoiled areas (Figure 2). Finally, the incidence of elevated EROD activity was 4% of individuals from oiled areas in 2013 (Figure 3), which was lower than estimates from oiled areas in previous years and similar to results from unoiled areas across all years.

DISCUSSION

We found that hepatic CYP1A levels in harlequin ducks captured in March 2013, based on EROD activity, were similar between areas oiled during the *Exxon Valdez* spill and in nearby unoiled areas. In fact, the point estimate of average EROD activity was slightly lower in oiled areas than in unoiled. This March 2013 sample constitutes the first time since initiation of harlequin duck CYP1A sampling in 1998 that EROD activity has not been higher in oiled areas than in unoiled areas of Prince William Sound. We interpret this to indicate that harlequin ducks are no longer exposed to residual oil from the 1989 *Exxon Valdez* spill. The timeline over which the observed return to baseline has occurred (24 years) is longer than anticipated at the time of the spill, given conventional assumptions at that time about duration of bioavailability of spilled oil (Peterson et al. 2003). Abatement of exposure to lingering oil implies that any potential direct, deleterious effects on individuals or populations also must have ceased. We recognize that evidence of exposure through 2011 could not necessarily be inferred to indicate ongoing damage (Lee and Anderson 2005), but absence of exposure in 2013 assumes that

any remaining damage is due to demographic or toxicological effects of previous exposure.

The observation of similar average EROD activity between oiled and unoiled areas in 2013 follows observations in 2011 of reductions in both average and incidence of CYP1A induction of harlequin ducks in oiled areas, compared to previous years. This suggests that the degree of exposure was declining at that time. This pattern of declines in metrics of exposure also was observed in Barrow's goldeneyes (*Buchephala islandica*), another nearshore-dwelling sea duck, although evidence of lack of exposure by all metrics occurred earlier in this species, by 2009 (Esler et al. 2011).

Differential CYP1A induction between oiled and unoiled areas has been described for other vertebrates in Prince William Sound, including Barrow's goldeneyes (Trust et al. 2000; Esler et al. 2011), adult pigeon guillemots (*Cepphus columba*; Golet et al. 2002), river otters (*Lontra canadensis*; Bowyer et al. 2003), and two demersal fishes (Jewett et al. 2002), masked greenlings (*Hexagrammos octogrammus*) and crescent gunnels (*Pholis laeta*). This body of evidence strongly supports the conclusion that harlequin ducks, along with other nearshore vertebrates, were being exposed to CYP1A-inducing compounds in areas of Prince William Sound, Alaska that received oil during the *Exxon Valdez* spill. It also demonstrates that the timeline for cessation of exposure varies across species, with harlequin ducks being one of the last to show cessation of exposure, likely due to natural history features that enhance risk of exposure (Esler et al. 2002).

Some authors have questioned the source of CYP1A inducing compounds in Prince William Sound (Harwell and Gentile 2006), recognizing that there may be multiple CYP1A-inducing compounds from multiple sources within a given area (Lee and Anderson 2005). Several authors (Page et al. 1996, 1997, Boehm et al. 2001, Harwell and Gentile 2006) have argued that non-*Exxon Valdez* sources of PAHs are more abundant and more likely to induce CYP1A responses than residual *Exxon Valdez* oil. However, the spatial correspondence between elevated CYP1A induction and history of contamination during the *Exxon Valdez* oil spill strongly suggests causation. Also, other studies have indicated that PAHs in the areas where elevated CYP1A was observed in vertebrates are predominately from the *Exxon Valdez* spill (Short et al. 2004), supporting the inference that *Exxon Valdez* oil was the inducing agent. Recent studies have indicated that sites with residual *Exxon Valdez* oil had bioavailable PAHs that elicited CYP1A induction when experimentally injected into fish (Springman et al. 2008). Other potential CYP1A inducers, specifically PCBs, were very low and below concentrations that would induce CYP1A induction, consistent with broad-scale atmospheric deposition (Short et al. 2008). In addition, Trust et al. (2000) and Ricca et al. (2010) considered the potential role of PCBs in observed CYP1A induction in sea ducks in Prince William Sound and found that plasma concentrations were very low and generally were not related to EROD activity. In addition, Short et al. (2006) calculated that, given the distribution of residual *Exxon Valdez* oil through 2003, benthic foraging vertebrates were likely to encounter lingering oil, further suggesting that residual *Exxon Valdez* oil was the inducing compound. Finally, our results indicating declines in CYP1A induction in both harlequin ducks and Barrow's goldeneye over time, and subsequent return to baseline, were

consistent with exposure to a source declining in availability over time, as would be expected with *Exxon Valdez* oil, rather than compounds predicted to be constant over time such as atmospheric PCBs or oil from natural seeps.

Vertebrates that inhabit intertidal and shallow subtidal environments, particularly those that consume benthic organisms, were most likely to have prolonged, elevated CYP1A (Esler et al. 2002). This is presumably due, in part, to that fact that intertidal areas of Prince William Sound received a large portion of the spilled *Exxon Valdez* oil (Galt et al. 1991, Wolfe et al. 1994) and sequestered lingering oil a decade or more post-spill (Hayes and Michel 1999, Short et al. 2004). Also, because certain molluscan invertebrates have a limited capacity to metabolize PAHs (e.g., Chaty et al. 2004) and are known to ingest and accumulate PAHs (Short and Harris 1996, Fukuyama et al. 2000, Rust et al. 2004), predators such as harlequin ducks may be more likely to ingest PAHs with their prey. Also, invertivores disturb sediment during foraging, which is a potential mechanism for release of hydrocarbons and ingestion (Bodkin et al. 2012).

Consistent with predictions of increased exposure to residual oil and vulnerability to subsequent effects, as well as empirical evidence of exposure (Trust et al. 2000, Bodkin et al. 2002, Esler et al. 2010, Esler et al. 2011), invertivorous, nearshore-dwelling vertebrates have been shown to have population demographic attributes outside of the normal range during the period since the *Exxon Valdez* oil spill. For example, sea otter numbers in heavily oiled regions of Prince William Sound were well below estimates of pre-spill numbers (Bodkin et al. 2002). Also, sea otter survival in oiled areas was depressed through at least 1998 (Monson et al. 2000). Similar evidence of post-spill demographic problems was described for harlequin ducks (Esler et al. 2002). Densities of wintering harlequin ducks in 1996 and 1997 were lower than expected in oiled areas of Prince William Sound, after accounting for effects of differing habitat (Esler et al. 2000a). Also, survival of wintering female harlequin ducks was lower in oiled areas than unoiled (Esler et al. 2000b) during 1995 to 1998. More recent estimates have indicated that harlequin duck survival during winters 2000 to 2003 did not differ between oiled and unoiled areas (Esler and Iverson 2010), suggesting that despite the evidence of continued exposure reported by Esler et al. (2010), oil-induced effects on demographic rates were diminishing. Given observed demographic rates, Iverson and Esler (2010) projected numeric population recovery would occur by approximately 2013.

In addition to potential relationships between oil exposure and demographic rates (Esler et al. 2002), more subtle effects at the suborganismal and molecular level are plausible. Rainbow trout (*Oncorhynchus mykiss*) showed increased mortality in response to viral challenge when they had been exposed to a CYP1A inducer (Springman et al. 2005). In mammals, CYP1A1 is known to activate PAH to toxic and mutagenic derivatives (Nebert et al. 2004). In birds, Trust et al. (1994) identified effects of PAHs on immune function and mixed-function oxygenase activity (e.g., EROD) in European starlings (*Sturnus vulgaris*). In controlled dose experiments, crude oil and PAHs have been linked to impaired reproduction, depressed weight gain, increased organ weight, increased endocrine activity, or mixed-function oxygenase activity in several avian taxa (Hoffman 1979, Naf et al. 1992, Peakall et al. 1980, Peakall et al. 1981). However, given the lack

of CYP1A induction observed in 2013, both lethal and sublethal direct effects of oil exposure can be considered to have ceased.

In summary, the EROD levels reported here provide evidence that CYP1A induction is similar between harlequin ducks from oiled areas and those from unoiled areas, which we conclude is due to lack of continued exposure to residual *Exxon Valdez* oil. This suggests the period of exposure of this species to lingering oil was between 22 and 24 years. We note that oil from other contamination events also has been reported to persist over long periods of time (Corredor et al. 1990, Burns et al. 1994, Vandermeulen and Singh 1994, Reddy et al. 2002, Peacock et al. 2005). We agree with Peterson et al. (2003) that the conventional paradigm that the duration of presence of residual oil and associated effects is limited to a few years should be abandoned and replaced with the recognition that these may occur over decades in certain, vulnerable species. We recommend that monitoring of indicators of CYP1A induction in harlequin ducks in Prince William continue for at least one more year, to confirm that EROD in oiled areas has returned to background levels.

ACKNOWLEDGEMENTS

This research was supported primarily by the *Exxon Valdez* Oil Spill Trustee Council. However, the findings and conclusions presented by the author are his own and do not necessarily reflect the views or position of the Trustee Council. This work was facilitated and conducted by many people, which is why I have used "we" rather than "I" throughout the other sections of the report. Those deserving thanks include those who helped with field work: Jon Brown, Pete Clarkson, Rian Dickson, Melissa Gabrielson, and Tim Bowman. Veterinary expertise during field work was provided by Drs. Malcolm McAdie and Gwen Myers. Thanks to Dean Rand and his crew of the motor vessel *Discovery* for safe and comfortable passage. Laboratory analyses were conducted through the collaboration of Keith Miles, Liz Bowen, Sarah Spring, Barry Wilson, and Jack Henderson. I also appreciate the institutional and logistical support provided by Dede Bohn, John Pearce, Kevin Sage, Brenda Ballachey, Connie Smith, Kim Kloecker, George Esslinger, Brian Uher-Koch, and Ian Semple.

LITERATURE CITED

Ben-David M., T. Kondratyuk, B. R. Woodin, P. W. Snyder, and J. J. Stegeman. 2001. Induction of cytochrome P4501A1 expression in captive river otters fed Prudhoe Bay crude oil: evaluation by immunohistochemistry and quantitative RT-PCR. Biomarkers 6: 218-235.

Bodkin, J. L., B. E. Ballachey, T. A. Dean, A. K. Fukuyama, S. C. Jewett, L. McDonald, D. H. Monson, C. E. O'Clair, and G. R. VanBlaricom. 2002. Sea otter population status and the process of recovery from the 1989 'Exxon Valdez' oil spill. Marine Ecology Progress Series 241: 237-253.

Bodkin, J. L., B. E. Ballachey, H. A. Coletti, G. G. Esslinger, K. A. Kloecker, S. D. Rice, J. A. Reed, and D. A. Monson. 2012. Long-term effects of the *Exxon Valdez* oil spill: sea otter foraging in the intertidal as a pathway of exposure to lingering oil. Marine Ecology Progress Series 447:273-287.

Boehm, P. D., D. S. Page, W. A. Burns, A. E. Bence, P. J. Mankiewicz, and J. S. Brown. 2001. Resolving the origin of the petrogenic hydrocarbon background in Prince William Sound, Alaska. Environmental Science and Technology 35: 471-479.

Bowyer, R. T., G. M. Blundell, M. Ben-David, S. C. Jewett, T. A. Dean, and L. K. Duffy. 2003. Effects of the *Exxon Valdez* oil spill on river otters: injury and recovery of a sentinel species. Wildlife Monographs 153: 1-53.

Burnham, K. P., and D. R. Anderson. 2002. Model selection and multimodel inference: a practical information theoretic approach. 2nd Edition. Springer-Verlag, New York.

Burns, K. A., S. D. Garrity, D. Jorissen, J. MacPherson, M. Stoelting, J. Tierney, and L. Yelle-Simmons. 1994. The Galeta oil spill. 2. Unexpected persistence of oil trapped in mangrove sediments. Estuarine, Coastal and Shelf Science 38: 349-364.

Carls, M. G., R. A. Heintz, G. D. Marty, and S. D. Rice. 2005. Cytochrome P4501A induction in oil-exposed pink salmon *Oncorhynchus gorbuscha* embryos predicts reduced survival potential. Marine Ecology Progress Series 301:253-265.

Chaty, S., F. Rodius, and P. Vasseur. 2004. A comparative study of the expression of CYP1A and CYP4 genes in aquatic invertebrate (freshwater mussel, *Unio tumidus*) and vertebrate (rainbow trout, *Oncorhynchus mykiss*). Aquatic Toxicology 69:81-93.

Collier, T. K., C. A. Krone, M. M. Krahn, J. E. Stein, S.-L. Chan, and U. Varanasi. 1996. Petroleum exposure and associated biochemical effects in subtidal fish after the *Exxon Valdez* oil spill. Pages 671–683 *in* S. D. Rice, R. B. Spies, D. A. Wolfe, and B. A. Wright, editors. Proceedings of the *Exxon Valdez* Oil Spill Symposium, Bethesda, Maryland. American Fisheries Society Symposium 18.

Corredor, J. E., J. M. Morell, and C. E. Castillo. 1990. Persistence of spilled crude oil in a tropical intertidal environment. Marine Pollution Bulletin 21: 385-388.

Esler, D. 2008. Quantifying temporal variation in Harlequin Duck cytochrome P4501A induction. *Exxon Valdez* Oil Spill Trustee Council Gulf Ecosystem Monitoring and Research Project Final Report (GEM Project 050777), Centre for Wildlife Ecology, Simon Fraser University, Delta, British Columbia, Canada.

Esler, D., T. D. Bowman, T. A. Dean, C. E. O'Clair, S. C. Jewett, and L. L. McDonald. 2000a. Correlates of harlequin duck densities during winter in Prince William Sound, Alaska. Condor 102: 920-926.

Esler, D., J. A. Schmutz, R. L. Jarvis, and D. M. Mulcahy. 2000b. Winter survival of adult female harlequin ducks in relation to history of contamination by the Exxon Valdez oil spill. Journal of Wildlife Management 64: 839-847.

Esler, D., T. D. Bowman, K. Trust, B. E. Ballachey, T. A. Dean, S. C. Jewett, and C. E. O'Clair. 2002. Harlequin duck population recovery following the Exxon Valdez oil spill: progress, process, and constraints. Marine Ecology Progress Series 241: 271-286.

Esler, D., and S. A. Iverson. 2010. Female harlequin duck winter survival 11 to 14 years after the *Exxon Valdez* oil spill. Journal of Wildlife Management 74:471-478.

Esler, D., K. A. Trust, B. E. Ballachey, S. A. Iverson , T. L. Lewis, D. J. Rizzolo, D. M. Mulcahy, A. K. Miles, B. R. Woodin, J. J. Stegeman, J. D. Henderson, and B. W. Wilson. 2010. Cytochrome P4501A biomarker indication of oil exposure in harlequin ducks up to 20 years after the Exxon Valdez oil spill. Environmental Toxicology and Chemistry 29:1138-1145.

Esler, D. 2011. Nearshore synthesis: sea otters and sea ducks (amendment). *Exxon Valdez* Oil Spill Trustee Council Restoration Project Final Report (Project 11100808), Centre for Wildlife Ecology, Simon Fraser University, Delta, British Columbia, Canada.

Esler, D., B. E. Ballachey, K. A. Trust, S. A. Iverson, J. A. Reed, A. K. Miles, J. D. Henderson, B. W. Wilson, B. R. Woodin, J. R. Stegeman, M. McAdie, and D. M. Mulcahy. 2011. Cytochrome P4501A biomarker indication of the timeline of chronic exposure of Barrow's goldeneye to residual *Exxon Valdez* oil. Marine Pollution Bulletin 62:609-614.

Flint, P. L., J. L. Schamber, K. A. Trust, A. K. Miles, J. D. Henderson, and B. W. Wilson. 2012. Chronic hydrocarbon exposure of harlequin ducks in areas affected by the *Selendang Ayu* oil spill at Unalaska Island, Alaska. Environmental Toxicology and Chemistry 31:2828-2831.

Fukuyama A. K., G. Shigenaka, and R. Z. Hoff. 2000. Effects of residual *Exxon Valdez* oil on intertidal *Protothaca staminea*: mortality, growth, and bioaccumulation of hydrocarbons in transplanted clams. Marine Pollution Bulletin 40: 1042-1050.

Galt, J. A., W. J. Lehr, and D. L. Payton. 1991. Fate and transport of the *Exxon Valdez* oil spill. Environmental Science and Technology 25**:** 202-209.

Goksøyr, A. 1995. Use of cytochrome P450 1A (CYP1A) in fish as a biomarker of aquatic pollution. Archives of Toxicology Supplement 17: 80-95.

Goksøyr, A., and H. E. Larsen. 1991. The cytochrome P450 system of the Atlantic salmon (*Salmo salar*): I. Basal properties and induction of P450 1A1 in liver of immature and mature fish. Fish Physiology and Biochemistry 9: 339-349.

Golet, G. H., P. E. Seiser, A. D. McGuire, D. D. Roby, J. B. Fischer, K. J. Kuletz, D. B. Irons, T. A. Dean, S. C. Jewett, and S. H. Newman. 2002. Long-term direct and indirect effects of the *Exxon Valdez* oil spill on pigeon guillemots in Prince William Sound, Alaska. Marine Ecology Progress Series 241: 287-304.

Gooch, J. W., A. A. Elskus, P. J. Kloepper-Sams, M. E. Hahn, and J. J. Stegeman. 1989. Effects of ortho and non-ortho substituted polychlorinated biphenyl congeners on the hepatic monooxygenase system in scup (*Stenotomus chrysops*). Toxicology and Applied Pharmacology 98: 422-433.

Harwell, M. A., and J. H. Gentile. 2006. Ecological significance of residual exposures and effects from the *Exxon Valdez* oil spill. Integrated Environmental Assessment and Management 2: 204-246.

Hayes, M. O. and J. Michel. 1999. Factors determining the long-term persistence of *Exxon Valdez* oil in gravel beaches. Marine Pollution Bulletin 38: 92-101.

Hoffman, D.J. 1979. Embryotoxic and teratogenic effects of petroleum hydrocarbons in mallards (*Anas platyrhynchos*). Journal of Toxicology and Environmental Health 5:835-844.

Iverson, S. A., and D. Esler. 2006. Site fidelity and the demographic implications of winter movements by a migratory bird, the harlequin duck. Journal of Avian Biology 37: 219-228.

Iverson, S. A., and D. Esler. 2010. Harlequin duck population dynamics following the 1989 Exxon Valdez oil spill: assessing injury and projecting a timeline to recovery. Ecological Applications 20:1993-2006.

Jewett, S. C., T. A. Dean, B. R. Woodin, M. K. Hoberg, and J. J. Stegeman. 2002. Exposure to hydrocarbons ten years after the *Exxon Valdez*: evidence from cytochrome P4501A expression and biliary FACs in nearshore demersal fishes. Marine Environmental Research 54: 21-48.

Kaiser, G. W., A. E. Derocher, S. C. Crawford, M. J. Gill, and I. A. Manley. 1995. A capture technique for marbled murrelets in coastal inlets. Journal of Field Ornithology 66: 321-333.

Kammann, U., T. Lang, M. Vobach, and W. Wosniok. 2005. Ethoxyresorufin-O-deethylase (EROD) Activity in Dab (Limanda limanda) as Biomarker for Marine Monitoring. Environmental Science and Pollution Research 12: 140-145.

Lee, R. F., and J. W. Anderson. 2005. Significance of cytochrome P450 system responses and levels of bile fluorescent aromatic compounds in marine wildlife following oil spills. Marine Pollution Bulletin 50: 705-723.

Lee, Y.-Z., F. A. Leighton, D. B. Peakall, R. J. Norstrom, P. J. O'Brien, J. F. Payne, and A. D. Rahimtul. 1985. Effects of ingestion of Hibernia and Prudhoe Bay crude oils on hepatic and renal mixed-function oxidase in nestling herring gulls (Larus argentatus). Environmental Research 36: 248-255.

Lindstrom-Seppa, P. and J. J. Stegeman. 1995. Sex differences in cytochrome P4501A induction by environmental exposure and b-naphthoflavone in liver and extrahepatic organs of recrudescent winter flounder. Marine Environmental Research 39: 219-223.

Marty G. D, J. W. Short, D. M. Dambach, N. H. Willits, R. A. Heintz, S. D. Rice, J. J. Stegeman, and D. E. Hinton. 1997. Ascites, premature emergence, increased gonadal cell apoptosis, and cytochrome P4501A induction in pink salmon larvae continuously exposed to oil-contaminated gravel during development. Canadian Journal of Zoology 75: 989-1007.

Mather, D. D., and D. Esler. 1999. Evaluation of bursal depth as an indicator of age class of harlequin ducks. Journal of Field Ornithology 70: 200-205.

McKnight, A., K. M. Sullivan, D. B. Irons, S. W. Stephensen, and S. Howlin. 2006. Marine bird and sea otter population abundance of Prince William Sound, Alaska: trends following the *T/V Exxon Valdez* oil spill, 1989-2005. *Exxon Valdez* Oil Spill Restoration Project Final Report (Restoration Projects 040159/050751), U.S. Fish and Wildlife Service, Anchorage, Alaska.

Miles, A. K., P. L. Flint, K. A. Trust, M. A. Ricca, S. E. Spring, D. E. Arietta, T. Hollmén, and B. W. Wilson. 2007. Polycyclic aromatic hydrocarbon exposure in Steller's eiders (*Polysticta stelleri*) and harlequin ducks (*Histrionicus histrionicus*) in the eastern Aleutian Islands, Alaska. Environmental Toxicology and Chemistry 26: 2694-2703.

Monson, D. H., D. F. Doak, B. E. Ballachey, A. Johnson, and J. L. Bodkin. 2000. Long-term impacts of the *Exxon Valdez* oil spill on sea otters, assessed through age-dependent mortality patterns. Proceedings of the National Academy of Sciences 97: 6562-6567.

Naf, C., D. Broman, and B. Brunstrom. 1992. Distribution and metabolism of polycyclic aromatic hydrocarbons (PAHs) injected into eggs of chicken (*Gallus domesticus*)

and common eider duck (*Somateria mollissima*). Environmental Toxicology and Chemsitry 11:1653-1660.

Nebert, D. W., T. P. Dalton, A. B. Okey, and F. J. Gonzalez. 2004. Role of aryl hydrocarbon receptor-mediated induction of the CYP1 enzymes in environmental toxicity and cancer. Journal of Biological Chemistry 279: 23847-23850.

Page, D. S., P. D. Boehm, G. S. Douglas, A. E. Bence, W. A. Burns, and P. J. Mankiewicz. 1996. The natural petroleum hydrocarbon background in subtidal sediments of Prince William Sound, Alaska, USA. Environmental Toxicology and Chemistry 15: 1266-1281.

Page, D. S., P. D. Boehm, G. S. Douglas, A. E. Bence, W. A. Burns, and P. J. Mankiewicz. 1997. An estimate of the annual input of natural petroleum hydrocarbons to seafloor sediments of Prince William Sound, Alaska. Marine Pollution Bulletin 34: 744-749.

Payne, J. F., L. L. Fancey, A. D. Rahimtula, and E. L. Porter. 1987. Review and perspective on the use of mixed-function oxygenase enzymes in biological monitoring. Comparative Biochemistry and Physiology 86C: 233–245.

Peacock, E. E., R. K. Nelson, A. R. Solow, J. D. Warren, J. L. Baker, and C. M. Reddy. 2005. The West Falmouth oil spill: 100 kg of oil persists in marsh sediments. Environmental Forensics 6: 273-281.

Peakall, D.B., D.S. Miller, R.G. Butler, W.R. Kinter, and D.J. Hallett. 1980. Effects of ingested crude oil on black guillemots: A combined field and laboratory study. Ambio 9:28-30.

Peakall, D.B., J. Tremblay, D.S. Miller, and W.B. Kinter. 1981. Endocrine dysfunction in seabirds caused by ingested oil. Environmental Research 24:6-14.

Peakall, D. B., R. J. Norstrom, D. A. Jeffrey, and F. A. Leighton. 1989. Induction of hepatic mixed function oxidases in the herring gull (*Larus argentatus*) by Prudhoe Bay crude oil and its fractions. Comparative Biochemistry and Physiology 94C: 461-463.

Peterson, C. H., S. D. Rice, J. W. Short, D. Esler, J. L. Bodkin, B. A. Ballachey, and D. B. Irons. 2003. Long-term ecosystem response to the *Exxon Valdez* oil spill. Science 302: 2082-2086.

Rattner, B. A., J. S. Hatfield, M. J. Melancon, T. W. Custer, and D. E. Tillitt. 1994. Relation among cytochrome P450, AH-Active PCB congeners and dioxin equivalents in pipping black-crowned night-heron embryos. Environmental Toxicology and Chemistry 13: 1805-1812.

Reddy, C. M., T. I. Eglinton, A. Hounshell, H. K. White, L. Xu, R. B. Gaines, and G. S. Frysinger. 2002. The West Falmouth oil spill after thirty years: The persistence of petroleum hydrocarbons in marsh sediments. Environmental Science and Technololgy 36: 4754-4760.

Ricca, M. A., A. K. Miles, B. E. Ballachey, J. L. Bodkin, D. Esler, and K. A. Trust. 2010. PCB exposure in sea otters and harlequin ducks in relation to history of contamination by the *Exxon Valdez* oil spill. Marine Pollution Bulletin 60:861-872.

Robertson G. J. and R. I. Goudie. 1999. Harlequin Duck. The Birds of North America. The American Ornithologists Union, Washington, DC, and The Academy of Natural Sciences, Philadelphia, PA.

Rosenberg D. H. and M. J. Petrula. 1998. Status of harlequin ducks in Prince William Sound, Alaska after the *Exxon Valdez* oil spill, 1995-1997. *Exxon Valdez* oil spill restoration project final report, No. 97427. Alaska Department of Fish and Game, Division of Wildlife Conservation, Anchorage, Alaska.

Rust, A. J., R. M. Burgess, B. J. Brownawell, and A. E. McElroy. 2004. Relationship between metabolism and bioaccumulation of Benzo[a]pyrene in benthic invertebrates. Environmental Toxicology and Chemistry 23: 2587-2593.

Short, J. W., and P. M. Harris. 1996. Petroleum hydrocarbons in caged mussels deployed in Prince William Sound after the *Exxon Valdez* oil spill. Pages 29–39 *in* S. D. Rice, R. B. Spies, D. A. Wolfe, and B. A. Wright, editors. Proceedings of the *Exxon Valdez* Oil Spill Symposium, Bethesda, Maryland. American Fisheries Society Symposium 18.

Short, J. W., M. R. Lindeberg, P. M. Harris, J. M. Maselko, J. J. Pella, and S. D. Rice. 2004. Estimate of oil persisting on the beaches of Prince William Sound 12 years after the Exxon Valdez oil spill. Environmental Science & Technology 38: 19-25.

Short, J. W., J. M. Maselko, M. R. Lindeberg, P. M. Harris, and S. D. Rice. 2006. Vertical distribution and probability of encountering intertidal Exxon Valdez oil on shorelines of three embayments within Prince William Sound. Environmental Science and Technology 40: 3723-3729.

Short, J. W., K. R. Springman, M. R. Lindeberg, L. G. Holland, M. L. Larsen, C. A. Sloan, C. Khan, P. V. Hodson, and S. D. Rice. 2008. Semipermeable membrane devices link site-specific contaminants to effects: Part II – a comparison of lingering Exxon Valdez oil with other potential sources of CYP1A inducers in Prince William Sound, Alaska. Marine Environmental Research 66: 487-498.

Sleiderink, H. M., I. Oostingh, A. Goksøyr, and J. P. Boon. 1995. Sensitivity of cytochrome P450 1A induction in dab (*Limanda limanda*) of different age and sex

as a biomarker for environmental contaminants in the southern North Sea. Archives of Environmental Contamination and Toxicology 28: 423-430.

Smith, C. M., R. I. Goudie, and F. Cooke. 1998. Winter age ratios and the assessment of recruitment of Harlequin Ducks. Waterbirds 24: 39-44.

Spies, R. B., J. J. Stegeman, D. E. Hinton, B. Woodin, R. Smolowitz, M. Okihiro, and D. Shea. 1996. Biomarkers of hydrocarbon exposure and sublethal effects in embiotocid fishes from a natural petroleum seep in the Santa Barbara Channel. Aquatic Toxicology 34: 195–219.

Springman, K. R., G. Kurath, J. J. Anderson, and J. M. Emlen. 2005. Contaminants as viral cofactors: assessing indirect population effects. Aquatic Toxicology 71: 13-23.

Springman, K. R., J. W. Short, M. R. Lindeberg, J. M. Maselko, C. Khan, P. V. Hodson, and S. D. Rice. 2008. Semipermeable membrane devices link site-specific contaminants to effects: Part I – induction of CYP1A in rainbow trout from contaminants in Prince William Sound, Alaska. Marine Environmental Research 66: 477-486.

Stegeman, J. J., P. J. Kloepper-Sams, and J. W. Farrington. 1986. Monooxygenase induction and chlorobiphenyls in the deep-sea fish *Coryphaenoides armatus*. Science 231: 1287-1289.

Trust, K. A., A. Fairbrother, and M. J. Hooper. 1994. Effects of 7,12-dimethylbenz[a]anthracene on immune function and mixed-function oxygenase activity in the European starling. Environmental Toxicology and Chemistry 13: 821-830.

Trust, K. A., D. Esler, B. R. Woodin, and J. J. Stegeman. 2000. Cytochrome P450 1A induction in sea ducks inhabiting nearshore areas of Prince William Sound, Alaska. Marine Pollution Bulletin 40: 397-403.

Vandermeulen, J. H. and J. G. Singh. 1994. ARROW oil spill, 1970-1990: Persistence of 20-yr weathered Bunker C fuel oil. Canadian Journal of Fisheries and Aquatic Sciences 51: 845-855.

Whitlock, J. P. Jr. 1999. Induction of cytochrome P4501A1. Annual Review of Pharmacology and Toxicology 39: 103-125.

Whyte, J. J., R. E. Jung, C. J. Schmitt, and D. E. Tillitt. 2000. Ethoxyresorufin-*O*-deethylase (EROD) activity in fish as a biomarker of chemical exposure. Critical Reviews in Toxicology 30: 347-570.

Wiedmer, M., M. J. Fink, J. J. Stegeman, R. Smolowitz, G. D. Marty, and D. E. Hinton. 1996. Cytochrome P-450 induction and histopathology in pre-emergent pink salmon from oiled spawning sites in Prince William Sound. Pages 509–517 *in* S. D. Rice, R. B. Spies, D. A. Wolfe, and B. A. Wright, editors. Proceedings of the *Exxon Valdez* Oil Spill Symposium, Bethesda, Maryland. American Fisheries Society Symposium 18.

Wolfe, D. A., M. J. Hameedi, J. A. Galt, G. Watabayashi, J. Short, C. O'Claire, S. Rice, J. Michel, J. R. Payne, J. Braddock, S. Hanna, and D. Sale. 1994. The fate of the oil spilled from the *Exxon Valdez*. Environmental Science and Technology 28: 561-568.

Woodin, B. R., R. M. Smolowitz, and J. J. Stegeman. 1997. Induction of Cytochrome P4501A in the intertidal fish (*Anoplarchus purpurescens*) by Prudhoe Bay crude oil and environmental induction in fish from Prince William Sound. Environmental Science and Technology 31: 1198-1205.

TABLES AND FIGURES

Table 1. Sample sizes of harlequin ducks captured in Prince William Sound, Alaska for analyses of cytochrome P4501A induction in March 2013. Numbers are listed by sex and age class cohort, and capture area (oiled during *Exxon Valdez* oil spill versus unoiled).

Cohort[a]	Oiled	Unoiled
AHY M	18	15
HY M	0	2
AHY F	7	7
HY F	0	1
TOTAL	25	25

[a]Cohort consists of an age class designation (HY = hatch-year, i.e., within one year of hatching; AHY = after-hatch-year) and sex (M = male; F = female).

Table 2. Results of information-theoretic analyses using general linear models to evaluate variation in hepatic7-ethoxyresorufin-*O*-deethylase (EROD) activity of harlequin ducks (n = 50) captured in Prince William Sound, Alaska during March 2013.

Model	K[a]	AIC_c[b]	ΔAIC_c[c]	w[d]
EROD = Area[e]	3	319.9	0.0	0.43
EROD = null	2	319.9	0.0	0.43
EROD = Area + Individual[f]	6	321.9	3.4	0.08
EROD = Individual	5	324.1	4.2	0.05

[a]K = number of estimated parameters in the model.
[b]AIC_c = Akaike's Information Criterion, corrected for small sample size.
[c]ΔAIC_c = difference in AIC_c from the best supported model.
[d]w = AIC_c weight.
[e]Area = categorical variable indicating areas either oiled during the *Exxon Valdez* spill or unoiled.
[f]Individual = a grouping of variables describing attributes of individuals (age, sex, and mass).

Table 3. Parameter likelihoods (P.L.), weighted parameter estimates, and unconditional standard errors (SE) derived from information-theoretic analyses using general linear models to evaluate variation in hepatic7-ethoxyresorufin-*O*-deethylase (EROD) activity (pmol/min/mg protein) of harlequin ducks captured in Prince William Sound, Alaska during March 2011.

	P.L.	Estimate ± SE
Intercept	1.00	25.99 ± 13.25
Area[a]	0.49	-4.76 ± 6.50
Sex[b]	0.13	1.57 ± 3.42
Age[c]	0.13	-1.12 ± 3.31
Mass (g)	0.13	-0.00 ± 0.02

[a]Area = categorical variable indicating areas either oiled during the *Exxon Valdez* spill or unoiled, with unoiled as the reference value.
[b]Sex = categorical variable (male versus female), with male as the reference value.
[c]Age = categorical variable (hatch-year versus after-hatch-year), with hatch-year as the reference value.

Figure 1. Average (± SE) hepatic7-ethoxyresorufin-O-deethylase (EROD) activity (pmol/min/mg protein) of harlequin ducks (n = 50) captured in Prince William Sound, Alaska in March 2013, contrasted with results from previous years (Esler et al. 2010, Esler 2011).

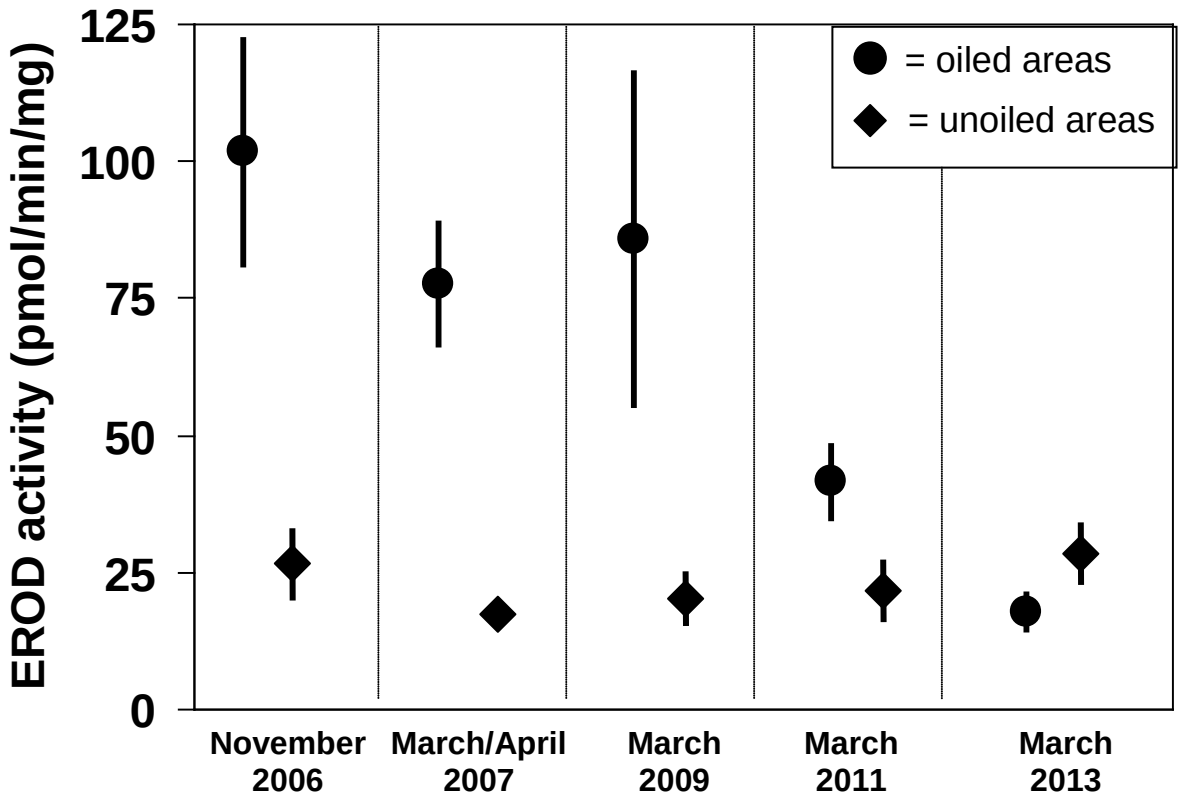

Figure 3. Average (± 95% CI) scaled hepatic7-ethoxyresorufin-O-deethylase (EROD) activity of harlequin ducks (n = 50) captured in March 2013 in areas of Prince William Sound, Alaska oiled during the Exxon Valdez spill relative to nearby unoiled areas, contrasted with results from previous years (Esler et al. 2010, Esler 2011). Results are scaled such that the average on unoiled areas for each year is set to 1; therefore, the data point for each year represents the multiplicative degree to which EROD is elevated on oiled areas (e.g., in 2011, EROD activity was approximately 2 times higher on oiled areas than on unoiled areas).

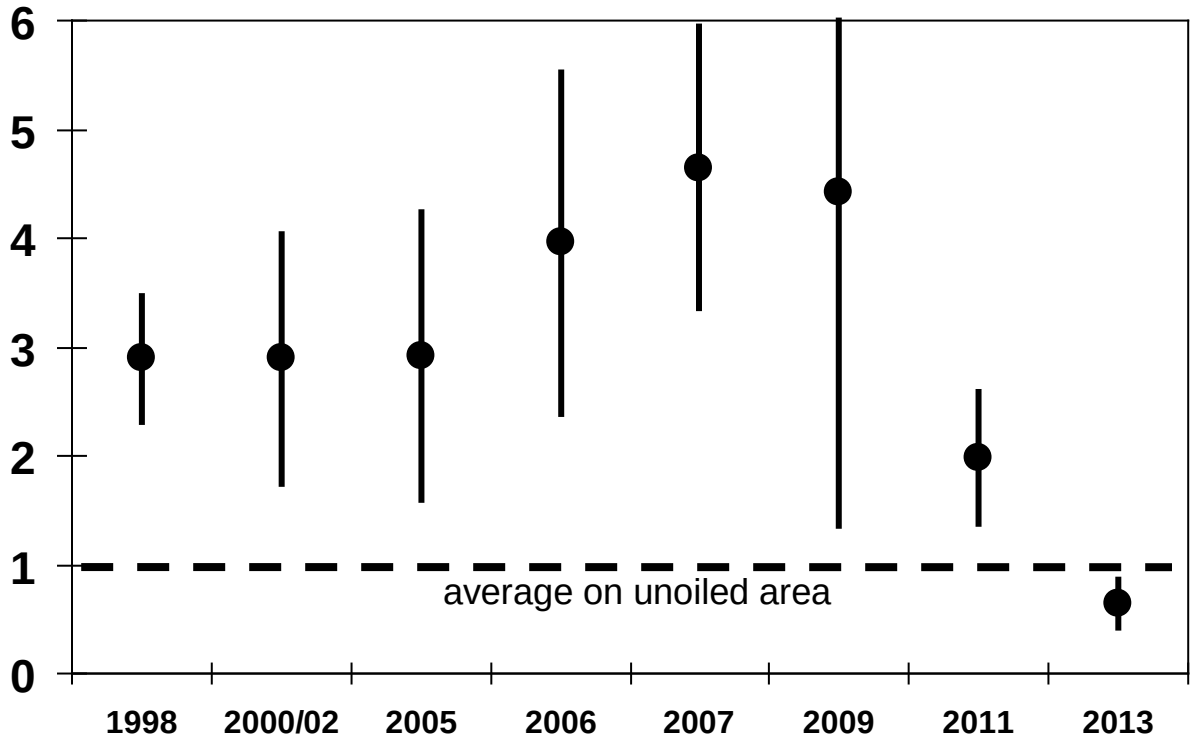

Figure 4. Proportion (y-axis) of captured harlequin ducks with elevated hepatic7-ethoxyresorufin-O-deethylase (EROD) activity, defined as 2 times the average among birds from unoiled areas. Data include results from this study (March 2013) contrasted against findings from previous studies (Esler et al. 2010, Esler 2011).