ESTIMATION AND IMPACTS OF SEABIRD MORTALITY FROM CHRONIC MARINE OIL POLLUTION OFF THE COAST OF NEWFOUNDLAND

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by

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ABSTRACT

The Grand Banks south off Newfoundland provide year-round feeding habitat for tens of millions of seabirds of numerous species, an abundance and diversity unparalleled in the North Atlantic. Dense ship traffic routes traverse this productive environment as vessels travel the Great Circle Route between Europe and North America. Oiled seabirds have washed up on beaches in Newfoundland for many decades. Most oil on their feathers has been identified as heavy fuel oil mixed with lubricants, the mixture found in the bilges of large vessels. Beached bird surveys conducted between 1984-1999 indicate that the incidence of chronic oil pollution along the southeast coast of Newfoundland is among the highest in world. More than 60% of all dead birds found over the 16-year period had oil on their feathers; 74% during the last five years. Auks, especially Thick-billed Murres (*Uria lomvia*), are the most affected.

In an effort to estimate overall mortality of seabirds in winter due to chronic oil pollution in Atlantic Canada, I performed a series of experiments to determine the fate of oiled and unoiled birds at sea and on beaches. First, I determined that carcasses persisted on average for only 3.3 ± 0.1 days on beaches in southeastern Newfoundland, after which they were no longer detectable due to scavenging or burial in the beach substrate. In addition, no differences were found in persistence rates between oiled and unoiled birds. I also determined deposition rates and detection probabilities of bird carcasses on beaches, and developed a model to estimate the number of birds arriving on a beach between periodic surveys. This model only performs well if survey intervals are less than 10 days.

Second, I designed a drift block that accurately mimics the movements of a seabird carcass drifting at sea. As drift blocks used in past studies showed little resemblance to actual carcass drift because they were overly influenced by wind, a more realistic drift block was needed to accurately interpret the number of birds that are found dead on beaches. Third, I measured murre carcass sinking rates and found that birds only float 8.2 ± 5.2 days before sinking, but that scavenging is important. Fourth, I carried out extensive drift block experiments using the new block design to determine the proportion of birds that die at sea and reach the shore, taking into account sinking rates of floating carcasses at sea. Recovery rates of blocks dropped at different locations varied, and the best predictor for the proportion of blocks lost at sea was the distance from shore where they were dropped, combined with the cumulative wind direction vector during the first three days following drift block drops. Based on wind patterns observed during the experiment, I was able to estimate wind specific recovery rates and catchment areas for birds that die at sea. Fifth, I constructed a general mathematical Oiled Seabird Mortality Model to assess seabird mortality due to chronic oil pollution along a given coastline.

I applied the Oiled Seabird Mortality Model to southeastern Newfoundland, based on periodic beached bird surveys conducted during the winters 1998/1999 through 2000/2001 and the parameters I determined earlier. Several assumptions were made to extrapolate seabird mortality due to oil to a large area at sea, and my most robust estimate is that on average, 315,000 ± 65,000 seabirds were killed annually in southeastern Newfoundland due to illegal discharges of oil from ships. Thick-billed Murres that overwinter on the Grand Banks made up 67 % of this kill. I examined the effects of this anthropogenic mortality, in combination with the estimated number of murres killed during the traditional murre hunt in Newfoundland, on Thick-billed Murre populations that breed in the eastern Canadian Arctic, by building a stochastic (demographic and environmental), age-structured, density independent, pre-breeding, Lefkovitch population projection matrix. The model suggested that chronic oil pollution has reduced potential annual population growth by 2.5 %. In combination with a further 2 % reduction in annual growth caused by hunting, these sustained anthropogenic causes of mortality have made Thick-billed Murre populations particularly vulnerable to environmental changes (e.g. global warming, ocean regime shifts). A series of actions are outlined to help reduce chronic oil pollution in Atlantic Canada, including increased year-round enforcement, imposition of minimum fines and higher imposed fines, the establishment of convenient oil disposal facilities on land, and increased education and awareness programs.

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CHAPTER 1 – INTRODUCTION

Much of the recent public concern with marine oil pollution arose from the effects of large catastrophic spills such as the *Torrey Canyon, Exxon Valdez, Braer* and *Sea Empress*. Consequently increased levels of public pressure have led governments world-wide to develop effective national and international legislative protection for the marine and coastal environments. Many countries, including Canada, have signed agreements, such as the International Convention for the Prevention of Pollution from ships (MARPOL), United Nations Convention on the Law of the Sea (UNCLOS) and regional protection agreements such as the Oslo-Paris and Helsinki Conventions, in which dumping of waste materials at sea is strictly controlled or expressively forbidden. According to MARPOL, the highest legally permitted concentration of oil that can be released into the ocean is 15 ppm. This low concentration can be achieved by running oily bilge water through an oil-water separator before disposal at sea, thus avoiding the formation of oil sheen on the surface of the water.

Seabirds are the most visible of marine organisms. They are defined as those birds whose normal habitat and food source is the sea, and include albatrosses, auks, boobies, gannets, cormorants, frigatebirds, gulls, loons/divers, pelicans, penguins, petrels, phalaropes, skuas, jaegers, storm-petrels, terns, skimmers, tropicbirds, and in some instances sea-ducks. Generally, seabirds are semi-colonial or colonial, and only come to land to breed. They are characterized by delayed maturity, low fecundity and high survival, and many undergo large fall and spring migrations. Seabirds feed on fish and invertebrates by a variety of methods, including skim feeding, plunge-diving, pursuit-diving and klepto-parasitism. The most abundant seabirds in the Northern hemisphere are Dovekies (*Alle alle*) and Murres (*Uria* spp.). Seabirds are also one of the most obvious victims of marine oil pollution.

Despite the obvious interest and attention directed towards large well-publicized oil spills, the absence of a consistent relationship between spill volume and the extent of seabird mortality pointed to the necessity to thoroughly investigate smaller spills (Burger 1993a). Concerns have been raised that many small chronic spills may kill as many seabirds as the more widely publicized large spills do, and as a result be as, or even more detrimental to long-term seabird population stability (Hunt 1987, Camphuysen 1989, Burger and Fry 1993, Oka et al. 1999).

Oiled birds keep washing up on shorelines worldwide (chapter 2, Camphuysen and Heubeck 2001), suggesting that national legislation, as well as international conventions and guidelines, are not being followed. As a result, several areas in the North Sea and in Atlantic Canada, where large seabird concentrations overlap with intense ship traffic, have been identified as extremely high oiling risk zones for seabirds (Chardine 1990, Lock et al. 1994, Skov et. al 1995, Fig.1.1).

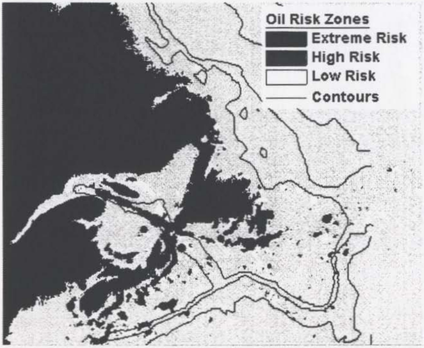

Fig.1.1. Annual oiling risk zones for seabirds in Atlantic Canada. Modeling based on known seabird distributions and ship traffic densities (Environment Canada 1998).

1.1 CHRONIC OIL POLLUTION IN ATLANTIC CANADA

The Grand Banks on the east coast of Canada are the most important wintering ground for seabirds in the North Atlantic. More than 40 million pelagic seabirds are estimated to reside or migrate annually through the waters surrounding the island of Newfoundland (Chardine 1995, Montevecchi 2000), which are is generally considered one the most productive marine areas in the world.

The degree of seabird mortality from oil pollution is largely determined by the timing and location of the discharge in relation to the distribution of birds (Burger 1993). Seabirds are the most conspicuous organisms of the marine ecosystem and most frequently used as an indicator of marine oil pollution (Furness and Camphuysen 1997, Montevecchi 2001). Although spilled oil undoubtedly affects the entire marine ecosystem (Kennish 1997), it is most visible in the presence of oiled birds on beaches. Tuck (1961) reported oiled birds coming ashore in Newfoundland since the 1950s and called for the halt of this pollution more than four decades ago. Yet one of the main problems to bring about changes in the interpretation and implementation of regional, national and international policies, conventions and laws, is the lack of solid scientific evidence that shows the degree of the problem in this region, and the absence of a reliable estimate of total seabird mortality caused by chronic oil pollution.

Various estimates of total seabird mortality caused by chronic oil pollution have been suggested for the North Atlantic, and specifically for Atlantic Canada and Newfoundland.

Nelson-Smith (1972) proposed that between 150,000 and 450,000 seabirds die annually in the North Atlantic due to oil pollution, but relied on approximations and guesses for most of the important parameters, such as carcass persistence on beaches and proportion of birds lost at sea. Tuck (1961) estimated that up to 500,000 birds died annually off the coast of Newfoundland alone, but arrived at his estimates in much the same way as the previous study. A second estimate of 20,000-100,000 seabird deaths due to oil pollution in Newfoundland was produced by Piatt et al. (1985). Their calculations involved extrapolating the density of beached carcasses/km, determined from monthly beached bird surveys, to the entire suitable coastline around the island of Newfoundland. They also assumed that 10-30 % of oiled birds made it to shore and were counted on the monthly surveys. As the authors point out, several large uncertainties exist with this estimate, due to a lack of correction for detection probabilities, no estimates of beached carcass persistence, the assumption that mortality is even around the island of Newfoundland, and the uncertainty in the proportion of birds lost at sea. In addition, other important aspects were not considered that could lead to a biased estimate: 1) the accuracy of drift blocks used to mimic seabird carcass drift, 2) the necessity to correct drift block recoveries for carcass sinking rates, 3) the accuracy of determining the number of birds deposited between surveys from monthly surveys, and 4) the lack of a relationship between seabird mortality at sea and the amount of beach deposition area available.

As a result of the remaining uncertainties in all these estimates, much discussion and confusion has persisted on the extent of seabird mortality due to chronic oil pollution in

Newfoundland and Atlantic Canada. It remained critical to accomplish changes in the interpretation and implementation of regional, national and international policies, conventions and laws, and to help reduce this problem on a local and global scale, especially given the region's geographic importance for seabirds. The aim of this thesis is to develop a model which allows an accurate determination of mortality caused by chronic oil pollution based on information collected from beached bird surveys, and based on this model, determine estimates for all necessary parameters and produce a more reliable estimate of the number of seabirds killed annually in Newfoundland waters. Finally, I wanted to investigate the potential impacts on populations of the species most affected, and to propose mechanisms through which chronic oil pollution in Atlantic Canada could be minimized.

1.2. CHAPTER OUTLINES

In chapter 2, I examine the extent of the problem of chronic oil pollution in southern Newfoundland, as assessed through beached bird surveys. The most common index for chronic oil pollution is the proportion of birds found which are oiled (Camphuysen 1989). This index does not provide information about the number of birds affected, but reflects the risk to birds of dying from spilled oil. Based on the assumption that physical processes (weather, currents, beach types) that influence whether oiled or unoiled birds wash up on the shore are equal over time, this index allows direct comparison of the extent of chronic oil pollution between geographical areas and among species. The

Canadian Wildlife Service of Environment Canada in St. John's, Newfoundland has conducted monthly beached bird surveys in southeastern Newfoundland since January 1984. Their data form the basis of this chapter, and this analysis sets the stage to estimate overall mortality due to this anthropogenic mortality.

In chapters 3 through 6, I empirically determine the parameters needed to accurately estimate seabird mortality due to spilled oil off the coast of Newfoundland. To use beached bird carcasses to determine the total number of birds killed, requires the knowledge of, 1) the proportion of birds that die at sea which reach the shore, 2) the ratio of birds which reach the beach alive or dead, 3) the fate of birds cast on the beach, and 4) the probability of detecting beached birds during surveys (Ford et al. 1987, Piatt et al. 1990, Burger and Fry 1993, Fowler and Flint 1997).

In chapter 3, I determine the rate at which birds are deposited on local beaches, and estimate carcass persistence, the amount of time a carcass remains detectable on the beach. Unless beach surveys are conducted daily, accurate determination of detection probabilities, deposition rates and beached carcass persistence are extremely critical (Armstrong et al. 1978, Stowe and Underwood 1984, Camphuysen 1989, Piatt et al. 1990, Bodkin and Jameson 1991, Burger 1992), and greatly influence the final estimate from any total seabird mortality model (Page and Carter 1986, Burger 1992, Burger and Fry 1993). The disappearance of birds from a beach is closely related to environmental conditions such wind, sea state, tidal levels, temperature, and precipitation. It is also related to scavenger presence, density and satiation, and to beach characteristics such as substrate, slope, and orientation (Camphuysen 1989, Bodkin and Jameson 1991, Burger 1992, Rodrigues 1995, Van Pelt and Piatt 1995, Fowler and Flint 1997). Due to the heterogeneity of beaches and conditions throughout the year, it is difficult to compare data from different locations (Burger 1992), and hence these factors have to be assessed locally.

Even when the total number of beached carcasses is known or can be well estimated based on deposition, detection and persistence rates, a large fraction of the total mortality remains unmeasured, as most birds are lost at sea and never reach the beaches (Ford et al. 1987). Several attempts have been made to assess the proportion of birds which die at sea and reach shore using seabird carcasses (Coulson et al. 1968, Hope Jones et al. 1970, 1978, Lloyd et al. 1974, Bibby and Lloyd 1977, Bibby 1981, Stowe 1982, Threlfall and Piatt 1982, Page et al. 1982, Piatt et al. 1990) and wooden drift blocks (Threlfall and Piatt 1982, Piatt et al. 1985, Hlady and Burger 1993, Chardine and Pelly 1994, Flint and Fowler 1998). The main conclusion from these studies is that onshore recoveries are heavily influenced by environmental conditions (ocean currents, wind speed and direction) and distance from shore, and that they differ among geographical regions. Consequently, reported onshore recoveries can not easily be compared, and the 10 % rule used in the past (Tanis and Morzer Bruijns 1968, Bourne 1970, National Research Council 1985, Canadian Coast Guard 1998) is generally inaccurate (Burger 1993). In addition, methodological deficiencies, such as not incorporating seabird sinking rates, and whether drift blocks previously used mimic seabird carcass drift, make the conclusions of the majority of these studies questionable.

To address these deficiencies, I examine the accuracy of drift blocks used in the past to mimic seabird carcass drift in chapter 4 (published as Wiese and Jones 2001), and the rate at which carcasses floating at sea lose their buoyancy and sink in chapter 5. In chapter 6, I determine the proportion of birds that die at sea and reach the shore by building on data gathered in chapters 4 and 5. Chapter 6 presents wind-specific recovery rates for the first time, and determines the area of ocean from which beached carcasses originate based on prevailing wind direction. It lays the ground work to estimate seabird mortality from chronic spills, a different process from estimating mortality after large catastrophic spills (Ford et al. 1987).

In chapter 7, I construct a general mathematical oiled seabird mortality model and describe parameters and equations needed to estimate seabird mortality from chronic spills, based on data from beached bird surveys. I then apply this model to southeastern Newfoundland, using the necessary parameter estimates determined in chapters 3-6. To my knowledge, this is the first report of an empirically-based seabird mortality estimate due to chronic oil pollution for a given area.

In chapter 8, I use this oiled seabird mortality estimate information to examine the impacts on Thick-billed Murre *(Uria lomvia)* populations. Although Thick-billed Murre breeding populations in the eastern Canadian Arctic are stable or increasing (Gaston 2002), it is important that the impacts of anthropogenic causes of mortality on Thick-billed Murre population be examined. Murres are the species most affected by oil spills in the Northern Hemisphere including Newfoundland (Piatt et al. 1991, Camphuysen and

Heubeck 2001). Much concern has been voiced over the potential impacts of chronic oil pollution on seabird populations (Bourne 1968, Croxall 1975, Camphuysen 1989, Dunnet 1982, Albers 1995, Newton 1998), yet to date no information existed that was accurate enough to estimate potential population impacts.

Finally, I present a list of research needs to help refine parameter estimates for the model, and to increase our understanding on the behaviour of oiled birds at sea. I conclude by outlining a series of mitigative measures, that, if implemented, could effectively address the illegal dumping of oily discharges from ships into our marine ecosystem,

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1.4. CO-AUTHORSHIP STATEMENT

Several individuals have contributed to one or more chapters of this thesis. Chapter 2 was based on data collected by Pierre Ryan from the Canadian Wildlife Service. I was responsible for formulating all research questions for chapters 2-8, and for data collection for chapters 2, 3, 4, 5, and 6. Beached bird data in chapter 7 was again provided by the Canadian Wildlife Service that was collected under contract by Tony Power (1998/99) and Stan Tobin (1999/2000 and 2000/01). Important unpublished demographic data used in chapter 8 was obtained from Dr. Tony Gaston from the Canadian Wildlife Service. I was also responsible for determining appropriate statistical analysis techniques, although most chapters profited greatly from the assistance of my co-supervisor Dr. G. J. Robertson, specifically the models developed in chapters 3, 7 and 8. I was solely responsible for writing the manuscripts. Prior to submission to the School of Graduate Studies, comments were obtained for chapter 2 from my first supervisor Dr. Ian Jones, from Dr. Tony Gaston for chapter 8, and for all chapters from my co-supervisors (Dr. Greg Robertson and Dr. Bill Montevecchi) and the third member of my supervisory committee (Dr. Richard Elliot).

CHAPTER 2 – THE EXTENT OF CHRONIC MARINE OIL POLLUTION IN SOUTHEASTERN NEWFOUNDLAND WATERS ASSESSED THROUGH BEACHED-BIRD SURVEYS 1984-1999

2.1 INTRODUCTION

Oil enters the marine environment from land runoff, natural seeps, vessels, pipelines and offshore exploration and production platforms (Clark 1992, Schmidt Etkin 1999, Wiese et al. 2001). Not including spills under 0.17 tonnes, vessels have contributed 64% of all accidental spills between 1978 and 1997 (the rest stems from onshore or offshore facilities), either through routine operations or large catastrophic spills, spilling an estimated total of 701,040 tonnes of oil into the marine environment (Schmidt Etkin 1999). Much of the public concern about oil affecting the marine environment arose following large catastrophic spills such as the *Torrey Canyon*, *Exxon Valdez*, *Braer* and the *Sea Empress* where 30,000-370,000 seabirds were killed. Total spillage from such incidents in terms of metric tons of oil, however, is less than that released as illegal discharges due to tank washings, dirty ballast and bilge pumping (Brander-Smith et al. 1990).

Seabirds are the most conspicuous marine organisms and have been used as monitors of the marine environment (Montevecchi 1993, 2001) and of the incidence of oil pollution for decades (Furness and Camphuysen 1997). Oil at sea is a threat to seabirds because it forms a thin layer on the ocean surface where many birds spend their time. The hydrophobic nature of oil causes plumage to readily absorb the oil, which decreases the birds' insulation, waterproofing and buoyancy, leading to death due to hypothermia or starvation. The many toxic compounds in the oil, when ingested or inhaled, can also lead to debilitating or fatal effects due to their impact on internal organs (Fry and Lowenstine 1985, Leighton 1993, Briggs et al. 1997). The amount of oil that is lethal to birds is very small (Levy 1980, Fry and Lowenstine 1985, Khan and Ryan 1991, Leighton 1995).

The extent of chronic oil pollution along a given shoreline is usually assessed by conducting systematic beached bird surveys, and determining the number and percentage of birds found that are oiled. Oiling rates (percentage of birds found that are oiled) are considered to indicate the risk to birds of becoming oiled at sea, and thus to mirror fluctuations in the amount of oil spilled over time and in different regions (Furness and Camphuysen 1997, Camphuysen and Heubeck 2001). Chemical analyses of oil samples taken from beaches and from stranded oiled birds in Newfoundland and Germany have shown that 90% are heavy fuel oils mixed with lubricant oils, the type of mixture found in bilges of large ocean-going vessels (A.R. Lock pers. comm., Averbeck et al. 1992). Based on the frequent overlap of shipping activities and seabird distribution, it is not surprising that beached bird surveys around the world have reported dead and live oiled seabirds resulting from chronic spills for decades (Canada East coast: Tuck 1961, Levy 1980, Piatt et al. 1985, Montevecchi and Tuck 1987, Elliot and Ryan 1988, Lock 1992, Chardine 1992, 1995, Chardine and Pelly 1994, Wiese and Ryan 1999; Canada West coast: Burger 1993a, Burger and Fry 1993; *USA* West coast: Speich and Wahl 1986, Bodkin and Jameson 1991, Nur et al. 1997; USA East coast: Simons 1985; Argentina:

Perkins 1983, Boersma 1995; Japan: Ohata et al. 1993; New Zealand: Veitch 1982; Australia: Raaymakers 1995; South Africa: Avery 1989, North Sea: Dahlmann et al. 1994; UK: Cadbury 1978, Stowe and Underwood 1984, Heubeck 1995; Denmark: Joensen and Hansen 1977; Belgium: Kuyken, 1978; Netherlands: Camphuysen 1989, 1998; Germany: Averbeck et al. 1992; France: Debout 1984, Raevel 1990; Lithuania: Vaitakus et al. 1994, Zydelis and Dagys 1997).

It has been suggested that these chronic effects may be as important or more important to long-term seabird population stability than occasional large spills (Hunt 1987, Burger 1992). This conclusion is supported by the absence of a strong correlation between the volume of oil spilled and consequent seabird mortality. It is rather the timing and location of the spill that determines the degree of seabird mortality (Burger 1993b).

The Grand Banks off the east coast of Canada are the most important wintering ground for seabirds in the North Atlantic. Over 40 million pelagic seabirds are estimated to reside or migrate annually through the waters surrounding the island of Newfoundland (Chardine 1995, Montevecchi 2000), and they are generally considered one the most productive marine areas in the world. Illegal discharges of oil have occurred in this region for decades (Tuck 1961, Montevecchi and Tuck 1987). Although spilled oil undoubtedly affects the entire marine ecosystem (Kennish 1997), it is most visible in the ongoing presence of oiled birds on beaches.

This paper presents the results of beached bird surveys conducted in Newfoundland during October to April of 1984-2000. Species-specific mortalities and vulnerabilities, trends and patterns in oiling rates, and the impacts of weather, ship traffic, and the local murre hunt on these trends are discussed. Finally, the value of beached bird surveys and oiling rates as indicators of chronic oil pollution in the region, as well as its wider implications and applications are assessed.

2.2 METHODS

2.2.1 Study area

The island of Newfoundland is located at the easternmost extremity of North America and lies near major shipping routes with Europe, many of which converge off Cape Race, the southeastern tip of the island (Fig. 2.1). Because most oil found on birds comes from large trans-Atlantic vessels travelling through this area (A. R. Lock pers. comm.), and because their routes overlap with large concentrations of seabirds vulnerable to oil pollution (Chardine 1990), the south coast of Newfoundland and other regions in Atlantic Canada, similar to areas in the North Sea (Skov et al. 1995), were identified as an extremely high oiling risk zone for seabirds by Environment Canada in 1994 (Fig. 2.2, Lock et al. 1994).

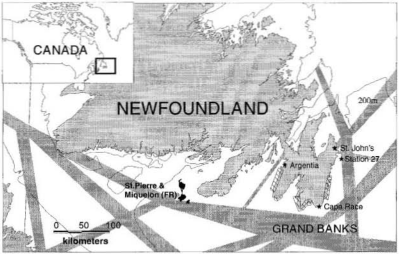


Fig. 2.1 Location of beached bird surveys in southeastern Newfoundland. Beached bird survey sections are shown as hatched areas, weather stations and place names are shown as stars and major shipping routes are greyed bars. FR= France.

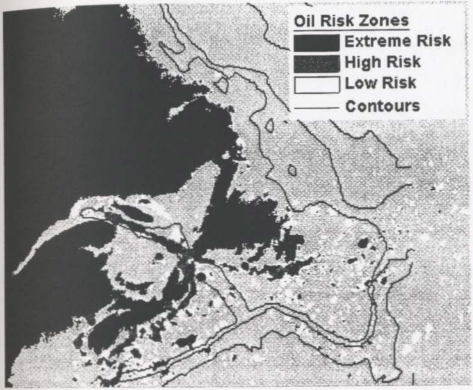


Fig. 2.2 Seabird Oiling Risk Zones in Atlantic Canada. Adapted from Environment Canada (1998).

2.2.2 Oiling rates

Beached bird surveys in Newfoundland have been coordinated and systematically carried out since 1984 by the Canadian Wildlife Service of Environment Canada in St. John's, Newfoundland. Monthly surveys are conducted during the winter months (October through April) by one to three observers covering different beaches, but restricted to a small section of the southeast coast of the island (Fig. 1). Beached bird surveys were conducted on foot, walking the upper and lower high tide line, recording and describing all corpses and parts of corpses encountered. Details on the age, sex, species, rate of decomposition, presence and amount of oil, survey effort and environmental and beach conditions at the time of the survey were recorded whenever possible, using international beached bird surveys protocol (Camphuysen and Heubeck 2001). Counted corpses were either collected or marked to avoid recounting later.

Species-specific and overall proportions of birds found that were oiled (oiling rate) and trends in these proportions over time were calculated, using only corpses complete enough to assure the presence or absence of oil (i.e. >75 % of body present), and using information only from those 13 beaches in southeastern Newfoundland that were consistently surveyed each month during the winters of 1984/85 through 1999/2000. In addition, all records related to known oil spill incidents or wrecks were eliminated from the database before the trend analysis was performed (Camphuysen and Francker 1992). Winters were treated as six months periods designated by the year in which they began, e.g. 1984 refers to the period of mid-October 1984 until mid-April 1985 inclusive. Since beached bird surveys were conducted during the last or first days of the month, months were coded to start and end on the 15th day in order to avoid bias to one specific month. For example, for the purpose of this analysis, the month of October refers to the period of 15 October - 15 November.

2.2.3 Analysis

Trend analyses were performed using logistic regression models (SAS 1990), where the proportion of birds that are oiled *p(oiled)* was analyzed with a General Linear Model where the response was treated as binary and using a logit link function. Firstly, a general model was built to investigate annual trends. Subsequently the influence of explanatory variables on *p(oiled)* were constructed, and finally a full model where:

$$p(oiled) = \beta_o + \beta_1 weather + \beta_2\, hunt + \beta_3\, ships$$

was assessed, where weather is a composite index based on wind speed and direction and air temperature, hunt is the number of murres killed in the winter murre hunt in Newfoundland, and ships is the number of container, cargo and tanker vessels in coastal Newfoundland waters (see relevant sections below). Where trends in oiling rates over time were found to be significant ($P<0.05$), analyses were performed using ANOVA/ANCOVA designs to determine whether changes in the number of clean or oiled birds were driving the change in the proportion of oiled birds. Furthermore, the influence of weather, hunting and ship traffic on these two components (oil and unoiled), as well as changes of each covariate over time, were investigated using standard correlation analysis. Finally, where the model did not converge due to insufficient data, some covariates were dropped from the analysis and investigated separately if possible. Chi-square values are presented instead of the usual F for ANOVA/ANCOVA designs, as a General Linear Model framework was used for all analysis. Correlation coefficients (r), obtained from Pearson's correlations are also presented for ANOVA/ANCOVA type analysis. All means are presented as ±1 SE, and all tests are 2-tailed.

2.2.4 Weather

Environmental conditions such as wind speed and direction, surface current variability, ambient air temperature and sea surface temperature can influence the number of oiled and clean birds found during beached bird surveys. If oiled and clean birds do not respond equally, fluctuations in these environmental parameters will affect the proportion of oiled birds and its trends over time. Therefore, hourly weather information (wind speed and direction, ambient air temperature) from weather stations at Argentia and Cape Race (Fig. 2.1) were obtained from the Environment Canada Atlantic Climate Center in Fredericton, New Brunswick, Canada for the period between 1984 - 2000. Daily means, minima and maxima were calculated for each weather parameter for each month, and the following summary and ratio parameters were derived from these means in order to give more weight to colder and windier days (adapted from Camphuysen 1989).

1. Temperature code, where the higher the code the colder the temperature. Days with temperatures greater than 0° Celsius get one point, days with temperatures less or equal than 0° but greater or equal to -5° get two points,

days with temperatures less than -5° but greater or equal to –10° get three points, days with temperatures less than -10° get four points, and then all points are summed.

2. Onshore wind ratio, defined as the number of days with onshore days (dependent on individual beach orientations and defined as any wind parallel or at a direct angle towards the beach) relative to all days.

3. Extreme wind velocity code was calculated by assigning the number of days with gales ($28 \text{ kts} \leq \text{wind speed} < 47 \text{ kts}$) one point, and the number of days with storms (wind speed ≥ 47 kts) two points, and adding them over the month.

Temperature codes, onshore wind ratios and extreme wind velocity codes were chosen because they appeared to represent the data well, reflected most relevant weather patterns for seabirds, and accounted for missing data in a reasonable way. In order to avoid the use of correlated variables, a correlation and subsequent principal component analysis of these weather parameters was carried out. All these weather indices were significantly correlated and were thus combined into one 'weather' index. The first principal component described 63% of variation in the weather, where high values indicated cold ambient air temperatures (eigenvector loading=0.616), strong winds (eigenvector loading=0.638), and high proportion of onshore winds (eigenvector loading=0.463).

In addition, representative daily sea surface temperatures (SST) were obtained for Station 27 (Fig. 2.1) from http://oceanography.nwafc.nf.ca:81/testpoint/english/stn27/index.html. SST had a significant negative correlation with the ambient air temperature code (n=88, (χ_1^2=61.8, P<0.0001, r=0.71) and was therefore not included in the analysis, as more detailed ambient temperatures could be obtained.

2.2.5 Murre hunt

Thick-billed Murres (*Uria lomvia*) and Common Murres (*U. aalge*) are hunted in a traditional fall and winter murre hunt along the coasts of Newfoundland and Labrador (Montevecchi and Tuck 1987, Elliot 1991, Chardine et al. 1999). A standardized murre hunt survey was introduced in 1977-78 by the Canadian Wildlife Service to assess timing, extent and characteristics of the harvest by permit holders and to monitor annual trends (Wendt and Cooch 1984). The hunt was regulated with bag limits and seasons in 1993, that reduced the hunt to about half of former levels (Chardine et al. 1999). As hunting effort could influence the number of crippled or non-retrieved murres, which would be found as clean dead birds, and hence influence the oiling rate, it needs to be accounted for. As a result, all available hunt estimates between 1984 and 2000 were obtained from the Canadian Wildlife Service. Data were available for only seven of the 16 years (1985-1987, 1990, 1995-1997). For the oiling rate analysis, the data were split into the period before and after the hunting restriction in 1993, and the influence of the murre hunt on oiling rates was analyzed together and separately for each period.

2.2.6 Ship traffic

Oil found on seabirds in Atlantic Canada originates from the illegal pumping of waste oil and oil-water mixtures of bilges from large trans-Atlantic tankers and cargo/container vessels. In order to include ship traffic volume of these types of vessels in the oiling rate analysis, we obtained all available container, cargo and tanker traffic information for Atlantic Canada from the Department of Fisheries and Oceans Atlantic Mapping Project in Dartmouth, Nova Scotia. Data were available between 1990-2000. This database includes all vessels reported to the Eastern Canada Vessel Traffic Services Zone Regulations (ECAREG), i.e. those destined or originating from Canadian ports. It does not include ships travelling through the area under 'innocent passage', i.e. those destined to foreign ports while keeping outside the 12 nm territorial water limit. Ship traffic volume was determined along the south coast of Newfoundland out to and including the major shipping lanes (Fig. 2.1), and its relationship to oiling rates was investigated.

2.2.7 Oil Vulnerability Index

Winter Oil Vulnerability Indices (WOVI) were calculated for species commonly found on beached bird surveys in this area throughout the winter months (October-April). These indices are based on the criteria outlined by King and Sanger (1979) and adapted by Camphuysen (1998), and take into account the birds' distribution, behaviour and exposure to oil. In general, the most vulnerable birds to oil pollution are those that roost on the sea, forage by swimming, escape by diving, have small wintering areas and overlap with areas of ship traffic throughout most of the year. WOVI were compared to observed oiling rates using Pearson correlation coefficients.

2.3 RESULTS

2.3.1 Patterns of covariates

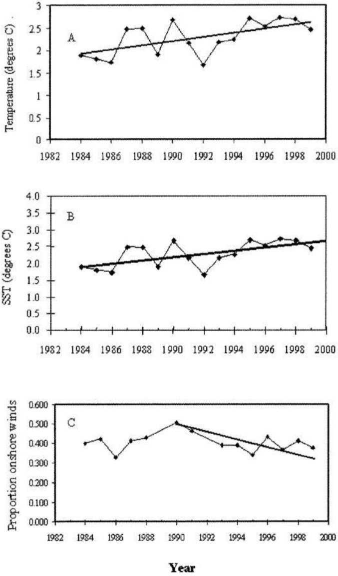
2.3.1.1 Weather

Analysis of weather parameters during winter showed a significant warming trend of ambient air temperature between 1984-2000, most strongly expressed by an increase in minimum ambient air temperature (χ_1^2 =10.35, P<0.005, Fig.2.3a). This trend was mirrored by increases of winter sea surface temperatures at Station 27 over the same time period (χ_1^2 =7.51, P<0.01,Fig. 2.3b), and a decrease in the proportion of onshore winds from 1990 to present (χ_1^2 =4.98, P<0.05, Fig. 2.3c).

2.3.1.2 Murre hunt

Due primarily to a bag limit and geographic and seasonal hunting restriction first implemented by the Canadian Wildlife Service in the winter of 1993-94 (Chardine et al. 1999), the number of murres hunted showed a significant overall decrease over time (n=42, χ_1^2 =9.3, P<0.005, r=0.45). No trends were apparent within the periods before and after the restriction, largely due to limited data.

Fig. 2.3. Changes in (a) minimum ambient air temperatures, (b) sea surface temperatures (SST), and (c) proportion of onshore winds, during winters (October-April) 1984/85-1999/2000 in southern Newfoundland. Linear correlation are shown, correlation of onshore winds with time was only performed from 1990 onwards (see results for details).


A
Temperature (degrees C)
0
0.5
1
1.5
2
2.5
3
1982
1984
1986
1988
1990
1992
1994
1996
1998
2000
B
SST (degrees C)
0.0
0.5
1.0
1.5
2.0
2.5
3.0
3.5
4.0
1982
1984
1986
1988
1990
1992
1994
1996
1998
2000
C
Proportion onshore winds
0.000
0.100
0.200
0.300
0.400
0.500
0.600
1982
1984
1986
1988
1990
1992
1994
1996
1998
2000
Year

2.3.1.3 Ship traffic

Container and tanker traffic information for Atlantic Canada was available between 1990-2000 from ECAREG, Ottawa. Ship traffic volume in winter along the south coast of Newfoundland increased significantly over time (χ_1^2=9.3, P<0.05, r=0.32). In addition, ship traffic records of individual vessels tracks coming in and out of Atlantic Canada showed that containers and tankers do not adhere strictly to defined shipping lanes and can be found throughout the area (Fig. 2.4 compared to Fig. 2.1).

2.3.2 Winter Oil Vulnerability Index

The species considered to be most vulnerable to oil pollution are those that roost on the sea, forage by swimming, escape by diving, have small wintering areas and overlap with areas of ship traffic throughout most of the year (Camphuysen 1998). Based on King and Sanger (1979) and Camphuysen (1998) WOVI were determined for the 11 species of birds most commonly found on Newfoundland beaches (Appendix 1). Murres (*Uria* spp.), Dovekies (*Alle alle*) and Common Eiders (*Somateria mollissima*) all scored high (above 30 out of a maximum score of 40), Herring and Great Black-backed Gulls (*Larus argentatus* and *L. marinus*) scored low (below 20), the others being intermediate (between 20 and 30). The predicted WOVI were significantly correlated with observed oiling rates (n=11, χ_1^2=9.9, P<0.005, Fig. 2.5 r=0.77).

2.3.3 Oiling rates

1676 birds were found on systematically surveyed beached during the winters 1984-1999 (Table 2.1), after those apparently resulting from known oil-spill incidents were removed

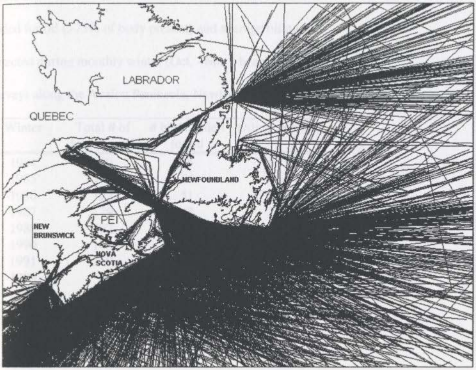


Figure 2.4 Typical annual ship track pattern of container, cargo and tanker vessels travelling to or through Atlantic Canada destined to Canadian ports in 1995. Patterns were similar in 1990-2000 (adapted from ECAREG).

Table 2.1. Corrected total number of birds, number of birds coded for oil (>75% of body present) and average oiling rate detected during monthly winter (Oct. - Mar.) beached bird surveys along the Avalon Peninsula, Newfoundland 1984-1999.

Winter	Total # of birds found	# birds coded for oil	Oiling rate (%)
1984	160	55	63.6
1985	106	54	42.6
1986	122	51	47.1
1987	179	68	70.6
1988	102	35	31.4
1990	106	68	48.4
1991	140	116	79.3
1993	73	58	41.4
1994	148	36	72.2
1995	33	20	85.0
1996	308	17	76.5
1997	49	35	80.0
1998	46	40	77.5
1999	104	55	61.8
TOTAL	**1676**	**708**	**62.0**

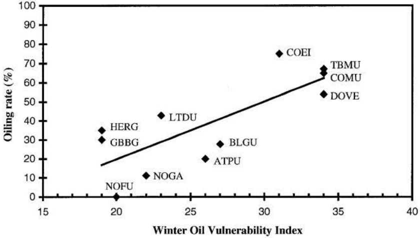


Fig. 2.5 Winter Oil Vulnerability Index (Appendix 1) in relation to observed oiling rates during winter (Oct-Mar) 1984-1999, r = 0.77). HERG=Herring Gull, GBBG=Great Black-backed Gull, NOGA=Northern Gannet, NOFU=Northern Fulmar, LTDU=Long-tailed Duck, ATPU=Atlantic Puffin, BLGU=Black Guillemot, COEI=Common Eider, TBMU=Thick-billed Murre, DOVE=Dovekie, COMU=Common Murre. WOVI adapted and derived from King and Sanger (1979) and Camphuysen (1998).

from the sample. In 708 (42.2 %) cases, more than 75 % of the carcass remained allowing an accurate coding for oil (Table 2.1). The overall oiling rate of those birds was 62.0 %, the mean number of oiled birds per km was 0.77, and the total number of corpses per km was 2.25. 75.6 % of all oiled birds had oil on less than 25 % of their bodies and most oiled birds were highly emaciated (absence of body and muscle fat, reduced pectoral muscles).

2.3.3.1. Annual trends

Oiling rates increased significantly at an average of 1.4 % per annum (χ_1^2=12.38, P<0.005, Fig. 2.6) while survey effort (number of beach surveys per month) showed no significant change over time (χ_1^2=1.74, P>0.1). The overall increasing trend in the proportion of oiled birds was only apparent in November and December, and was due to decreasing trends in the number of clean birds found per km in those months (χ_1^2=4.15, P<0.05). The number of clean birds were positively related to increasing ambient temperatures (χ_1^2=5.08, P<0.05) and for murres, which made up 65.1 % of all birds reported (Table 2), with increases in the murre hunt (χ_1^2=16.91, P<0.0001, Fig. 2.7). Overall the number of oiled birds per km was positively correlated to the weather index (χ_1^2=4.89, P<0.05), but no association with ship traffic volume was found (χ_1^2=0.18, P>0.5). A full model where the relative importance of all covariates could be examined was not possible due to insufficient overlap in time.

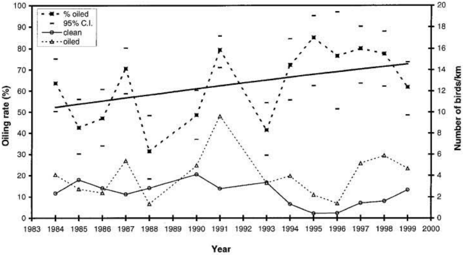


Fig. 2.6 Change in the proportion of oiled birds, and the density of clean and oiled birds per km found on southeastern Newfoundland beaches during winter (Oct – Apr) 1984-1999.

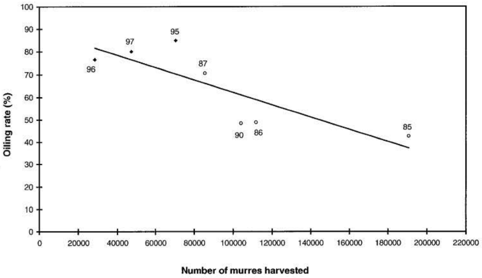


Fig. 2.7 Relationship between the number of murres harvested on the south coast of Newfoundland and the proportion of birds found oiled during winter (Oct-Apr), r = 0.83. Years are shown, while circles represent years before hunting restrictions, diamonds after restrictions.

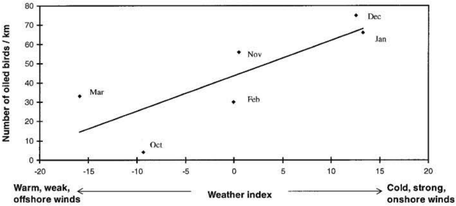


Fig. 2.8 Relationship between the total number of oiled birds/km found during winter (Oct-Apr) on beaches on southeast coast of Newfoundland, and the sum of the weather index for each winter month (1984-1999 pooled), r = 0.81.

2.3.3.2 Monthly trends

Monthly oiling rates differed from each other (χ^2_5=75.83, P<0.0001). The proportion of oiled birds peaked in December (Fig. 2.9), a pattern likely related to significant fluctuations in the monthly number of oiled birds per km (χ^2_1=18.15, P<0.005), as the monthly number of clean birds per km did not change significantly (χ^2_1=9.61, P>0.05). As above, monthly occurrence of oiled birds was found to be significantly influenced only by weather, with high incidence of oiled birds corresponding directly to periods of cold ambient air temperatures, high winds and periods of increased onshore winds (χ^2_5=18.15, P<0.0005, Fig. 2.8, r=0.81)

2.3.3.3 Species

The two species most affected in numbers and proportion oiled were murres (Common and Thick-billed Murres), for which a combined oiling rate of over 70 % was recorded (Table 2.2). Overall, the oiling rate for murres increased between 1984 and 1999 by an average of 0.9 % annually (χ^2_1=4.16, P<0.05), and showed less variability among years than all species combined. When the two species of murres were considered separately, this increase was only evident for Thick-billed Murres. Inclement weather was a significant predictor of the number of oiled murres found on beaches (χ^2_1=5.21, P<0.05). Murre harvest explained a significant amount of variation in the proportion of oiled murres (χ^2_1=15.02, P<0.0001) because the decrease in harvest over the years was

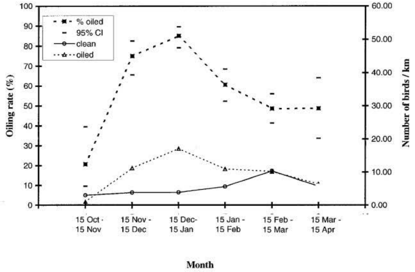


Fig. 2.9 Changes in the proportion of oiled birds, and the total number of clean and oiled birds found per km on beaches in southeastern Newfoundland between mid-October and mid-April, 1984-1999.

Table 2.2. Percent occurrence of the corrected total number of birds coded for oil, and oiling rates (proportion of birds found oiled) of species found on beached bird surveys 1984-1999.

Species	Number	Frequency (%)	Oiling rate (%)
Murres (*Uria* spp.)	461	65.1	70.5
Thick-billed Murres (*Uria lomvia*)	164	23.2	67.1
Common Murres (*Uria aalge*)	111	15.7	64.9
Unidentified Murres (*Uria* spp.)	186	26.3	76.9
Dovekies (*Alle alle*)	115	16.3	53.9
Herring Gulls (*Larus argentatus*)	20	2.8	35.0
Black Guillemot (*Cepphus grylle*)	18	2.5	27.8
Common Eider (*Somateria mollissima*)	16	2.3	75.0
Great Black-backed Gull (*Larus marinus*)	10	1.4	30.0
Atlantic Puffin (*Fratercula arctica*)	10	1.4	20.0
Northern Gannet (*Sula bassana*)	9	1.3	11.1
Northern Fulmar (*Fulmarus glacialis*)	8	1.1	0.0
Long-tailed Duck (*Clangula hyemalis*)	7	1.0	42.9

significantly correlated to a decrease in the number of clean murres found on beaches (χ_1^2=17.29, P<0.0001). When annual changes in the murre harvest were accounted for, the oiling rate for all murres combined, as well as for Thick-billed Murres alone, remained unchanged over time (χ_1^2=1.29, P>0.05).

2.4. DISCUSSION

2.4.1 Oiling rates

Birds found in more northern latitudes or colder oceans are expected to have higher oiling rates (proportions of birds found oiled), because oil on the water persists longer in cold temperatures, and because the thermal stress of birds is increased after oiling (Hartung 1967, Jenssen et al. 1985, Culik et al. 1991, Doerffer 1992). Yet, Newfoundland's overall oiling rate of 62% is among the highest documented in the world (Burger 1993a, Vaitakus et al. 1994, Heubeck 1995, Nur et al. 1997, Zydelis and Dagys 1997, Camphuysen 1998, Camphuysen and Heubeck 2001). As in studies elsewhere, auks and diving ducks are the avian groups most affected by oiling in Newfoundland, likely because they spend most of their time on, or under, water. Auks and sea ducks are known to dive as an escape response, a behaviour that makes them more vulnerable to oil at sea than species more likely to evade a slick by taking flight. The much higher number of murres compared to all other species certainly reflects their high abundance during the winter months, but the overall oiling rate for murres of 70.5 %, and of 80.4 % during the last 5 years surveyed, also ranks among the highest in the world (3-82 %, Camphuysen

1998). High oiling rates are of special concern for the conservation of murres and other auks. Owing to their low-fecundity and high-survival life histories, even slightly increased sustained adult mortality can have appreciable population impacts and could threaten long-term population stability (Nur et al. 1997).

As would be expected, coastal species such as gulls and Black Guillemots, and highly aerial species such fulmars and gannets, were less affected by oil, and thus Winter Oil Vulnerability Indices were a good predictor of observed oiling rates. Larger gulls and birds of prey may be contaminated when they prey on or scavenge, injured and oiled birds. However, gulls and auks have been observed to successfully preen themselves clean of oil and survive, although perhaps with lower reproductive success (Corkhill 1973, Esler et al. 2000). Oiling mortality is underestimated by excluding birds whose carcasses lacked external oil but that died due to internal contamination (Leighton 1995, Briggs et al 1997). Vauk (1984) suggested that this could be the cause of death for as many as 20 % of clean birds found, but as few corpses were autopsied, this percentage could not be determined for our study. Alternatively, oiling rates may be overestimated by including birds oiled post-mortem (Kuyken 1978, Camphuysen and Franeker 1992). However, recent drift block experiments around the Newfoundland coast suggest that post-mortem oiling in this region is negligible, because only very few blocks recovered show signs of oiling (Wiese unpublished data). Additionally, as there is no difference in the persistence time of oil and unoiled birds on beaches (chapter 3), observed oiling rates appear to be a reasonable representation of a bird's risk of dying from spilled oil.

In addition to the high proportion of oiled birds found in our study area, the linear density of oiled birds on studied beaches (0.77/km) is also markedly higher than in other regions of the world during a comparable time period (0.02-0.33, Burger 1993a, Vaitakus et al. 1994, Heubeck 1995), and has remained at these high levels for the last 16 years.

2.4.2 Annual trends

Rates of oiling of seabirds in parts of the North Sea have declined significantly over the last 20 years (Camphuysen 1998, Camphuysen and Heubeck 2001). This was interpreted as an indication of a decline in overall chronic oil pollution, and attributed to more extensive aerial surveillance and decisions to clean-up oil slicks rather than wait for natural dissipation. While changes in North Sea oiling rates appear to reflect changes in the risk to birds of being oiled, this may not apply in the Newfoundland region. Observed oiling rates in our study indicate an increase in the proportion of oiled birds during the 16-year period of 1.4 % per annum. More detailed investigation, however, showed that this was mainly due to a reduction in the numbers of clean birds found on beaches, and this was related to increasingly warmer winter ambient air temperatures. The number of oiled birds found per kilometer showed no significant changes over time. Hence, the increasing oiling rates of murres appear to reflect a combination of a decrease in winter murre harvest leading to fewer deaths due to crippling, and improving weather during winter (including higher proportions of onshore winds) causing a likely reduction in natural mortality. Such a relationship supports the observation that processes on the winter grounds seem to determine Thick-billed Murre population dynamics (Gaston *in review*).

2.4.3 Ship traffic

Although ship densities are higher in certain regions and vessels generally tend adhere to defined shipping lanes (Lock et al. 1994), the data suggest that container and tanker vessels can be found throughout the area. Seabird oiling risks may occur in more areas in Atlantic Canada than previously defined (Environment Canada 1998). However, ship traffic volume was not related to overall oiling rates when changes in weather and the murre harvest were taken into account. This is somewhat surprising given the known origin of the oil found on birds, and may be an artifact of the limited dataset that does not include ships traveling under innocent passage. A more detailed analysis of ship traffic volume in Atlantic Canada is needed. But, if true, the lack of a relationship between oiling rates and shipping volume, given the increase in ship traffic and the constant number of oiled birds found, would suggest that it is a relatively constant number of ship operators illegally discharging their oily bilge into the ocean, rather than a consistent proportion of ships passing through the area. This result sends an encouraging message to local enforcement agencies and implies that stricter control of chronic polluters, increased fines for multiple offenders, and increased aerial surveillance, have a high probability of reducing the number of birds killed (Camphuysen 1998, chapter 6). Alternatively, a major reductions in spilled oil is needed before a relationship is apparent.

2.4.4 Monthly patterns

While decreases in the number of clean birds recorded seemed to explain annual trends of oiling rates since 1984, fluctuations in the number of oiled birds most closely explain monthly oiling rate patterns. The occurrence of oiled birds throughout the winter was highest during periods of cold ambient air temperatures, high wind speeds and frequent onshore winds. This is consistent with earlier findings (Kuyken 1978). High winds and cold ambient air temperatures may increase seabird mortality due to oil indirectly because it influences the amount of time oil remains on the surface of the water, and directly because of increased thermal and metabolic stress due to the lack of insulation and buoyancy. In addition, observations indicate that most birds have depleted fat reserves after bad weather periods, suggesting that it is hard to feed during such environmental conditions (R. D. Elliot pers. comm.). Alternatively, ships may accumulate more oily bilge water during bad weather conditions, a situation which may at the same time provide more opportunities to discharge these wastes without being detected. In this case, as for annual trends elsewhere in the world, oiling rates during the winter months may accurately represent the risk for birds to die due to oil. The relatively high numbers of oiled birds during months with frequent onshore winds indicates that oiled birds float ashore with wind-driven currents or fly ashore aided by the wind (Camphuysen 1989).

As in the North Sea (Camphuysen and Heubeck 2001), oiling rates are significantly higher in winter than summer (Wiese and Ryan 1999). Certainly, ships have more need to discharge bilges in winter because they accumulate more during storms and high seas, and ship operators have more opportunity to illegally discharge oil at sea in winter because they are less likely to be detected during the long periods of darkness or bad weather. However, it is doubtful that chronic oil pollution only occurs in the winter. Rather, lower oiling rates in the summer may reflect 1) reduced abundance of highly vulnerable birds such as murres, dovekies and eiders, 2) reduced overlap of nesting common murres with shipping lanes (Lock et al. 1994), 3) reduced susceptibility of seabirds to oil in warmer ambient and sea surface temperatures, 4) faster dispersion of refined oils at higher sea surface temperatures, and 5) faster loss of volatile and soluble toxic components of crude oils components and faster reduction of these oils to biologically inert solids such as tar balls at higher sea surface temperatures (Bourne and Bibby 1975, Kennish 1997). Observations of body condition and the degree of oiling of beached birds furthers strengthens this interpretation. Small quantities of oil on a carcass, combined with high emaciation, indicates that the bird lived for a period of time after contacting the oil before it perished, likely from hypothermia and starvation during harsh weather conditions (Camphuysen and Franeker 1992).

2.5 CONCLUSIONS

Data collected during beached bird surveys in southeastern Newfoundland were useful to determine trends in chronic oil pollution. However, the underlying patterns influencing oiling rates such as weather, ship traffic volume and, in Newfoundland, the murre harvest, must be investigated thoroughly to fully understand observed trends. Although weather parameters can explain variation in oiling rates, hunting plays a significant role for some species, and the influence of shipping needs to be further investigated. All these factors need to be compiled and considered for future analysis. To discern whether the data presented here are typical for the entire coastline or for all of Atlantic Canada, it is necessary that beached bird surveys be expanded geographically.

The Grand Banks provide suitable year-round feeding habitat for tens of millions of seabirds of many species, an abundance and diversity unparalleled in the North Atlantic (Nettleship and Birkhead 1985, Montevecchi and Tuck 1987). It is disturbing that oil continues to be a big threat to many seabird species in this region. Our results indicate that chronic oil pollution off the coast of Newfoundland is among the highest recorded in the world, and has remained high for the last 16 years. The damage caused to seabirds and the entire marine ecosystem through the continuing illegal flushing of oily bilge water from large container and tanker vessels along this coast is a significant and continuing problem.

To help reduce and hopefully eliminate this worldwide chronic problem, ship operators must be aware of a strong year-round enforcement presence in Atlantic Canada and elsewhere. The adopted polluter-pay principle must be fully applied and not succumb to political and economic pressure from large shipping companies. Clearly, as proven effective elsewhere, enforcement needs to increase, awareness and imposed fines (including minimum fines) need to be raised substantially, and accessible and convenient oil disposal facilities need to be established in Atlantic Canada to help reduce the illegal dumping of oil at sea (Averbeck et al. 1992, Camphuysen 1998, Wells 2001, Wiese 2002).

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CHAPTER 3 – ASSESSING SEABIRD MORTALITY FROM BEACHED BIRD SURVEYS: DEPOSITION, PERSISTENCE AND DETECTION RATES OF BIRDS BEACHED ON THE COAST OF NEWFOUNDLAND

3.1. INTRODUCTION

The assessment of total seabird mortality due to a catastrophic oil spill event, or due to chronic oil pollution over a long period, is a complicated process that requires knowledge of a variety of parameters. A starting point for many such analyses is the number of birds that wash up on shore, generally assessed through beached bird surveys (Camphuysen and Heubeck 2001). However, only a proportion of birds that die at sea make it to shore (Wiese and Jones 2001), and those that do can be covered by the sand, rocks or jetsam, get washed back to sea, or removed by scavengers before they can be tallied (Burger and Fry 1993). Thus, the use of beached bird carcasses to determine the total number of birds killed requires knowledge of 1) the proportion of the birds that die at sea which reach the shore, 2) the numbers and ratio of live to dead birds on beaches, 3) the fate of birds cast on the beach and 4) the detection probability of dead birds on beaches (Ford et al. 1987, Piatt et al. 1990, Burger and Fry 1993, Fowler and Flint 1997).

Difficulties in determining the proportion of those birds that die at sea which reach the shore result in large uncertainties in mortality estimates (Piatt et al. 1985). A rule of thumb of 10 % often gives erroneous estimates, due to pronounced regional differences and methodological deficiencies (Burger 1993a, Wiese and Jones 2001). Other researchers have called for better estimates of persistence, the amount of time a carcass remains detectable on the beach (Bodkin and Jameson 1991, Burger 1992). Unless beach surveys are conducted daily, the accurate determination of deposition rates, detection probabilities, and beached carcass persistence, is extremely important (Armstrong et al. 1978, Stowe and Underwood 1984, Camphuysen 1989, Piatt et al. 1990) and greatly influences the final estimate of any seabird mortality model (Page and Carter 1986, Burger 1992, Burger and Fry 1993).

Several researchers have used models that incorporate carcass persistence rates on beaches, to estimate the total number of seabirds killed following an oil spill (Ford et al. 1987, Page et al. 1990, Piatt et al. 1990, Burger 1993b, van Pelt and Piatt 1995, Fowler and Flint 1997), or after large wrecks of beached seabirds (van Pelt and Piatt 1995). The same approach, however, has not been applied to the worldwide ongoing problem of chronic oil pollution, a problem often assumed to be much larger and more detrimental to seabirds than occasional large spills (Hunt 1987, Burger 1992, Wells 2001, chapter 2).

Levels of chronic oil pollution are usually assessed by conducting regular beached bird surveys (chapter 2). In southeastern Newfoundland, Canada, such surveys have been conducted since 1984. Recent analysis of these data indicated that oil pollution levels in coastal Newfoundland waters are among the highest in the world, and that this problem has been ongoing for more than four decades (chapter 2, Montevecchi and Tuck 1987). Thick-billed (*Uria lomvia*) and Common Murres *(U. aalge)* made up 65% of all birds found on beaches between 1984-1999, with oiling rates of 80% during the last 5 years.

These species warrant special attention, given that murres are already subject to other anthropogenic impacts such as the Newfoundland turr hunt (Elliot 1991). In addition to knowing what proportion of birds found are oiled, it is also important to accurately determine how many are being affected by this illegal dumping of oil at sea.

It is the purpose of this study to addresses possible biases in observed oiling rates, determines deposition and detection rates on survey beaches in Newfoundland for the first time, and presents a new tool to analyze deposition, persistence, and detection data to contribute to total seabird mortality models that use counts from systematic beached bird surveys. This paper presents the results from six 10-day persistence trials carried out during winters of 1999/2000 and 2000/01 on beaches in southeastern Newfoundland. Our results present an important initial step in developing a model to determine total yearly seabird mortality due to chronic oil pollution in Atlantic Canada.

3.2 METHODS

3.2.1 Study Area

The island of Newfoundland is located at the easternmost extremity of North America and adjacent to the Grand Banks, the most important wintering ground for seabirds in the North Atlantic (Fig. 3.1). More than 40 million pelagic seabirds are estimated to reside or migrate annually through the waters surrounding insular Newfoundland, and the region is one the most productive marine areas in the world (Chardine 1995, Montevecchi 2000). Newfoundland also lies near major shipping routes between North America and Europe,

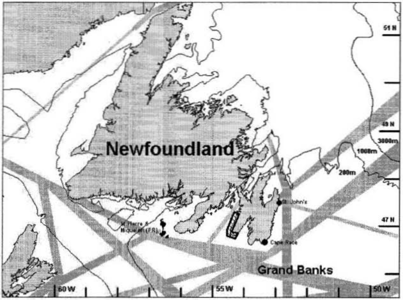


Fig.3.1 Study area in southeastern Newfoundland shown as cross-hatched coastline. Major shipping routes are shown as grey bars (adapted from Lock et al. 1994).

many of which converge off Cape Race, the southeastern tip of the island. The routes overlap with areas where large concentrations of seabirds occur, which are vulnerable to oil pollution (Chardine 1990), similar to areas in the North Sea (Skøv et al. 1995). As a result, the south coast of Newfoundland along with other regions in Atlantic Canada, were identified as extremely high oiling risk zones for seabirds by Environment Canada in 1994 (Lock et al. 1994). Illegal discharges of oil from bilges of large trans-Atlantic vessels have occurred in this region for decades (Tuck 1961), and although spilled oil undoubtedly affects the entire marine ecosystem (Kennish 1997), it is most visible in the ongoing presence of oiled birds on beaches.

3.2.2 Study design

Six beaches that are systematically surveyed during regular beached bird surveys by the Canadian Wildlife Service were chosen for this study. They were selected to provide a representative range of substrate type and exposure to the sea (Ford et al. 1987), and are referenced to here by their 2-letter beach code (Table 3.1). During the trials, large signs were posted at strategic access points to the beaches, informing people of the experiment and asking them to not remove any beached birds. In addition, several key people in each community were made aware of the experiments to further avoid human interference.

During March 1999, a pilot study was conducted on Big Barasway to determine appropriate sample sizes and method. Murres (*Uria* spp.) which had been confiscated were obtained from Canadian Coast Guard and Environment Canada enforcement

Table 3.1. Study beaches, description of beach substrates, general beach characteristics, and mean persistence time of bird carcasses on beaches in southeastern Newfoundland (t_p in days, includes adjusted persistence times for chickens, see results for details). All other parameters pooled. Persistence differed significantly among beaches χ^2_5=92.960. P < 0.0001.

Beach name	**Orientation**	**Beach code**	**substrate**	**General characteristics**	**# birds**	**Mean persistence time t_p**	**SE**
Big Barasway	W	BB	boulder	exposed, long, steep dynamic, unfrequented	163	2.85	0.23
Gooseberry Cove	W	GC	sand	in a deep cove, flat, Provincial Park	83	2.14	0.24
Patrick's Cove	W	PC	cobble	in deep cove, few nearby houses, dynamic, periodically large amounts of algae	144	2.29	0.16
St. Bride's	W	SB	cobble	exposed, in community	172	4.12	0.32
Point Lance	SW	PL	sand	exposed, flat, long, in community	140	4.33	0.36
Branch	SE	BR	boulder	exposed, steep, dynamic, in community,	148	3.45	0.25

officers. Murres are the species most frequently found during beached bird surveys and have the highest oiling rates in the area (chapter 2). Persistence was investigated for 12 or 24 carcasses, partitioned evenly between oiled and unoiled birds, and between those on the high and low tideline. It was determined that using 24 carcasses per beach was preferred. Scavengers did not seem to be satiated at either sample size, as birds disappeared in the same number of days whether 24 or 12 were initially deposited. Using 24 birds per beach also increased the sample size to allow the detection of statistical differences between oiled and unoiled birds, and those placed high and low on the beach, with 6 birds in each treatment.

The main experiments were conducted 6 times on all six beaches, in October and December 1999, February, October and December 2000, and March 2001. 840 carcasses were needed to carry out the experiments six times on all beaches simultaneously, depositing 24 birds per beach (except 20 birds on Patrick's Cove due to its smaller size). As insufficient murres were available, their numbers were supplemented with domestic fully-feathered chicken carcasses weighing between 700g and 1 kg, the range of emaciated and healthy murres (Wiese unpubl. data). In addition, carcasses of Dovekies (*Alle alle*) killed in a wreck in October 1999 (Wiese unpubl. data) were used in the study. In total, 58 murres, 173 dovekies, and 666 chickens were available. Altogether, most experiments were conducted using only chickens, on 6 occasions (BB in December 1999, 2000, and March 2001, and BR and PC in March 2001) chickens and murres were deposited in pairs to allow direct comparison and eventual correction for data obtained from chickens. On 7 occasions (BB, GC, PC, PL, SB in October 1999, and PL, SB in March 2001) chickens and dovekies were deposited in pairs.

All birds were intact, and thawed two nights before the experiments. Birds were individually marked with concealed 1 x 2 x 0.2 cm aluminum tags attached by transparent plastic tie wraps on the base of the right wing. Bunker C oil, commonly found on beached seabird carcasses in Newfoundland (A. R. Lock pers. comm.) was applied to 25% or more of the body of birds used as oiled carcasses. Birds were oiled when fully thawed, and allowed to dry and oxidize for 2 days under ambient conditions prior to deposition. This eliminated contamination of the beach with oil from the birds and simulated the natural weathering of oil that occurs on floating carcasses (Doerffer 1992).

3.2.3 Persistence and detection

Oiled and clean carcasses were deposited in 12 pairs along the low and high tideline during low tide at irregular intervals (10 – 30 m apart). Wings were folded to the body to mimic the position of most naturally beached birds, and to conceal the tag to minimize attraction of scavengers. Numbers and location of scavengers present at the time of deposition was noted. The beach was subsequently surveyed at low tide once a day for 10 days. The following data were recorded during each survey: presence or evidence of scavengers; sea state (Beaufort scale); wind velocity and direction; air temperature; precipitation; presence of oil, snow/ice or algae on the beach; and for each carcass present, scavenging code, location, body position and general description. Scavenging codes were defined as 0 (body intact), 1 (some flesh exposed, all organs present), 2 (body cavity open, some organs removed), 3 (some flesh present, all organs removed), and 4 (only feathers and bones remaining).

3.2.4 Natural deposition

The presence and condition of birds deposited naturally on beaches was noted daily during the experiments.

3.2.5 Data analysis

3.2.5.1 Persistence

Persistence time t_p, was defined as the number of days elapsed from the moment of deposition to the day it was recorded missing. Thus, birds that disappeared during the first day after deposition had a persistence time of 1. In cases where birds were not seen on one or more consecutive surveys but seen again thereafter, birds were assumed to have been present but not detected, and persistence time was adjusted accordingly (see detection below).

Differences in persistence time (t_p) among location, degree of oiling, date (month and year), and beach were analyzed using the SAS LIFEREG procedure (SAS/STAT 1990), which fits models to failure time data. The models for the response variable consist of a linear effect composed of the covariables together with a random disturbance term (SAS/STAT 1990). Mean persistence times (t_p), with associated errors, were calculated using the SAS LIFETEST procedure, as a normal ANOVA design would underestimate persistence times as our data were right-censored at 10 days. Nonparametric estimates of the survival (persistence) distribution, and rank tests for associations of the response variable with other variables within defined strata were obtained.

The daily persistence rate p, the probability that a carcass will remain on the beach between searches, was calculated by dividing the number of birds found on a given day (N_d) by the number of birds present during the previous survey (N_{d-n}):

$$p = \frac{N_d}{N_{d-n}} \qquad [1]$$

where d is the day of survey and n is the number of days since the previous survey.

We also calculated the persistence coefficient (s), traditionally used to estimate the number of carcasses deposited on the beach at day 0 (N_o), using the number of carcasses found (N_d) on the beach d days after initial deposition (Ford et al. 1987), where:

$$s = \left(\frac{N_0}{N_d}\right)^{\frac{1}{d}} \qquad [2]$$

We found s and p to vary over time as did van Pelt and Piatt (1995). A result, we used a regression equation to calculate s_t, a time dependent persistence coefficient, by solving equation [2] for N_o and substituting s with s_t. To test the accuracy of this calculation, we compared s to s_t by using the observed data for each day to calculate back to N_o following the method described by van Pelt and Piatt (1995). Again using the regression equation, we also calculated p_t, the time dependent persistence rate.

3.2.5.2 Natural deposition and detection

The mean number of birds found per day that were naturally deposited on beaches was calculated and used as a measurement of average daily deposition rates. Because in several instances birds were not found on one or more days, but were found again thereafter, we assumed these birds had been present throughout, but not detected. Birds did not usually remain undetected for more than 4 days. One bird, however, remained undetected for nine days. This was considered an outlier and not included in the analysis in order to not skew the results based on one bird. Detection probabilities (d_p), the daily probability of seeing a bird on the beach that is actually there, for different species, substrates, degree of oiling and days after initial deposition were calculated using the program MARK (White and Burnham 1999). Days after initial deposition was also used to determine whether the degree of scavenging affected detectability, as scavenging rates of birds present increased over time (see results). Our model set included time dependence, oiling, species and substrate as potential factors influencing detection rate (i.e. capture rates in MARK). Point estimates were extracted from the most parsimonious models achieved, by selecting models with low AIC (Akaike Information Criterion) values (Burnham and Anderson 1998).

We did not extract persistence rates p (i.e. survival) from MARK as our data were right-censored (see section on persistence above). Means are presented as ±1 SE, unless otherwise indicated, and all tests are 2-tailed.

3.2.6 Persistence model

Deposition, detection, and persistence rates have to be integrated to accurately calculate the number of carcasses present on the beach between systematic beached bird surveys (Fowler and Flint 1997). We therefore constructed a simulation model in MATLAB. In this model, the deposition process was simulated by randomly drawing a Poisson-distributed variate based on a measured mean deposition rate. To determine whether a deposited bird would persist to the next day, a value k was drawn from a random uniform distribution between 0 and 1 and compared to the time dependent daily persistence rate p_t (equation from Fig. 2b). If p_t was greater than k on any given day, the bird persisted to the next day; otherwise it disappeared from the beach. If it persisted, the bird was added to any birds deposited the next day and the process repeated for the specified time interval, always using p_t for each bird based on its initial date of deposition. This process was repeated for 300 beaches, as approximately 300 beach surveys are carried out in a winter season in Newfoundland. This entire process was repeated 10,000 times. Based on the distribution of pseudovalues, a mean with 95% confidence intervals for the number of birds deposited during each trial (actual number deposited) was calculated for each possible number of birds left on the last day of the interval (number of birds found on a survey). Finally, these means were corrected based on the probability of detection (d_p).

3.3 RESULTS

3.3.1 Natural deposition

A total of 29 new birds was found during 166 beach surveys carried out on six beaches during two winters. All birds were dead when found. The mean natural deposition rate of was 0.44 ± 1.43 birds / km. Deposition rates per km were not significantly different among beaches (χ^2_4 =6.98, $P>0.1$) nor among substrates (χ^4_4 =5.89, $P>0.05$). The overall mean daily deposition rate was 0.175 ± 0.516 birds/beach and this distribution fit a Poisson distribution well (Kolmogorov-Smirnov $D=0.022$, $P=1.0$).

3.3.2 Persistence

Birds disappeared from beaches through removal of whole carcasses by scavengers, sequential scavenging, burial in the substrate, and possibly due to backwash into the sea. Most carcasses (75.6 %) disappeared from the beach without prior signs of scavenging (Table 3.2) and were thought to have been carried off by scavengers, buried in the substrate by wave action, or washed back to sea. Carcasses that did not disappear intact were scavenged at various rates. Scavengers seemed to prefer fresh carcasses over older ones, as there were significant differences in persistence times (t_p) depending on the degree of scavenging (χ^2_3 = 393.00, $P<0.0001$, Table 3.2). Scavenging took place mostly on the ventral surface of the bird, either at the height of the intestines or through the throat, resulting in a quick removal of the head. This pattern seemed consistent regardless of the location of oil on the body.

Table 3.2. Mean persistence time (t_p) in days for carcasses on beaches in southeastern Newfoundland subject to different degrees of scavenging at the time of disappearance. Mean values of t_p differed significantly from each other $\chi^2=392.00$, $P < 0.0001$. Data used include adjusted t_p for chickens, all other parameters pooled.

Degree of scavenging[1] at time of disappearance	**# birds**	**mean t_p (days) ± SE**	**%**
0	643	2.1 ± 0.1	75.7
1	27	5.3 ± 0.7	3.2
2	58	4.7 ± 0.4	6.8
3	32	6.5 ± 0.7	3.8
4	90	8.2 ± 0.4	10.6

[1] 0=body intact, 1= some flesh exposed, all organs present, 2= body cavity open, some organs removed, 3= some flesh present, all organs removed, 4= only feathers and bones remaining

Scavengers seen on beaches included Herring and Great Black-backed Gulls (*Larus argentatus* and *L. marinus*), American Crows (*Corvus brachyrhynchus*), dogs (Canus *familiaris*) and Red Foxes (*Vulpes vulpes*). The presence of foxes was confirmed by prints in the snow, while the others were seen on more than 10 occasions leaving carcasses upon approaching the beach to commence the survey. Given the pattern of scavenging, it appeared that carcasses were predominantly eaten by gulls and crows, while dogs and foxes appeared to remove carcass from the beach in one piece. No movement of carcasses between beaches was observed, and movement of carcasses on the same beach was minimal and restricted mostly to relocation by wave action between the high and low tideline.

Ninety five percent of all experimental birds disappeared during the 10-day experiments. There were no significant differences in persistence times (t_p) between oiled and unoiled birds for all species combined, nor for each species separately, but birds of each species persisted longer if located on the higher rather than on the lower tideline (Table 3.3). Overall, persistence times (t_p) of chickens were significantly shorter than those of murres and dovekies, while murres and dovekies were not significantly different from each other (Table 3.3). Chickens persisted 45.2 % less time than seabirds on the higher tideline and 39.3 % less on the lower. Therefore persistence days for chickens were adjusted upward depending on location in order to include chicken carcasses in the overall analysis. Once corrected, oiled and unoiled bird were pooled, and all birds combined persisted on average for 3.3 ± 0.1 days.

Table 3.3. Summary of mean persistence time (t_p) in days and statistical test results among species, location of carcasses on beaches, dates and years for birds deposited on beaches in southeastern Newfoundland. Results presented for location on beach, dates, years and overall include adjusted persistence time (t_p) for chickens (see results). Tide lines were pooled unless otherwise indicated.

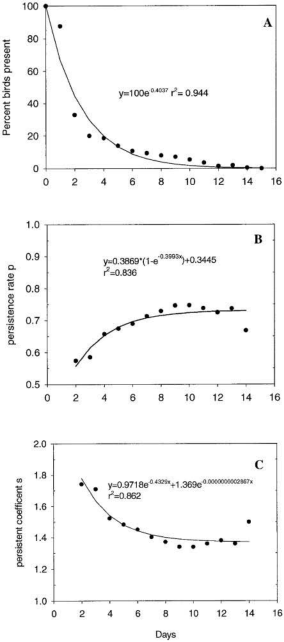
Parameter	**n**	**mean t_p**	**SE**	χ^2	**P**
Oiled	515	2.5	0.1	2.5	0.112
Clean	335	2.7	0.1		
Chickens low tide	328	1.8	0.1	57.0	< 0.001
Chickens high tide	314	2.9	0.2		
Murres low tide	26	2.5	0.5	6.7	0.010
Murres high tide	29	4.2	0.5		
Dovekies low tide	19	2.5	0.5	0.8	0.373
Dovekies high tide	134	3.2	0.2		
Chickens	642	2.3	0.1	11.1	0.001
Dovekies	153	3.2	0.3		
Murres	55	3.4	0.4	9.0	0.003
Chickens	642	2.3	0.1		
Murres	55	3.4	0.4	0.7	0.417
Dovekies	153	3.2	0.3		
Low tide	373	1.9	0.1	59.4	< 0.001
High tide	477	3.1	0.1		
Oct 1999	281	2.3	0.1	465.1	< 0.001
Dec 1999	111	6.9	0.4		
Feb 2000	109	4.9	0.4		
Oct 2000	120	1.6	0.0		
Dec 2000	113	2.8	0.2		
Mar 2001	116	2.7	0.2		
Winter 1999-2000	501	3.9	0.2	86.9	< 0.001
Winter 2000-2001	349	2.3	0.1		
Overall		**3.3**	**0.1**		

Corrected persistence time still differed significantly between birds from the higher and lower tideline, and among beaches, dates and years (Table 3.1, 3.3). On average, birds at the lower tideline persisted 1.4 days less than birds at the higher tideline, but there were no patterns among beaches (Table 3.1) or dates (Table 3.3). Overall, birds persisted significantly less during the winter of 2000-01 than during the winter of 1999-2000 (Table 3.3), a difference possibly explained by heavier snow cover and colder air temperatures during the 2000-01 season.

We combined data for all birds (including adjusted values for chickens) to determine persistence rates (*p*) and persistence coefficients (*s*) that were representative for the area of Newfoundland where systematic beached bird surveys are carried out. We thus integrated existing natural variation among beaches, location of carcasses on beaches, seasons and years. Due to the upward correction of persistence for chickens (see Methods) no accurate values remained to for day 1. As a result, no data is shown for this day in Fig. 3.2.

Over 60 % of carcasses disappeared from beaches within 2 days of initial deposition, and disappearance followed an exponential decline (Fig. 3.2a). The median daily persistence rate (*p*) for all birds combined was 0.689 ± 0.057, but was better described by a time-dependent function (Fig. 2b). Similarly, the median persistent coefficient (*s*) was 1.451 ± 0.151, again best described by a time-dependent function (Fig. 2c). Using the time dependent persistent coefficient (s_t) clearly provided a better prediction of the number of

Fig. 3.2 The overall pattern of disappearance of carcasses from experimental beaches (A, fitted with a single exponential decline), the change in persistence rate (B) and persistence coefficient (C) over time, fitted with an exponential rise (B) and a double exponential decay (C).


A
Percent birds present
100
80
60
40
20
0
0 2 4 6 8 10 12 14 16
y=100e^-0.4037 r^2= 0.944
B
persistence rate p
1.0
0.9
0.8
0.7
0.6
0.5
0 2 4 6 8 10 12 14 16
y=0.3869*(1-e^-0.3993x)+0.3445
r^2=0.836
C
persistent coefficent s
2.0
1.8
1.6
1.4
1.2
1.0
0 2 4 6 8 10 12 14 16
y=0.9718e^-0.4329x+1.369e^-0.0000000002867x
r^2=0.862
Days

birds initially deposited (N_o) than using the median (s), deviating from the actual value by only 1.5 ± 29.3 %, while predictions using s deviated by 34.8 ± 65.0 % over a 14 day period (Fig. 3.3). Higher accuracy and errors below 10 % were achieved once survey intervals remained at 7 days, as using s_t underestimated N_0 by only 0.3 ± 7.4 %.

3.3.3 Detection

The best fitting model (lowest AIC, i.e. lowest ΔAICc) for detection probabilities (d_p) was one dependent on beach type and species, and was strongly supported with an AIC weight (wAICc) of 0.92 (Table 3.4). On several occasions carcasses were found with only parts of one wing above the substrate, and on sandy beaches a mound in the sand was sometimes the only indication of the presence of a bird. There was no evidence for differences in detection probabilities (d_p) between different days (i.e. degrees of scavenging) or degree of oiling. Because scavenging of birds increased with time (Table 3.2), the lack of difference in d_p between different days was also considered as good evidence for a lack of difference in d_p between degrees of scavenging. The difference in detection due to species was largely due to one cell (oiled murres on cobble beaches) with an extremely low sample size of 2. Based on the ranking of the models with only substrate and only species, we consider the evidence for species specific detection rates to be weak. Combining species, birds were best detected on boulder beaches, followed by cobble and sand (Table 3.5). Overall detection probability (d_p), taking into account substrate variation, was 84.8 ± 1.3% (SE).

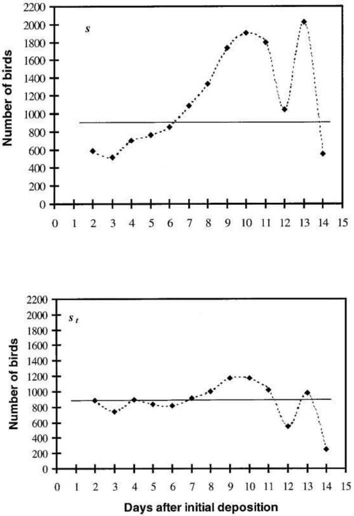


Fig. 3.3 Calculation of the initial number of birds deposited using the median persistence coefficient (s) and the time dependent persistence coefficient (s_t). Initial number of birds deposited is shown as the straight line.

Table 3.4. Model fitting results based on Akaike Information Criterion for persistence and detection rates of bird carcasses on beaches in southeastern Newfoundland.

Model	ΔAICc	wAICc	# parameters
$p_{\text{days}}\ d_{p(\text{species*substrate})}$	0.00	0.91837	16
$p_{\text{days}}\ d_{p(\text{species*substrate*oil})}$	4.84	0.08158	22
$p_{\text{days}}\ d_{p(\text{days})}$	20.18	0.00004	19
$p_{\text{days}}\ d_{p(\text{substrate})}$	24.14	0.00001	13
$p_{\text{days}}\ d_{p(\text{species})}$	26.62	0.00000	12
$p_{\text{days}}\ d_{p}$	26.86	0.00000	11
$p_{\text{days}}\ d_{p(\text{oil})}$	28.87	0.00000	12
$p_{\text{days}}\ d_{p(\text{oil*substrate})}$	29.44	0.00000	16
$p_{\text{days}}\ d_{p(\text{oil*species})}$	30.16	0.00000	14
$p_{(\text{species*substrate*oil})\text{*days}}$, $d_{p(\text{species*substrate*oil})}$	82.64	0.00000	132
$p_{(\text{species*substrate*oil})\text{*days}}$, $d_{p(\text{species*substrate*oil})\text{*days}}$	114.74	0.00000	188

Table 3.5. Detection probabilities (d_p) for carcasses on different substrate types. All other categories pooled.

Substrate	d_p	SE	lower 95% C.I.	upper 95% C.I.
sand	0.7924	0.0288	0.7302	0.8433
cobble	0.8552	0.0218	0.0807	0.8930
boulder	0.8786	0.0189	0.8365	0.9110
ALL	**0.8480**	**0.0131**	**0.8204**	**0.8719**

3.3.4 Persistence model

The persistence model was meant to be representative for the entire study area and was built on overall averages to include variation due to substrate, date, and location of carcasses on the beach. Based on the daily average deposition rate per beach, the overall time dependent probability of birds persisting on the beach from one day to the next (p_t), the overall mean detection probability (d_p), and assuming a survey interval of seven days, we were able to estimate average correction factors for the number of birds found on weekly beached bird surveys in the study area (Table 3.6).

3.4 DISCUSSION

The persistence of birds can vary among month and years even on the same beach. Due to the heterogeneity of beaches and conditions through the year, it is difficult to compare data from different locations (Burger 1992). Variation within eastern Newfoundland and comparable variation world-wide (Table 3.7) points to the necessity to determine simultaneous daily persistence rates for a series of representative beaches within an area.

When dead or dying birds reach the shore, they are only detectable for limited time before they disappear from the beach due to scavenging, removal by predators and scavengers, through burial in the beach substrate, or backwash into the sea (Ford 1987, Burger and Fry 1993, van Pelt and Piatt 1995). The disappearance of beached bird carcasses from a beach is related to environmental conditions such as wind, sea state, tidal levels,

Table 3.6. Comparison between the number of birds found on the beach on a weekly beached bird survey, and the estimated total number deposited during the survey interval, corrected for time dependent persistence (p_t), and mean deposition rates and detection probabilities (d_p).

# birds found	Total # deposited	SE	lower 95% C.I.	upper 95% C.I.
0	1.00	0.08	0.85	1.15
1	2.17	0.13	1.93	2.42
2	3.34	0.29	2.81	3.96
3	4.53	0.83	3.54	6.49
4	5.68	1.04	4.72	8.25
5	6.92	1.12	5.90	9.43
6	8.02	0.76	7.08	9.43

Table 3.7. Seabird carcass persistence rates from various studies (Burger and Fry 1993, van Pelt and Piatt 1995)

Location	Species	Study Duration	No. carcasses	Daily persistence rate p	Reference
Alaska	Oiled King Eiders	5 days	177	0.52 - 0.99	Fowler and Flint (1997)
Alaska	Murres	100 days	398	0.93 (0.89-0.98)	Van Pelt and Piatt (1995)
Alaska	Oiled seabirds, mostly murres	2 days	198	0.84	Jones (1989)
Alaska	Radio-tagged seabirds	8 days	9	0.80	ECI (1991)
Alaska	Radio-tagged auklets	7 days	23	0.47	ECI (1991)
Alaska	Oiled alcids	no data	no data	0.80 - 0.84	Piatt et al. (1990)
British Columbia	Seabirds	4 days	12	0.54	Burger *in* ECI (1991)
British Columbia	Intact chickens	3	no data	0.84	Humphries (1989)
British Columbia	Chicken portions	2-3	no data	0.31-0.44	Dale (1989)
British Columbia	Shearwaters and kingfishers	4 days	10 and 2	0.45	Burger (1991)
Washington	Common Murres	5 days	81	0.74	Ford et al. (1991)
Washington	Guilllemots, de-oiled	4 - 5 days	81	0.47 - 0.84	ECI (1991)
California	Oiled alcids	3 days	235	0.59 (0.38 - 0.72)	Page et al. (1990)
California	Seabirds	11 month	71	0.96 (0.89-0.99)	Bodkin and Jameson (1991)
Newfoundland	**Oiled and clean murres and chickens**	**15 days**	**850**	**0.69** **(0.57 - 0.75)**	**This study**
Newfoundland	Dovekies and gulls	weekly *	Unknown	weekly: 0.05-0.26	CWS (unpubl. data)
Newfoundland	Murres and eiders	30 days	40	0.26 - 0.58	Rodrigues (1995)
Belgium	Seabirds	no data	no data	0.93	Kuyken (1978)
Netherlands	Alcids	5 months	44	0.98	Camphuysen (1989)

* Carcasses were marked during weekly surveys that were conducted during 2 winter seasons (November- March)

temperature and precipitation; and to factors related to scavengers, such as carcass availability, and scavenger presence, density and satiation; and to beach characteristics such as substrate, slope, and orientation (Camphuysen 1989, Bodkin and Jameson 1991, Burger 1993b, Rodrigues 1995, van Pelt and Piatt 1995, Fowler and Flint 1997). To determine the number of birds deposited on a beach between surveys requires the knowledge and integration of region-specific deposition and persistence rates, as well as detection probabilities.

3.4.1 Deposition rates

The rate of deposition of carcasses on a beach has often been determined after large oil spills, when it shows a strong decline after some peak period (Page et al. 1990, van Pelt and Piatt 1995). During normal conditions, however, deposition rates are most likely related to seasonal abundance of birds, wind regimes and cold weather periods. Our estimated annual deposition rate of 5.0 birds/km of surveyed beach compares well with similar estimates determined elsewhere (Table 3.8). However, because this rate was determined in winter, it is likely a slight overestimate, as about 15 % more birds are found during winter than summer (Wiese and Ryan 1999).

Estimation of deposition rates are a concern when establishing persistence of birds on beaches (Ford et al. 1987, Bodkin and Jameson 1991). Contrary to Bodkin and Jameson (1991), we found no significant differences in deposition rates among beaches or substrates, although some variation occurred. The south coast of Newfoundland is made up of steep rocky shorelines, infrequently interrupted by small pocket beaches that are

Table 3.8. Annual deposition rates determined during systematic beached bird surveys in different areas. Our estimate was based on a natural deposition during winter of 0.014 birds/km/day.

Location	# birds/km/year	Reference
Puget Sound (WA)	2.9	Speich and Wahl 1986
California	5.4	Bodkin and Jameson 1991
Gulf of Mexico	3.5	Simons 1985
Newfoundland	5.0	This study
Western Atlantic	5.2	Simons 1985
Shetland	4.5	Heubeck 1995
Belgium	3.7	Kuyken 1978

collect jetsam and bird carcasses. Ford et al. (1987) pointed out that it is important to include every combination of coast and beach type in a study of carcass deposition, and that daily counts on representative beaches is the best way to estimate the total number of carcasses that arrive on shore in a given area. Given our range of substrates and exposures, and the levels of variation we detected, we feel that the overall deposition rate per km is representative for the region and can be used in future studies.

3.4.2 Persistence

Daily carcass persistence rates (*p*) measured were within the range of estimates measured elsewhere (Table 3.7). The variation in results from studies around the world may be attributed to a variety of factors, including geographic location, weather, season, beach substrate, exposure, tidal ranges, density and activity of scavengers, carcass detection probabilities, sample sizes, species, and study duration (Camphuysen 1989, Bodkin and Jameson 1991, Burger 1993b, Rodrigues 1995, van Pelt and Piatt 1995, Fowler and Flint 1997). It is essential to determine persistence rates for a specific area to correctly interpret the number of birds found on local beaches, and to address potential differences in persistence and detection rates between oiled and unoiled birds. For the first time, we present conclusive evidence that persistence and detection rates do not differ for oiled and unoiled birds. Thus, with the exception of problems associated with post-mortem oiling and birds with internal oiling that appear clean (Camphuysen and Franeker 1992, Burger 1993a, chapter 2), two factors that seem to offset each other (chapter 2), oiling rates detected during regular beached bird surveys appear to be accurate.

We did, however, detect a major difference in rates of persistence for birds located at the low versus high tideline. All birds from the lower tideline disappeared in less than 2 days, while 5 % of carcasses still remained on the high tideline after 10 days. This difference is most likely due to more wave action at the low tideline over a longer period causing a higher probability for burial in the substrate and potential backwash of deposited birds out to sea. Backwash and subsequent sinking may be a factor influencing persistence if birds deposited on beaches sink after being re-floated. We are, however, not aware of any empirical evidence to support this. On the other hand, backwash and subsequent deposition did not seem to be a frequent occurrence, as almost no movement of birds along the beach was observed and no birds that had disappeared from one beach were ever found on another beach. Relocation of birds on the same beach was almost exclusively restricted to birds from the low tideline moving straight up to the high tideline subsequent to the first high tide after initial deposition. The location of a bird on the beach may indicate how long it has been there.

In addition to differences in persistence due to location on the beach, we found variation between days, among beaches and seasons. We concur with Fowler and Flint (1997) that it is the combined effect of weather and scavenging that causes these variations between beaches and seasons. During similar environmental conditions, however, day to day variation in persistence most likely reflects scavenger density, preference for fresh carcasses and possibly satiation.

3.4.3 Detection rates

Estimation of detection rates of carcasses is another major concern when establishing persistence rates (Fowler and Flint 1997). Comparable to other studies, we found different detection probabilities (d_p) among substrates, finding at best 87.9 % of birds on boulder, and at worst 79.2% on sand beaches. As detection probabilities (d_p) did not vary over time as scavenging of birds increased, we did not encounter a bias towards lower number of birds found as scavenging progressed during the trial, as reported by Fowler and Flint (1997). However, weather parameters, such as snow fall, could also influence detection probabilities (d_p) on beaches of different substrates in different ways, and perhaps explain the differences we detected. In our study, about 15 % of birds were missed during any given survey, although detection probabilities (d_p) are likely observer dependent and should ideally be established for each person conducting surveys, to allow accurate correction of the data.

3.4.4 Persistence model

The use of time dependent persistence coefficients (s_t) proved to be the most accurate method to estimate the total number of carcasses that washed ashore between surveys (N_0). Due to the low average persistence of 3.3 days, estimates of N_0 are most reliable if surveys are conducted within 7 days. However, using the persistence rate alone is clearly insufficient. Deposition and surveyor dependent detection rates must be estimated when assessing the total number of carcasses deposited based on counts from systematic beached bird surveys. As van Pelt and Piatt (1995) pointed out, the major flaw in extrapolation models is the assumption that all carcasses found on a given day are those persisting from a single cohort deposited the day after the last survey, when they are actually comprised of carcasses from many different cohorts deposited over the survey interval.

To overcome this resulting bias, Bodkin and Jameson (1991) assumed that deposition occurred midway between surveys, but only changed the direction of the bias by doing so. We believe that we can avoid any such bias by simulating deposition based on the distribution of observed deposition rates from a variety of beach types, and having them persist in a time dependent manner based on their initial day of deposition. The model we applied to weekly surveys indicates that, in some cases, the actual number of birds occurring on the beach between surveys can be more than double the number of birds found.

To our knowledge, estimates of persistence, deposition and detection rates have never been applied together to interpret results from systematic beached bird surveys. Estimating those rates for a given area allows such an approach, when taking into consideration variation among location of carcasses on the beach, beach substrate, times of year, and different years, differences in detection rates between surveyors. It is essential to apply these correction factors to survey data, as it will have large implications for any estimates of total oiled seabird mortality.

3.5 CONCLUSIONS AND RECOMMENDATIONS

An accurate method to interpret the number of birds found during beached bird surveys is by integrating a simulation approach based on area representative data, with estimates of region-wide time-dependent persistence rates, and detection probabilities calculated using Capture-Mark-Recapture methods. To assess seabird mortality due to oil, the total estimated number of birds that arrived on a beach between surveys can be multiplied by the proportion of oiled birds that were detected, because detection and persistence rates do not differ between oiled and clean birds. Although biological impacts of low-level, chronic oiling on populations are still poorly understood (Burger and Fry 1993), we can use this model as a starting point to determine seabird mortality due to continuous marine oil pollution in Atlantic Canada. This methodology can also be used to aid in the assessment of similar mortality elsewhere.

As deposition, persistence and detection rates will vary from those determined here should a large oil spill occur in the area, detailed studies need to take place immediately after such a spill (van Pelt and Piatt 1995, Fowler and Flint 1997). Our approach can be used in such instances, as well as in the interpretation of systematic beached bird survey results. As such, ours is an objective methodology with global applications for the assessment of stranded birds. Model parameters, however, need to be assessed locally so that the model can be applied in different regions. Instances where deposition of carcasses may be substantially higher will yield estimates for these numbers beyond the example presented here.

However, the number of birds that reach the shore represents only a small fraction of those birds that die at sea. Additional regional studies are needed to assess sinking rates for seabird carcasses, as well as overall proportions of birds lost at sea, to evaluate total seabird mortality due to ongoing marine oil pollution in Atlantic Canada.

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CHAPTER 4 – EXPERIMENTAL SUPPORT FOR A NEW DRIFT BLOCK DESIGN TO ASSESS SEABIRD MORTALITY FROM OIL POLLUTION

4.1 INTRODUCTION

The effect of chronic oil pollution and large oil spills on seabirds is frequently significant (e.g. Piatt et al. 1990) and usually assessed by counting dead oiled birds that wash up on shore. Birds that die at sea may sink, drift away from shore, decompose at sea, get scavenged, or may be overlooked after they wash ashore on beaches (Ford et al. 1987, Page et al. 1990, Hlady and Burger 1993). Estimates of the proportion of those birds that actually die at sea that both reach the shore and are retrieved during beach surveys (Proportion of Dead Birds Retrieved, PDBR) are necessary to estimate the total number of birds killed by oil. This is true regardless of whether the seabird mortality event resulted from a large catastrophic oil spill or from a small deliberate or accidental release of oil at sea. Attempts to estimate PDBR have included several experiments using carcasses (Coulson et al. 1968, Hope Jones et al. 1970, 1978, Lloyd et al. 1974, Bibby and Lloyd 1977, Bibby 1981, Stowe 1982, Threlfall and Piatt 1982, Page et al. 1982, Piatt et al. 1990) and wooden drift blocks (Threlfall and Piatt 1982, Piatt et al. 1985, Hlady and Burger 1993, Chardine and Pelly 1994, Flint and Fowler 1998). Those experiments were conducted with different bird species and block types, from different distances from shore, in varying environmental conditions (ocean currents, sea and air temperatures, wind speeds), seasons, and geographical regions. Consequently, results for onshore recoveries varied from 0 - 59 % and 0 - 66%, for carcasses and drift blocks, respectively (Table 4.1) and can not easily be directly compared. Generally consistent conclusions stated in the existing literature were: 1) wind seems to be the principal factor determining carcass and block drift movement (2.2 - 4% of wind velocity), and 2) currents, especially tidal currents, may influence deposition on specific beaches once the drifting carcass or block approaches the shore.

The greatest uncertainty in estimating the number of birds killed by oil arises from not knowing what proportion of birds oiled actually reach shore and are recovered (PDBRs, Piatt et al. 1985). Although drift block experiments may provide biased estimates of seabirds that drift ashore (due to e.g. increased detectability, passive drift versus birds flying to land), they are crucial for estimating PDBRs. Use of drift blocks has been criticized in the past because they produce poor estimates of mortality by not taking into account the number of oiled birds that actively fly or swim to shore, and because they do not mimic bird-drift appropriately (Hlady and Burger 1993). Ford et al. (1987) outlined a method of how to take the former into account in mortality estimates after large oil spills. Hlady and Burger (1993) varied the length of softwood drift blocks to mimic different species of alcids, and Burger (1991) conducted a buoyancy experiment of oiled carcasses and determined that most alcid carcasses sank within two weeks. Nevertheless, relevant drift characteristics of birds and drift blocks have never been compared and lack of drift similarities between seabird carcasses and previously used drift blocks have never been addressed experimentally. It is sometimes assumed, without supporting evidence, that

Table 4.1. Summary of the results of carcass drift and drift block experiment conducted in Europe and North America (adapted from Piatt et al. 1990).

Location	Distance off-shore released (km)	Bird species	Drift block type (cm)	n	% recovered
England[1]	?	Shag	-	?	25
English Channel[2]	?	Gulls	-	144	20
Irish Sea[3]	< 100	Alcids	-	400	20
Irish Sea[4]	< 100	Gulls	-	347	59
				300	11
				305	44
Irish Sea[5]	?	alcids, gulls	-	?	7.5
North Sea[6]	10 - ?	gulls	-	600	9.8
North Sea[7]	?	gulls	-	?	40.8
	?	gulls	-	?	11.3
Alaska[8]	10	murres	-	100	3
Alaska[9]	6	-	9x9x20	302	0.7- 61
California[10]	?	alcids	-	63	0
	?	gulls, alcids	-	186	29.9
British Columbia[11]	1-2	-	4x9x10/20/40	300	43 - 53
	35 – 56	-	4x9x10/20/40	150	18.6
	86 – 116	-	4x9x10/20/40	150	0.6
Newfoundland					
St. John's to Hibernia[12]	10 – 500	murres	-	115	0
		-	10x10x20	400	0
Cape Race to Sable Is.[12]	?	alcids	-	129	0
		-	10x10x20	600	24
off Cape St. Mary's[13]	50	-	10x10x20	100	30 - 66
Placentia Bay[14]	5	-	10x10x20	120	7

References: [1]Coulson et al. 1968, [2]Hope Jones et al. 1978, [3]Hope Jones et al. 1970, [4]Bibby and Lloyd 1977, [5]Lloyd et al. 1974, [6]Bibby 1981, [7]Stowe 1982, [8]Piatt et al. 1990, [9]Flint and Fowler 1998, [10]Page et al. 1982, [11]Hlady and Burger 1993, [12]Threlfall and Piatt 1982, [13]Piatt et al. 1985, [14]Chardine and Pelly 1994.

about 10% of seabirds that die at sea as a result of an oil spill reach the shore (Tanis and Morzer Bruijns 1968, Bourne 1970, National Research Council 1985, Canadian Coast Guard 1998), and so results of beach bird surveys conducted following after oil spills have sometimes been multiplied by that PDBR factor. We concur with Burger (1993) who pointed out that the 10% PDBR figure is rarely likely to be correct due to the high degree of weather dependence and distance from shore at the time of the pollution event, buoyancy loss, and because drift blocks used in the past had few of the drift characteristics of seabird carcasses. The aim of our study was to experimentally compare drift properties of different block designs to seabird carcasses at sea and thus develop a drift block with appropriate properties for useful estimation of PDBR.

4.2 METHODS

4.2.1 Drift block design, theory and practice

For two bodies to float and drift equally through a medium under the same circumstances, physical characteristics such as drag, density, mass, and projected areas exposed to wind and ocean currents need consideration. We used murres (*Uria* spp.) as a comparison for the drift block design, because they are the most vulnerable seabird species to oil pollution in the northern hemisphere (Camphuysen 1989, Wiese 1999, Wiese and Ryan 1999), and represented over 63% of all the birds found dead on Newfoundland shores between 1984 and 1997 (Wiese 1999, Wiese and Ryan 1999, chapter 2). Murres are approximately 37 cm long from bill tip to tail tip and 10 cm wide at their widest point on the back. Dead, but not decomposed, oiled murres collected during beached bird surveys had an average mass of 650 g, a density of 0.76 g cm^{-3}, and 150 $cm^2$and 50 cm^2 exposed to water and wind, respectively. A commonly used drift block in the past (Y block) measures 9 x 9 x 20 cm (or 10x10x20cm), weighs approximately 1000 g, has a density of 0.5 g cm^{-3} , and exposes 130 cm^2 to the water and wind (Table 4.2).

Drag (resistant force exerted on a body parallel but contrary to its movement; Kundu 1990), is essential when comparing the drift velocity of two bodies. The general equation for drag is:

$$\text{Drag} = ½ * c_d * A * \rho * u^2 \qquad [1]$$

where c_d is the drag coefficient, 'A' the projected area exposed to the flow, ρ the density of the medium (water or air) and u the fluid velocity (water current; Streeter 1981). Because medium density and fluid velocity are equal for all bodies, the relevance lies in the differences between the drag coefficient (which is particular to the shape and the surface texture) and the projected area to the water flow. In the case of a murre, whose shape can be approximated by a cylinder (c_d = 1.2; Streeter 1981) with a length of 25 cm, a diameter of 8 cm, and a projected area to the water flow of 150 cm^2 (25 * 6 cm, where 6 cm is the part of the body submerged in water), this 'effective drag' (ED) equals 90 cm^2; where ED is the drag on the portion of the body immersed in water, without taking into consideration 'ρ' or 'u', (calculated as ED= ½*c_d*A) they are equal for all.

Table 4.2. Comparison of relevant physical characteristics of murre carcasses and different drift block designs.

Physical characteristic	Murre carcasses	9 x 9 x 20 cm (Y block)	9 x 9 x 15.5 cm (S block)	4 x 9 x 30 cm (L block)	9 x 9 x 14.5 cm (W block)
Mass (g)	650	1000	695	380	1000
Density (g cm^{-3})	0.76	0.50	0.50	0.50	0.85
Effective drag (cm^2)	90	78	60	60	102
Projected area in water (cm^2)	150	130	101	60	102
Projected area in air (cm^2)	50	130	101	60	29

Effectively, it is the projected area submerged (in cm^2) that gets impacted by the drag through the water. Because Y blocks float with the waterline along their diagonal axis, they are also most closely approximated by a cylinder, but of different size ($A = 130$ cm^2), resulting in an ED of 78 cm^2. The main difference between murres and Y blocks seemed therefore not to be ED but mass and area exposed to wind (Table 4.2).

Past studies suggested wind force to be the most important factor for block drift (Bibby and Lloyd 1977, Bibby 1981, Hlady and Burger 1993, Flint and Fowler 1998). As denser types of wood are not readily available locally in Newfoundland, we chose to shorten the Y block to approximate wind exposed area and mass, while keeping ED within two-thirds of the murres. This new shorter block (S) was 15.5 cm long (Table 4.2). Another way to reduce the wind-exposed area while keeping ED within two thirds of the murres was a 4 x 9 cm softwood block (L block). It was found that a 30 cm long L block achieved that (Table 2), because they float flat in the water as a rectangle, changing the drag coefficient (c_d) to 2 (Streeter 1981). As a result, each of the three block types approximated the physical characteristics of a dead murre in one or more ways (Table 4.2).

4.2.2 Drift block experiment

We tested 11 blocks of each type (Y, S, L) in the field against six murre carcasses that were freshly dead when collected, had been immediately frozen, and thawed during the night before use. All 33 blocks were uniquely marked and dropped simultaneously with the murres from an inflatable boat in Conception Bay, Newfoundland, between Bell Island and the town of St. Philip's on 11 September 1999 (Fig. 4.1). Every 15 min, the position of the birds and the different blocks types were recorded with a hand-held GPS (Garmin GPS III, Garmin, Olathe, Kansas), as was wind speed and direction. Separation and spread was determined. 'Separation 'was defined as distance separating blocks of different types and carcasses, whereas 'spread' was distance separating blocks or carcasses of the same type. After 4 hours, the experiment was stopped and the blocks and birds collected. A second similar field experiment was performed on 6 December 1999 using an additional modified block design based on the results of the first experiment. This new block (W block) was weighted down with 450 g of steel (1.25 x 5 x 8.75 cm) attached to one side of the block with a zinc-coated screw. That addition of extra mass resulted in a block with 2 cm of the block exposed to the wind and made them float flat (c_d=2; Streeter 1981). Both drag though the water and the mass of the block increased as a result, so we shortened the block to 14.5 cm to partially compensate for that change (Table 4.2).

4.3 RESULTS

Birds and blocks drifted with the direction of the wind, but at different speeds. During the first experiment, block types started to separate from each other (separation) after about 30 min, and a clear pattern of separation became apparent after 75 min. Birds moved the slowest, whereas especially the L blocks drifted very quickly (Fig. 4.1). Despite initial use of six murre carcasses, it was only possible to follow two for a longer period of time.

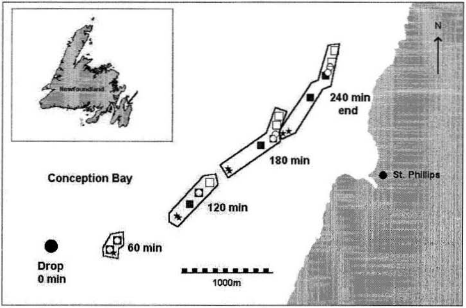

Fig. 4.1. Study area in Conception Bay, Newfoundland, Canada. Comparison of drift between Y blocks (open circle), S blocks (solid square), L blocks (open square), and murres (stars), 11 September 1999. 33 blocks and six murres were dropped. Positions are shown every hour. Points for blocks represent the extremes of their spread. Four murres were lost after 30 min, so only two were tracked the entire time.

Birds floated very low in the water, mostly with their dark-colored backs up, making them difficult to detect again after a 15 min period. The experiment was ended after 4 hours because the separation between blocks and birds made it increasingly difficult to track the objects. After 4 hours, L blocks were furthest away from the birds, followed by the Y blocks. S blocks clearly approximated murre carcass drift the closest (Table 4.3, Fig. 4.1). They stayed within 360 - 600 m of the birds, while the Y and the L blocks were at much greater distances (690 - 740 m and 780 - 850, respectively).

Observations during the experiment clearly confirmed that the predominant factor driving movement of drift blocks was wind, and hence the area of the block exposed to it. Although L blocks accounted for that, they seemed too light, so the experiment was repeated replacing L blocks with the new W blocks (Table 4.4, Figure 4.2).

During this second trial, W blocks clearly showed the closest proximity to the birds because several blocks drifted among them throughout the experiment. Y and S blocks separated from birds after only 30 min and behaved the same way as previously. Winds during both experiments were southwest, yet wind speeds varied (9 - 20 km h^{-1} and 15 - 25 km h^{-1} in the first and second trial). Unfortunately, the second experiment had to be ended after 105 min due to bad weather and the increasing separation between birds and blocks, making it difficult to track the objects. After 105 min, the separation of the W blocks from the carcasses was 20 - 80 m, whereas the S and Y blocks were at 170 - 250 m and 330 - 370 m respectively.

Table 4.3. Movement (distance in meters) from the drop point for murre carcasses and different drift block types during the trial on 11 September 1999. Distance ranges reflect the spread within each block type. No range means no spread or just one bird.

Time after drop (min)	Murres	Y blocks	S blocks	L blocks
60	810	630-740	630-740	630-740
105	1280-1400	1970	1790-1830	1820-1970
120	1500	1790	1630-1750	1780-1920
180	2250	2780-2790	2670-2740	2900-2960
240	3020	3710-3760	3380-3620	3800-3870

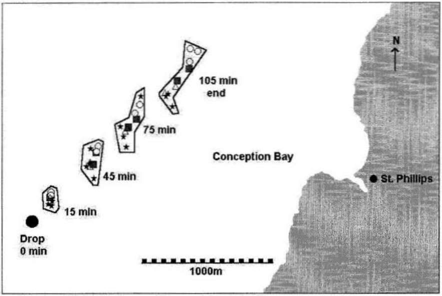

Fig. 4.2. Comparison of drift between Y blocks (open circle), S blocks (solid square), W blocks (triangles), and murres (stars), 6 December 1999. 30 blocks and five murres were dropped. Positions are shown only at set intervals for clarity. Points for blocks represent extremes of their spread. One murre was lost after 75 min, and one after 90 min.

Table 4.4. Results of the second drift block calibration experiment conducted on 6 December 1999. Movement (distance in meters) from drop point for murre carcasses and different drift block types. Distance ranges reflect the spread within each block type. No range means no spread.

Time after drop (min)	Murres	Y blocks	S blocks	W blocks
15	180-230	250-270	230	200-220
45	570-710	730-760	640-720	600-630
75	910-1250	1130-1200	1030-1110	970-1010
105	1370-1410	1700-1780	1540-1660	1390-1490

4.4 DISCUSSION

To quantitatively evaluate the effect of chronic or large oil spills on seabirds, it is essential to quantify the number of oiled birds that die at sea. One method to estimate PDBR involves use of wooden drift blocks. Previously collected oiled carcasses are not suitable for that purpose, because they have already been exposed to variable periods of drift, begun to decompose, and therefore no longer have the same buoyancy as a seabird recently killed by exposure to oil (Burger 1991, chapter 5).

To minimize error in drift times and path induced by using a wooden block rather than a bird, we designed a drift block type that closely mimicked murre carcass drift. Several relevant physical characteristics of murre carcasses and drift blocks were examined and empirically compared. Both trials showed a clearly different drift behaviour between blocks and birds. Y blocks used in the past, which approximated ED and area exposed to the water, were found at 690 - 740 m from the birds in the first trial after 240 min and at 330 - 370 m after 105 min in the second trial. Extrapolated over a two week drift period (which is less than the recovery time of blocks usually included in results of past studies) assuming constant wind direction and speed, they would be located between 58 - 71 km from the murre carcasses. Such a distance could clearly subject birds and blocks to different weather and tidal regimes and puts the relevance of past studies using this block type in serious doubt. S blocks, a shorter unweighted version of the Y blocks, approximated murre mass and were intermediate between the areas exposed to wind and water of murre carcasses. Although that did result in smaller distances to birds than Y blocks, extrapolated again over two weeks assuming constant wind speed and direction, they would be located 30 - 48 km away from the birds and hence be similarly inaccurate. The reason for that difference between carcasses and S blocks appeared to be the substantial difference in surface area exposed to wind, a physical characteristic that, together with their minimal mass, seemed to be the most relevant factor creating differences between the drift of blocks and carcasses. This seemed to be substantiated by the L blocks, that did approximate wind exposed area, but were very light, resulting in the greatest distance from the murres of all block types tested.

W blocks met the right criteria. They approximated murres in ED, had a small wind-exposed area and were of an approximate mass. Most of them stayed with carcasses throughout the experiment or at a maximum distance of 80 m. Again translated over a two week period, they would mostly be with the carcasses or a maximum of 15 km away. Given a spread of 100 m among the seabird carcasses over the same time period, we believe this separation was acceptable.

4.5 CONCLUSION

Drift-block experiments are a useful tool in assessing the proportion of seabirds that die at sea and drift to shore. They are an important component in the estimation of overall avian mortality caused by an oil spill incident or through chronic oil pollution throughout the year. We found that blocks used in the past drifted significantly faster than seabird carcasses, so past results may have misinterpreted drift times of bird carcasses and possibly provided correspondingly misleading estimates of mortality and drift patterns. Because drifting carcasses are subject to buoyancy loss through water-logging and scavenging, and may not stay afloat longer than two weeks (Burger 1991), the accuracy of drift time estimates is essential. Using W blocks as described here is not only likely to accomplish that, but may also allow the modification of mathematical models for carcass drifts. This would result in the description of more accurate carcass drift-paths for forecasts and to determine the possible origin of the oil retrospectively. We strongly recommend that all future drift-block studies carried out in areas where murres are a dominant victim of oil pollution use the W block. In areas where other seabird species are the predominate victims of oil pollution, similar experiments to ours should be carried out to design drift blocks that match relevant physical drift properties of the species involved.

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CHAPTER 5 – SINKING RATES OF DEAD BIRDS AT SEA: IMPROVING ESTIMATES OF SEABIRD MORTALITY DUE TO OILING

5.1 INTRODUCTION

The number of birds killed as a result of an oil spill is usually estimated from the number of birds found on beaches. Two measurements are essential in order to extrapolate from that number accurately: the number birds dying at sea which do not reach the shore, and the number that subsequently disappears on the beach (Ford et al. 1987, Piatt et al. 1990, Burger and Fry 1993, Fowler and Flint 1997, chapter 3). Many efforts have been made in different locations where oil spills have occurred to determine rates of carcass persistence on beaches, and lately some of these studies have included assessments of both deposition and detection rates (Fowler and Flint 1997, Van Pelt and Piatt 1995, chapter 3). However, even when the total number of beached carcasses is known or can be well estimated, a large fraction of the total mortality remains unmeasured due to dead birds being lost at sea (Ford et al. 1987).

The main methods used to estimate the number of birds lost at sea are carcass drift and drift block experiments. Wiese and Jones (2001) pointed out that experiments using carcasses previously washed up on shore significantly overestimated loss at sea, because buoyancy of these birds will have already decreased significantly. On the other hand, results from drift block experiments tend to largely underestimate loss at sea, because they generally use blocks that do not accurately mimic seabird drift (Wiese and Jones

2001) and do not take into account sinking rates of carcasses floating at sea. Thus, with the exception of a study by Hlady and Burger (1993), where only blocks recovered within one month after the drop were included, most studies have included all recoveries in their subsequent mortality models, even if these occurred months after blocks were deployed. To include all block recoveries without a cut-off period can greatly influence final estimates. It is thus imperative to accurately determine sinking rates of bird carcasses at sea, and to investigate whether these may differ between oiled and clean birds. This could potentially lead to subsequent biases in the proportion of oiled birds found, and hence in the total seabird mortality estimated due to the spilled oil.

Murres are the most vulnerable species to spilled oil in the Northern Hemisphere and generally the most common victim after large and chronic oil spills (Camphuysen and Heubeck 2001, chapter 2). The purpose of this study was to test sinking rates of murres (*Uria* spp.) under natural conditions, and to accurately quantify possible differences in sinking rates due to different degrees of oiling. Sinking rates were also determined for beached carcasses with different degrees of oiling, to investigate whether one of the mechanisms of bird removal from beaches could be back wash into the sea and subsequent sinking. The importance and timing of scavenging on all sinking rates was investigated. Accurate cut-off times after which recoveries from drift block experiments should no longer be considered in estimating total seabird mortality due to oil were determined, and their influence on past mortality estimates investigated.

5.2 METHODS

5.2.1 Experimental design

Murres were collected from hunters during the 1999/2000 hunting season or were from confiscations made by Environment Canada enforcement officers. To ensure the integrity of carcasses, murres were frozen immediately after collection, and only those with minimal damage to the plumage were used.

Birds were randomly assigned into clean, lightly oiled or heavily oiled categories. Because all birds available for this study were clean, birds were manually oiled on the ventral side using weathered Bunker C (details in chapter 3), a type of oil commonly found on stranded seabirds in the study area. Birds in the lightly oiled category were covered with oil on about 25% of their bodies, while birds assigned to the heavily oiled category were covered with oil on roughly 50% of their bodies. Within each category, three further groups were randomly assigned: intact, scavenged, and beach. Each group was subject to a different treatment throughout the experiment (see below). A minimum sample size of 6 was needed to determine one-day differences in the number of days afloat between groups. This lead to a total of 18 birds required in each category, and a total of 54 birds were used in the experiment.

The experiment was conducted between 10 November and 3 December 2000. A floating three-chambered wooden-framed pen (3 x 1.5 x 1.5 m), was constructed and attached to a wharf in a partially-sheltered bay in Long Pond, Conception Bay, Newfoundland, Canada.

The pen was lined with chicken wire to keep potential scavengers away, and to contain birds in the cage once they sunk. All inside walls were lined with oil adsorbent material provided by the Canadian Coast Guard Oil Response Team in St. John's Newfoundland, to collect any oil which may have come off the bird plumage. Twelve birds of each oiling category (6 for the intact, and 6 for the scavenged treatment) were placed in each chamber. The remaining six birds per oiling category (beach treatment) were placed in a mesh-screened pen on the wharf.

After nine days, the six birds assigned for the scavenged treatment in each floating chamber, and all 18 birds on land, were opened to simulate partial scavenging (i.e. body cavity cut open and pulmonary air sacs punctured). After 19 days, all flesh and muscles were removed from remaining birds to simulate complete scavenging.

Changes in buoyancy were tested daily by determining the amount of added weight (in 5 g increments) necessary to sink them (Burger 1991, cited in Burger and Fry 1993). In order to simulate the effects of high wave conditions during testing, birds were agitated in the water to allow the escape of large air bubbles trapped in the plumage. Environmental condition (air and water temperature, wind speed, precipitation) were recorded during the study. After 25 days the experiment was terminated because all floating birds had sunk, and all remaining beach birds had frozen solid.

5.2.2 Data analysis

Differences in sinking rates among birds subjected to different degrees of oiling, degrees of simulated scavenging, and interactions between the two, were analyzed using ANOVA/ANCOVA designs . Linear regression analysis was used to determine average daily buoyancy losses. Means are presented with ± 1 SE, and all tests are 2-tailed. Significant differences between groups and categories were determined at $P<0.05$.

5.3 RESULTS

5.3.1 Weather

Environmental conditions varied throughout the experiment. Large rainfalls occurred during the first week where ambient temperatures varied between 3^o and 7^o C. After day 10, temperatures dropped to between 0^o and -4^o C. Snow fell on seven days during the second and third week of the experiment. Birds on land were often covered in frost and snow and were partially or fully frozen for 6 days between day 14 and 25. Water temperatures ranged between 1^o and 4^o C. Overall, these conditions are considered average for this area in November (Gerard Morin, Atlantic Climate Centre, Fredericton, New Brunswick, Canada, personal communication).

5.3.2 Birds floating

Birds varied in buoyancy at the beginning of the experiment, but there was no relationship between the initial body mass and initial buoyancy (slope = -0.106 ± 0.076, $P>0.1$; Fig. 5.1). Birds in the floating pen quickly lost buoyancy during the first few days

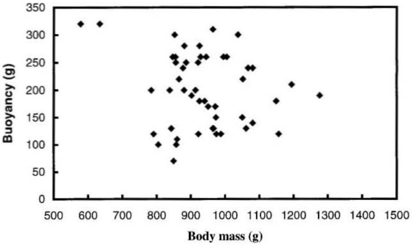


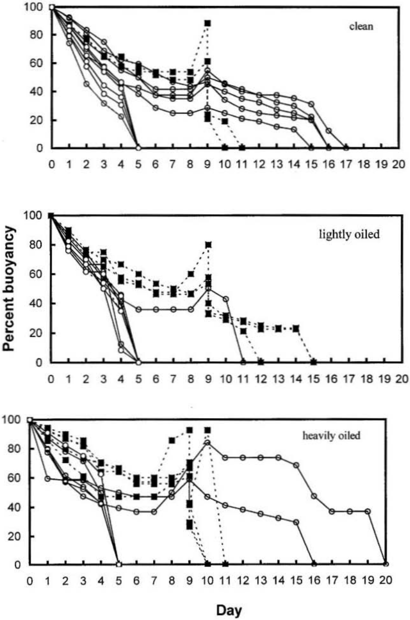
Figure 5.1. Relationship between the initial body mass and initial buoyancy of 54 murre carcasses used in the experiment. Buoyancy is expressed as the additional weight needed to sink birds.

(Fig. 5.2). Changes in the proportion of unscavenged clean, lightly oiled and heavily oiled birds that remained afloat followed a step-wise decline over the 20-day period. After only five days, 70 % of carcasses had sunk, and all had disappeared within 20 days (Fig. 5.3). Over 36 % (13 of 36) of carcasses that had previously sunk (all at day 5) resurfaced between day 7 and 9. These carcasses proceeded to lose buoyancy again quite quickly, especially if scavenged at day 9. Ninety percent (27 of 30) of carcasses still afloat experienced an average buoyancy gain of 45.9 ± 38.4 % between days 7 and 9. The first time of sinking for all re-surfaced birds was taken as the actual sinking day because under natural conditions birds would sink to a depth from which re-floating would not occur (see discussion).

The number of days that unscavenged birds remained afloat was not significantly different among birds with different degrees of oiling overall ($\chi^2_2 = 4.58$, $P>0.1$), and averaged 8.2 ± 5.2 days. Although no differences in days afloat were detected between heavily oiled and mildly oiled birds ($\chi^2_1 = 1.56$, $P>0.1$) nor between both oiled categories pooled and clean birds ($\chi^2_1 = 3.34$, $P>0.05$), lightly oiled birds sank significantly faster than clean birds ($\chi^2_1 = 5.40$, $P<0.05$, Table 5.1).

Birds subject to partial simulated scavenging at day 9 lost an average of 60.0 ± 21.3 % of their buoyancy. Two heavily oiled birds sank in less than a day after scavenging. Daily buoyancy losses, calculated as the slope from individual linear regression lines, did not

Figure 5. 2. Cumulative buoyancy loss of clean, lightly oiled and heavily oiled floating birds. Dotted lines (solid squares) indicate birds subject to partial simulated scavenging at day 9, solid lines (open circles) indicate birds left intact.


clean
lightly oiled
heavily oiled
Percent buoyancy
100
80
60
40
20
0
0 1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20
Day

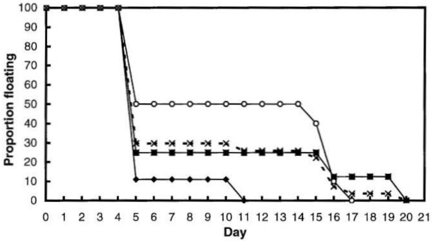


Figure 5.3. Proportion of clean (open circles), lightly oiled (solid diamonds), heavily oiled (solid squares), and all (x on dotted line) unscavenged murres in an experimental floating pen, afloat over a 20 day period, in southeastern Newfoundland, winter 2000. Each line is an individual bird.

Table 5.1. Mean number of days afloat for unscavenged murres in an experimental floating pen in southeastern Newfoundland, winter 2000.

Category	**Mean (days)**	**95% C. I.**
Clean	10.5	6.3-14.7
Lightly oiled	5.7	4.1-7.2
Heavily oiled	8.3	3.1-13.4
All	**8.2**	**6.2-10.3**

differ significantly between birds with different degrees of oiling (χ^2_2 =2.47, P>0.1), and averaged -12.3 ± 5.8 % per day.

Comparisons among birds partially scavenged at day 9, showed, in contrast to unscavenged birds, that lightly oiled birds remained afloat significantly longer (χ^2_2 =13.97, P<0.001, 5.0 ± 0.7 days) after they were scavenged than birds in the other two oiling categories (clean 1.5 ± 0.7 days, heavily oiled 1.3 ± 0.5 days). Regardless, scavenged birds for all degrees of oiling sank significantly faster after scavenging than birds left intact (χ^2_2 =12.00, P<0.001), disappearing on average within 2.6 ± 2.1 days. Unscavenged birds remained afloat for another 6.9 ± 2.5 days after day 9.

5.3.3 Birds on beach

Unlike floating birds, birds on beach lost their buoyancy slowly during the initial days (Fig. 5.4), decreasing 4.6 ± 0.4 % per day. There were no significant differences detected in daily buoyancy losses among birds with different degrees of oiling (χ^2_2 =3.49, P>0.1). Almost 78 % (14 of 18) of birds experienced an average buoyancy gain of 18.3 ± 17.1 % between day 7 and 9, and another gain of 140.7 ± 93.7 % at day 17, the latter due to freezing. At day 9, birds lost an average of 43.2 ± 7.3 % of their buoyancy due to scavenging, and those scavenged again at day 19 lost, on average, another 66.0 ± 13.8 %. Birds which were fully scavenged at day 19, irrespective of the degree of oiling, showed a significantly lower degree of buoyancy loss compared to birds left partially scavenged

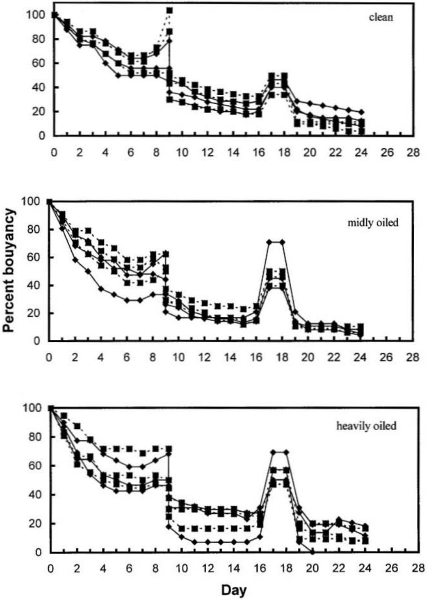
(χ^2_2 =11.7, P<0.001), averaging a daily loss of 0.7 ± 0.6 %, compared to 2.8 ± 1.8 % for partially scavenged birds. After 24 days, birds on land retained 9.4 ± 5.1 % of their initial buoyancy, and only one bird sank during the duration of the study (Fig. 5.4).

5.4 DISCUSSION

5.4.1 Birds floating

The importance of assessing the number of seabirds lost at sea due to sinking was first recognized by Hope Jones et al. (1970) while trying to estimate total seabird kill after the *Hamilton Trader* oil spill in 1969. It was determined that at least 50 % of drifting murres sank within two weeks. Based on the results, Ford et al. (1987), applied a constant loss-rate for murres drifting at sea due to sinking of 15 % per day. This same per day loss rate was also used by Page et al. (1990) to assess the number of seabirds killed after the *Apex Houston* spill in 1986 off the California coast. It was pointed out in both studies that this rate may not be constant, and that loss at sea may vary among species, with more neutrally buoyant birds such as loons, alcids and cormorants sinking more quickly than birds such as gulls that float higher in the water (Ford et al. 1987, Burger and Fry 1993). Consequently, Ford et al. (1996) compared sinking rates of free-floating radio-tagged carcasses in Alaska and found no differences between large and small alcids (murres and auklets, respectively).

Figure 5.4. Cumulative percent buoyancy loss of clean, lightly oiled and heavily oiled birds on land. Dotted lines indicate birds subject to partial simulated scavenging at day 9 and complete simulated scavenging at day 19, solid lines indicate birds only subject to partial simulated scavenging at day 9.


clean
midly oiled
heavily oiled
Percent bouyancy
Day
0
20
40
60
80
100
0
2
4
6
8
10
12
14
16
18
20
22
24
26
28

This latter supports our finding that body mass did not influence the number of days that oiled murres remained afloat, even though differences among species were not investigated in this study. Results presented here are thus applicable to seabird mortality due to large spills as well as to chronic spills. They also apply to birds that die immediately, as well as to those that live for a period of time after contacting the oil and deplete their fat reserves in the process before dying. In contrast, I found that clean birds in poor condition (i.e. lower body mass) floated longer than those with plentiful body fat and large pectoral muscles. Perhaps this difference is accounted for by the high proportion of dense muscle mass in birds in good condition, as completely scavenged birds that remained on the beach, but had been reduced to feathers and bones, still floated in the absence of muscle tissue.

Freshly dead and intact seabirds floated, but began to sink rapidly during the first five days when large amounts of air trapped in the plumage was replaced by water. As bodies decompose, gases may get trapped inside the body cavity and increase buoyancy (Burger 1991). This was seen in this study between days 7 and 9. Because the floating birds were placed in a cage with a bottom that held them at approximately 1 m below the surface, buoyancy gained by decomposition was enough to re-surface them. In reality, such a scenario is unlikely as sinking bodies are subjected to large increases in relative atmospheric pressure in the first 10 m, and would eventually become negatively buoyant. The substantially increased pressure would cause any air remaining in the plumage, and decomposing gases in the body, to escape. Also, carcasses would most likely be scavenged by fish and invertebrates once at or near the bottom, which would further increase the loss of buoyancy, and prevent sunk birds from re-surfacing.

When intact, lightly oiled birds had faster sinking rates than clean birds, although no evidence was found that sinking rates differed between clean and heavily oiled birds nor between birds with different degrees of oiling. The effects of oil on buoyancy seem twofold. On one hand, large amounts of oil on the plumage increases the overall density of the bird, largely due to emulsification of the oil with seawater to form a dense mousse (National Research Council 1985). On the other hand, oil may trap the air in the plumage and partially slow down water-saturation. For heavily oiled birds, these processes seemed to be balanced, so that they sank at comparable rates to clean birds. However, the plumage of lightly oiled birds seemed not able to keep enough air trapped to counteract the added density due to oil. This finding is consistent with results from previous studies (Ford et al. 1991). If birds remain unscavenged at sea, this result implies that the number for birds that arrived dead on the beach is potentially biased towards clean birds, because 75 % of oiled birds found on beached bird surveys show only light oiling (chapter 2). If birds are however scavenged at sea, the reverse may be true, as lightly oiled birds floated significantly longer than clean birds after they were scavenged. Clearly more information on scavenging rates of birds at sea is required. In the absence of such information, however, it is plausible that overall, the two processes offset each other, and that the proportion of oiled birds assessed on beached bird surveys reflects the actual proportion of birds oiled at sea (chapter 2).

Estimates of accurate sinking rates for seabirds at sea are sparse. Ford et al. (1991) found that tethered murres in sheltered and exposed seas in Oregon remained afloat for an average of 8.4 and 7.6 days respectively (about 13% lost per day), with more than 90 % disappearing within 14 days. Burger (1991) tested heavily oiled Common Murres, Ancient Murrelets (*Synthliboramphus antiquus*) and Horned Grebes (*Podiceps auritus*) in a tank simulating benign sea conditions. He found that the former two species were similar and measured sinking rates of 2 % per day, while the latter sank more quickly at 9 % per day. He concluded that a loss exceeding 5 or 10 % of oiled carcasses at sea per day (i.e. all birds lost after 10-20 days) is probable under normal at sea conditions. Experiments during spring with free-floating radio-tagged murres and auklets in Prince William Sound showed median days afloat of 15-20 days, but included many birds that washed up and remained on the beach for up to 5 days before re-floating (Ford et al. 1996). The present study, however, showed that birds on the beach retain their buoyancy significantly longer than when they are afloat, biasing that study towards longer floating rates. An identical experiment in the summer in the Gulf of Alaska, where no birds beached, showed median days afloat of 7, 9, 11 and 18 days respectively (Ford et al. 1996). The latter higher value was attributed to much calmer sea conditions.

The number of days that birds remained afloat determined in the present study therefore comparable to the findings of Ford et al. (1991) and to the Gulf of Alaska study by Ford et al. (1996), even though the authors point out that tethered birds and previously frozen birds may sink faster than fresh carcasses. Although the initial loss in our experiment was greater, as 70 % of birds sank within 5 days, overall daily linear sinking rates in this study were comparable, at slightly over 12 %. Some researches have questioned whether sinking rates are linear (Ford et al. 1987, Page et al. 1990) and Ford et al. (1996) fitted a Weibull distribution to their data. The declines observed in the current study were certainly steeper during the first 5 days, but as most birds were lost during this early period, a linear model to assess overall loss-at-sea due to sinking seems appropriate, at least under winter conditions in Newfoundland.

Ninety five percent of all unscavenged floating birds in this study sank within 10 days. This value is comparable to the median sinking rates of Ford et al. (1996) if the 18 day period observed under calm conditions are omitted. Winter conditions in Newfoundland include extremely cold temperatures and frequent periods of high winds, factors that would likely lead to faster sinking rates than those observed during calm summer months in Alaska (Ford et al. 1991). For murres, and probably other auks, as shown by Ford et al. (1996), I recommend using a 10 day estimate for birds floating at sea, and to assume daily sinking rates of 10-14 % in a model to estimate proportion of birds lost at sea. Such rates will result in conservative estimates of overall mortality because 70 % of birds sank within the first five days, and birds may be scavenged at sea at any time after death, in which case buoyancy loss would occur approximately 2.5 times faster than if birds remained intact. Scavenging of carcasses at sea has not been quantitatively documented, but has often been observed by duck and murre hunters, especially close to shore.

5.4.2 Application to past drift block experiments

It seems reasonable that studies using drift blocks to assess the numbers of birds lost at sea only include blocks that come ashore within 10 days after they are dropped, as 95 % of birds in this experiment sank within that time period. Wiese and Jones (2001) summarized results of past drift block experiments and examined possible biases of recovery estimates due to faulty drift block design. I conclude that recovery estimates from most past drift block experiments are generally large overestimates, because they included blocks that were recovered far after seabird carcasses would have sunk. The degree to which these estimates are biased varies, and little information is generally provided about chronology of block arrival on shore to allow for a re-interpretation in light of the present findings.

Threlfall and Piatt (1982) concluded that roughly 24 % (0-66 %) of birds that die in offshore Newfoundland waters eventually reach beaches, and acknowledged the high variance and dependency on weather. However, only 5 % of blocks set adrift reached shore within a period of 10 days (Threlfall and Piatt 1982). Chardine and Pelly (1994) concluded that 7 % of birds that died in Placentia Bay, Newfoundland during their study arrived ashore, but included blocks recovered up to almost 2 years after being dropped. Very few arrived during the first 2 weeks (J. W. Chardine pers. comm.). Flint and Fowler (1998) concluded that up to 61 % of birds could be recovered after an oil spill close to St. Paul Island, Alaska during onshore winds. However, systematic surveys to find blocks did not start until 11 days after they were dropped at sea, likely making this estimate high. Finally, Burger (1991) illustrated the need for integrating accurate sinking rates of birds at sea into oil spill mortality models, and calculated a 23 % higher mortality estimate from the *Nestucca* spill (equal to 6860 birds) after he accounted for birds lost at sea due to sinking.

5.4.3 Birds on beach

As expected, carcasses resting on the shore lost their buoyancy at a much slower rate than those at sea. Scavenging, which usually occurs almost immediately after carcasses are cast ashore under natural conditions (chapter 3), sharply decreased buoyancy, although not to the point where sinking took place. In addition the buoyancy of carcasses that freeze after being washed ashore may be markedly increased for long periods of time. In this study, with the exception of one bird, all carcasses left on the beach remained afloat after 24 days. Because average persistence rates of beached carcasses in most areas is less than 24 days (chapter 3), and because scavenged birds are likely to retain some buoyancy in the absence of dense muscle mass, I conclude that subsequent sinking after backwash into the sea is not likely to be a significant mechanism of carcass removal from beaches. However, backwash and subsequent burial in the beach substrate is likely (chapter 3), because birds of the beach can retain positive buoyancy for long periods of time.

5.5 CONCLUSIONS

Drift block experiments, if interpreted correctly, are the most accurate and efficient way to determine the proportion of those birds that die at sea which reach the shore. Based on the present results, in areas where murres, or similarly-sized alcids are the main victims of oil spilled at sea, I recommend that only those drift blocks that reach the shore within 10 days after they are dropped at sea be included in modeling oil seabird mortality. This is still a conservative approach, recognizing that most birds may sink within 5 days.

In areas where other species are commonly killed by marine oil pollution, it would be useful to determine their sinking rates using the approach presented here, although evidence exists that it may not vary between large and small alcids (Ford et al. 1996). This study points to the importance of determining scavenging rates of birds floating at sea, as they strongly influence sinking rates and may further explain possible biases in the proportion of oiled birds found on beaches.

5.6 ACKNOWLEDGMENTS

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CHAPTER 6 – DRIFT BLOCK EXPERIMENTS TO ASSESS SEABIRD MORTALITY RATES DUE TO CHRONIC OIL POLLUTION OFF THE COAST OF NEWFOUNDLAND

6.1 INTRODUCTION

Seabirds are the most visible victims of catastrophic and chronic marine oil spills. Despite the obvious impact, it is often difficult to accurately assess the extent of seabird mortality, because only a small proportion of birds that die at sea wash up on shore. A large proportion of birds sink, get scavenged or drift away from shore, and so will never be tallied during beached bird surveys (Ford et al. 1987, Page et al. 1990, Hlady and Burger 1993a, chapter 3). Several attempts have been made to assess the proportion of birds that die at sea and which do reach shore, by using experiments with seabird carcasses (Coulson et al. 1968, Hope Jones et al. 1970, 1978, Lloyd et al. 1974, Bibby and Lloyd 1977, Bibby 1981, Stowe 1982, Threlfall and Piatt 1982, Page et al. 1982, Piatt et al. 1990) or wooden drift blocks (Threlfall and Piatt 1982, Piatt et al. 1985, Hlady and Burger 1993, Chardine and Pelly 1994, Flint and Fowler 1998). Resulting estimates have varied between 0 – 59 % and 0 – 66 % of drifting birds reaching shore, respectively (Wiese and Jones 2001).

Wiese and Jones (2001) pointed out that using seabirds previously cast ashore for such experiments is inappropriate due to prior buoyancy loss of the carcasses. Wiese and Jones (2001) also showed that most drift block designs used in the past did not accurately mimic seabird carcass drift and designed and tested blocks that did. Their experiments showed that although drift block experiments are a useful way to determine the proportion of birds lost at sea, results of most past studies are unusable due to faulty block designs. In addition, buoyancy loss experiments showed that 95 % of murre (*Uria* spp.) carcasses floating at sea sink within 10 days (chapter 5). As a result, only blocks that arrive on shore within that time period should be considered, in order to accurately estimate the proportion of birds that die at sea and reach the shore.

Regardless of these discrepancies, past studies concur that onshore recoveries are heavily influenced by environmental conditions (particularly wind speed and direction and ocean currents), distance from shore, and geographical regions. They also show that wind seems to be the principal factor determining carcass and drift block movement largely due to wind induced surface currents, and that tidal currents may influence deposition on specific beaches once drifting carcasses or blocks approach the shore. Consequently, reported onshore recovery rates can not easily be compared and need to be assessed for each region. Both Burger (1993c) and Wiese and Jones (2001) pointed out that the 10 % rule-of-thumb used in the past (Tanis and Morzer Bruijns 1968, Bourne 1970, National Research Council 1985, CCG 1998) is largely unjustified.

Most seabird mortality estimates have been related to catastrophic oil spills (Ford et al. 1987, Page et al. 1990, Piatt et al. 1990, Burger 1993b, Van Pelt and Piatt 1995, Fowler and Flint 1997). However, chronic oil spills may be more important to long-term seabird population stability than occasional large spills (Hunt 1987, Burger 1992, Nur et al.

1997). Over the last 17 years, the coast of Newfoundland has been subject to some of the highest rates of chronic oil pollution in the world (chapter 2), and records of oiled birds on Newfoundland beaches date back to the late 1950s (Tuck 1961, Montevecchi and Tuck 1987). It is therefore important to determine the extent of seabird mortality associated with chronic spills in this region.

The purpose of this study was to use appropriately designed drift blocks to define an area of ocean adjacent to the coast of Newfoundland from which birds counted during beached bird surveys originate and to determine that proportion of birds that die within that area and reach the shore. Determining this area and its associated recovery rate is a crucial component of estimating seabird mortality from chronic oil pollution (chapter 7). It was also the purpose to determine drift rates (drift speed relative to the wind speed), as well factors that best predict onshore recovery rates.

6.2 METHODS

6.2.1 Drift block construction

Wooden drift-blocks were constructed according to the design of Wiese and Jones (2001). Blocks of 15.5 cm in length were cut from rough 9 x 9 cm fir and pine, and a 5 x 1.25 x 8.75 cm steel flatbar, weighing approximately 450 g, was screwed onto one side. Blocks were painted bright orange for maximum visibility, and labeled with a self-adhesive label commonly used on boats attached to a plastic tag. Tags were nailed to the blocks using galvanized roofing nails. Each label carried a unique identification number, a return mail address, email address and telephone number.

6.2.2 Drift block experiment

Four drift block experiments (trials) were carried out. During 21-22 February 2000, a total of 2047 blocks were dropped from offshore supply boats, Canadian Coast Guard vessels and a Canadian Coast Guard helicopter. Blocks were spread evenly over 10 drop locations, five off the east coast, and five of the south coast of Newfoundland, Canada (Fig. 6.1). Drop locations on the east coast were at 5, 10, 20, 40 and 80 nautical miles (nm) from the closest point of land, while drops along the south coast were all at 25 nm from shore. Locations were chosen to overlap major shipping routes and, in this instance, to determine a maximum distance from which birds may passively drift to shore. During 7-12 January 2001, 2500 blocks were dropped by Canadian Coast Guard vessels, spread evenly over 25 locations off the southeast and south coasts of Newfoundland (Fig. 6.2). Distances of drops in each of the five drop lines (S1-S5) were 1, 5, 10, 15 and 25 nm from the nearest land. Drops at 25 nm were replicates from the previous year.

Between 22 February and 17 March 2001, 23 of those 25 drops were repeated (the exception were the two outer drops of S1, Fig. 6.3). On 9 March 2002, 900 blocks were dropped in Placentia Bay, spread evenly over 9 locations (Fig. 6.4). Location A was a replicate of previous drops (S3-25 nm). The other 8 locations were divided into two lines

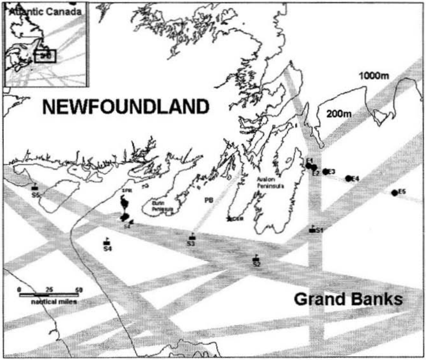


Fig. 6.1 Location of drift block drops and recoveries for 21-22 February 2000. Southern locations (S1-S5) are 25 nautical miles (nm) from the closest land, eastern locations (E1-E5) are at 5, 10, 20, 40 and 80 nm from shore. Recoveries on shore are indicated according to drop location. SPM = St. Pierre et Miquelon, PB = Placentia Bay, CSM = Cape St. Mary's. Grey bars indicate major shipping lanes.

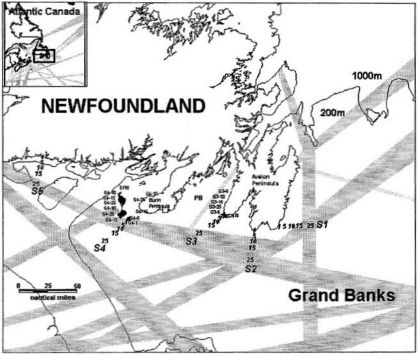


Fig. 6.2 Location of drift block drops and recoveries for 7-12 January 2001. Distance from closest land are indicated in nautical miles at drop locations. Recoveries on shore are indicated according to drop location and represent one or more blocks found. SPM = St. Pierre et Miquelon, PB = Placentia Bay, CSM = Cape St. Mary's. Grey bars indicate major shipping lanes

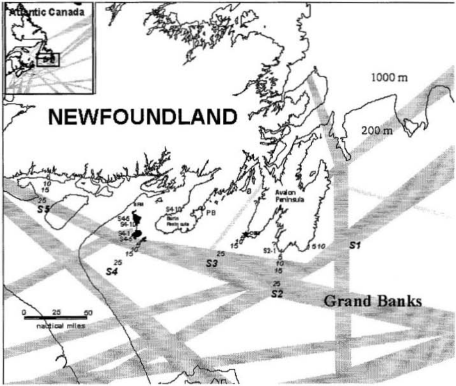


Fig. 6.3 Location of drift block drops and recoveries for 22 February – 17 March 2001. Distance from closest land are indicated in nautical miles at drop locations. Recoveries on shore are indicated according to drop location and represent one or more blocks found. SPM = St. Pierre et Miquelon, PB = Placentia Bay, CSM = Cape St. Mary's. Grey bars indicate major shipping lanes. Question marks indicate blocks of unknown origin due to tag loss.

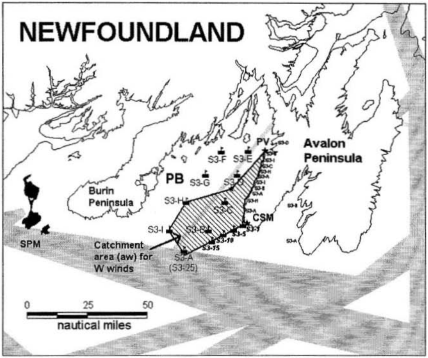


Fig. 6.4 Location of drift block drops (S3-A to S3-I) and recoveries for 9 March 2002. S3-A is the same location as S3-25 in other trials. Drop locations from previous trials (S3-15 to S3-1) are indicated. Distance from closest point of land for drops varies between 6 and 31 nm. Recoveries on shore are indicated according to drop location and represent one or more block found. Prevailing winds during this trial were from the West. Estimated catchment area (a_w; 2066 km^2) for blocks dropped in this area that are found between Cape St. Mary's (CSM) and Point Verde (PV) is indicated. SPM = St. Pierre et Miquelon, PB = Placentia Bay. Grey bars indicate major shipping lanes.

approximately parallel to shore, one line on either side of the main shipping lane, and between 6 and 31 nm from land.

All four drift block experiments were well publicized on local radio stations and in newspapers, as well as through local contacts of the Canadian Wildlife Service. A one dollar reward for every block reported onshore was offered as an incentive to increase public participation in the search for blocks, and were paid within two months of the drops. In addition, the Service d'Agriculture and interested individuals in St. Pierre et Miquelon (SPM), greatly assisted with searches for blocks along their shores. In areas on the Avalon Peninsula where it seemed most likely that blocks would wash ashore, targeted searches were carried for two weeks following releases. In some instances, weather made it impossible to survey beaches every day and public effort was certainly variable. Because we could not always assess for certain how much time had elapsed since blocks washed ashore, but most people were confident that blocks found had not been on the beach for more than 3 or 4 days, we considered blocks found within a 14-day period in the analysis, instead of only 10 days determined from buoyancy experiments (chapter 5). Hourly wind speed and direction for the different regions during the entire study period were made available by the Environment Canada Atlantic Climate Centre in Fredericton, New Brunswick. Cumulative wind vectors (the overall sum of wind vectors, which include wind direction and speed) were calculated for all 14-day periods following drops.

Differences among recovery rates (percent of blocks recovered that were dropped) among trials, drop lines and distances were analyzed with a General Linear Model where the response was treated as binary and using a logit link function (see below).

6.2.3 Predictive model

The relationship between onshore recovery rates *p(recovered)* and cumulative wind vectors over different time periods in drop lines S3 and S4 was examined using model selection procedures that employed Akaike's Information Criterion (AIC). Wind vectors for each trial in the S3 and S4 lines were calculated for 1 to 7, and 14 days after the drops. The influence of distance, wind vectors over different time periods, the combined effect of distance and wind vectors, and interactions between the wind vectors and distance (Eqn 1), on onshore recovery rates, was analyzed with a General Linear Models where the response was treated as binary and using a logit link function, where the full model was:

$$p(recovered)=\beta o+\beta_1\text{wind(days)}+\beta_2\text{distance}+\beta_3\text{distance*wind(days)} \qquad [1]$$

Log Likelihoods were extracted from each analysis, and the AIC (Eqn 2) was calculated as:

$$\text{AIC} = 2*np-2*(\text{Log Likelihood}) \qquad [2]$$

where *np* is the number of parameters, and the AIC is a compromise between model fit (log-likelihood) and the number of parameters. Models were ranked in ascending order of AIC scores and the most parsimonious model was selected as the model with the lowest AIC. In cases where AIC scores were within 2 points of each other, the model with the lowest number of parameters was considered the best (Burnham and Anderson 1998). The influence of ocean currents on recovery rates were included in this model because no data on surface currents exists for the south coast of Newfoundland to allow such an integration.

6.2.4 Drift rates

Average drift rates, defined as the speed of the block relative to the average wind speed, were determined for each block for which the drop origin was known, by measuring the linear distance each block traveled between the release at sea and recovery on the beach, and by determining the number of days elapsed between the two dates. From that, we estimated the average drift speed of the block, and by calculating the average wind speed for the same time period, we estimated the drift rate for each block recovered. Such calculations assume that these drift rates will be a representative average over the entire time period, recognizing that wind forcing on birds and to a degree on blocks is reduced as they get water-logged. All means are presented as ±1 SE, and all tests are 2-tailed.

6.3 RESULTS

6.3.1 Drift block recoveries

6.3.1.1 21-22 February 2000 trial

Winds at the time of the east coast drops (E1-5) were 15 km/h from the NW. Winds varied between W and NW, and between 6 and 33 km/h over the 14-day period following the drop (Fig. 6.5). Average wind speeds were 20 km/h with an overall wind vector from the NW. None of the 1013 blocks dropped were found during the 14-day period (East coast blocks Fig. 6.1). Winds at the time of the south coast drops were from the NW, and decreased from E to W (S1 to S5, Fig. 6.1) from 26 to 13 km/h. Wind conditions remained similar over the entire period for S1 and S2 blocks, creating an overall WNW wind vector with average speeds of 26 km/h (Fig. 6.5). In the S3 region, winds varied little during the first week, but made a full circle during the second week (Fig. 6.5). This created an overall WNW wind vector with average wind speeds of 25 km/h. Blocks in region S4 were subjected to periods of SW and NW winds during the first week, but switched to strong NE and NW for the second week after the drop (Fig. 6.5). Overall, the wind vector was from the NNW and averaged 10 km/h. Finally, wind directions in the S5 area were comparable to S4, but stronger (Fig. 6.5). Overall, the wind vector was from the N and averaged 14 km/h.

Only 0.29% (3) of 1034 blocks dropped on the south and southeast coast (S1-S5) between 21-22 February were found during the 14-day period following the drop (on days 6, 9 and

Figure 6.5. Wind vectors showing speed and direction during 14-day period following the block drops off the east and south coast on 21-22 February 2000.

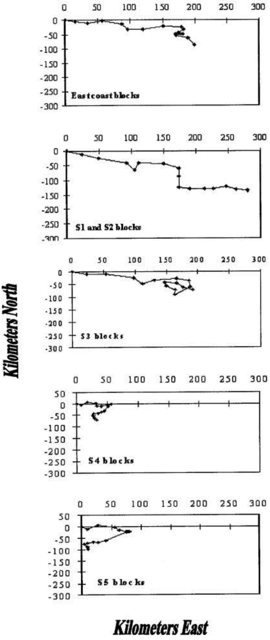

Eastcoast blocks
S1 and S2 blocks
S3 blocks
S4 blocks
S5 blocks
0
50
100
150
200
250
300
-50
-100
-150
-200
-250
-300
Kilometers North
Kilometers East

Table 6.1. Summary of recoveries from drift blocks dropped off the south coast of Newfoundland during 7-12 January 2001 and 22 February – 17 March 2001. 100 blocks were dropped at each location on each date. Numbers represent the numbers of blocks found onshore within 14 days of each drop.

Distance (nm)	S1	S2	S3	S4	S5	Total #	Total %
	07-Jan-01	07-Jan-01	08-Jan-01	12-Jan-01	08-Jan-01		
25	0	0	2	62	0	64	12.8
15	0	0	14	48	0	62	12.4
10	0	0	2	90	0	92	18.4
5	0	0	23	94	0	117	23.4
1	0	0	0	96	0	96	19.2
Total	0	0	41	390	0	*436**	
Total %	0	0	8.2	78.0	0		17.4*
	22-Feb-01	27-Feb-01	02-Mar-01	17-Mar-01	27-Feb-01		
25		0	0	0	0	0	0.0
15		0	0	0	0	0	0.0
10	0	0	0	2	0	2	0.4
5	0	0	0	12	0	12	2.4
1	0	12	0	1	0	13	2.6
Total	0	12	0	15	0	*37***	
Total %	0	2.4	0	3.0	0		1.6**
			9 Mar-02				
A			8			8	
B			12			12	
C			31			31	
D			0			0	
E			0			0	
F			0			0	
G			0			0	
H			30			30	
I			29			29	
Total			110			110	
Total %			12.2				
Total #	**0**	**12**	**151**	**408**	**0**		**586**
Total %	**0**	**1.0**	**7.5**	**33.8**	**0**		**8.8 +/- 0.4**

* includes 5 blocks for which the origin is unknown due to tag loss

** includes 10 blocks for which the origin is unknown due to tag loss

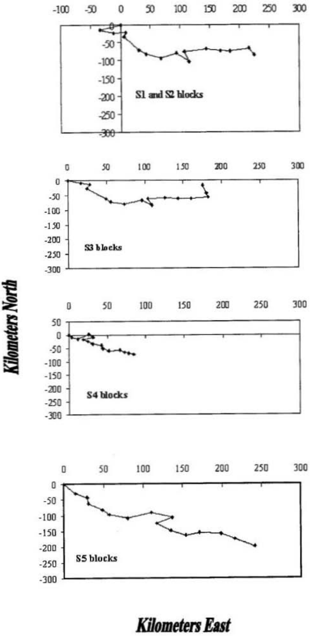
12), all of them originating from the S4 drop and recovered in St. Pierre et Miquelon (SPM, Fig. 6.1).

6.3.1.2 7-12 January 2001 trial

Winds at the time of block releases varied in different locations and on different days. For S1 and S2 blocks, all dropped on 7 January, winds were strong from the NE on the first day and then switched to WNW for the remainder of the 14-day period. The only exception was a brief period of SE winds on day 9 (Fig. 6.6). Overall, the wind vector over the 14 day period after the release was WNW, averaging 26 km/h. No blocks from these two drop lines were recovered (Fig. 6.2, Table 6.1).

Winds for S3 blocks, dropped Jan 8, were 6 km/h from the S for approximately 5 hours after the drop, and then varied in a similar fashion to the winds for S1 and S2 blocks, although two days of S winds were recorded for days 13 and 14 (Fig. 6.6). Overall, winds averaged 20 km/h with an overall WNW vector. 8.2 % (41) of the blocks dropped along this line were found, all in Newfoundland, with the majority from 5 and 15 nm offshore (Fig. 6.2, Table 6.1). 87 % of these blocks were recovered within 6 days of the drop. Winds in the S4 area, where blocks were dropped on 12 January, were light from the NW, and with the exception of strong SW winds on January 14, and strong NE winds on January 15, remained that way over the 14-day period (Fig. 6.6). Winds speeds averaged 11 km/h, and created an overall NW vector. 78 % (390) of blocks from this line were recovered, with 88 % of these in SPM (Fig. 6.2, Table 6.1). Recovery rates showed a

Figure 6.6. Wind vectors showing speed and direction during 14-day period following the block drops off the south coast on 7-12 February 2001.


-100
-50
0
50
100
150
200
250
300
-50
-100
-150
-200
-250
-300
S1 and S2 blocks
0
50
100
150
200
250
300
0
-50
-100
-150
-200
-250
-300
S3 blocks
0
50
100
150
200
250
300
50
0
-50
-100
-150
-200
-250
-300
S4 blocks
0
50
100
150
200
250
300
0
-50
-100
-150
-200
-250
-300
S5 blocks
Kilometers North
Kilometers East

decreasing trend with increasing distance from shore (χ^2_1=66.1, P<0.0001, Table 6.1). All blocks, except one from the S4 line found in Newfoundland originated from 15 nm. 99 % of blocks found from that line were recovered within 3 days.

Blocks from the S5 line, dropped 8 January, were exposed to almost steady NW winds through the entire 14-day period following the drop (Fig. 6.6). Wind speeds averaged 26 km/h. No blocks from this drop line were found (Fig. 6.2, Table 6.1).
Five additional blocks were found on SPM during this period, but their origin could not be determined due to tag loss (Table 6.1).

Overall, 17.4 % of blocks released between 7-12 January were recovered (Table 6.1). Recovery rates differed significantly among drop lines (χ^2_4=1487.82, P<0.0001) and distances offshore (χ^2_4=30.39, P<0.0001), and increased with decreasing distance from shore (χ^2_1=20.08, P<0.0001). However, recoveries from the closest drops (1 nm) were slightly fewer than those from 5 nm. There was no relationship between distance and recovery rate for the S3 drop line (χ^2_4=1.62, P>0.1).

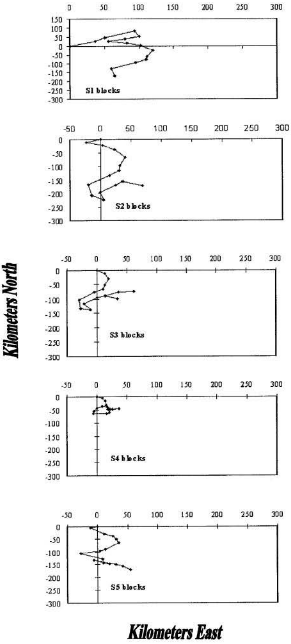
6.3.1.3 22 February – 17 March 2001 trial

Winds during this trial were variable. For S1 blocks, dropped on 22 February, winds were SW for the first 4 days, then from the opposite direction for 2 days. At the end of the first week winds were NW, and then NE by the end of the 14-day period. The overall wind vector was NNW averaging 30 km/h (Fig. 6.7). No blocks were recovered from this line (Fig. 6.3, Table 6.1).

For S2 blocks, dropped 27 February, winds were ENE at the onset and then varied between NE and NW until day 11, after which they switched to S and SE, and to NW at the end of the 14-day period. Overall winds averaged 28 km/h and had a NNW (Fig. 6.7) vector. 2.4 % (12) of blocks dropped on the S2 lane were found (Table 6.1). All were from 1 nm offshore, and all were recovered in Newfoundland on days 12 and 13 after the drop.

Wind conditions for S3 blocks, dropped 2 March, were quite similar to the S2 area, although they were NW at the onset. Winds averaged 23 km/h with an overall vector from the NW (Fig. 7). No blocks were found from this line (Fig. 6.3, Table 6.1).S4 blocks, dropped 17 March, were exposed to 20 km/h winds from the SW for several hours on that day, but they quickly turned N and NW for the rest of the first week (Fig. 6.7). Winds made a complete circle thereafter, and ended from the E on March 31. This pattern created an overall NNW vector and averaged 12 km/h. 3 % (15) of S4 blocks were found, all from within 10 nm of the coast, and all at the end of the 14 days. All blocks but one were recovered in SPM (Fig. 6.3, Table 6.1).

Figure 6.7. Wind vectors showing speed and direction during 14-day period following the block drops off the south coast on 22 February to 17 March 2001.


0
50
100
150
200
250
300
150
100
50
0
-50
-100
-150
-200
-250
-300
S1 blocks
-50
S2 blocks
S3 blocks
S4 blocks
S5 blocks
Kilometers North
Kilometers East

Finally, winds for S5 blocks, dropped 27 February, were similar to those for S2 blocks, except that winds remained almost entirely from the NW after day 11. No S5 blocks were recovered (Fig. 6.3, Table 6.1).

Ten additional blocks were found on the Avalon Peninsula during this period, but their origin could not be determined due to tag loss. It is likely that the majority originated from S2 line based on their locations, and perhaps two from the S3 or S4 line.

Overall, only 1.61 % (37) of blocks dropped between 22 February and 17 March 2001 were recovered, and recovery rates decreased with increasing distance from shore (χ^2_1=28.66, P<0.0001). Recovery rates also differed among lines (χ^2_4=45.74, P<0.0001) and distances from the shore (χ^2_4=33.85, P<0.0001) overall, but not between S2 and S4 (χ^2_1=0.34, P>0.5), the two drop lines that had recoveries.

6.3.1.4 9 March 2002 trial in Placentia Bay

Winds in Placentia Bay at the time of drops were 13 km/h and SW. During the first 7 and the last 2 days, winds were predominantly W and SW (Fig. 6.8), but prevailed from the N for 4-day period in between. Overall, winds averaged 33 km/h with a total wind vector from the NNW.

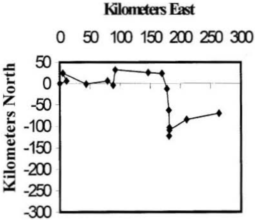


Fig. 6.8 Wind vectors showing speed and direction during 14-day period following the block drops in Placentia Bay on 9 March 2002.

12.22 ± 1.10 % (110) of blocks were recovered, 75% of them within 8 days of the drop (Table 6.1). There were no recoveries from locations D, E, F, and G, which were all I he northern part of Placentia Bay. Recovery rates among the other locations varied and no trend in recovery rates with distance from shore was detected. All blocks but three were found on the west coast of the Avalon Peninsula (Fig. 6.4).

6.3.1.5 All south and southeast drops combined

Recovery rate from all four trials along the south and southeast coast combined was 8.84 +/- 0.35 %. Rates increased significantly with decreasing distance from shore (χ^2_1=41.2, P<0.0001) overall, as well as for S3 (χ^2_1=133.0, P<0.005) and S4 (χ^2_1=122.1, P<0.0001) alone. Substantial differences were also found among lines (χ^2_4=1144.6, P<0.0001) and dates (χ^2_3=607.89, P<0.0001) overall, as well between S3 and S4 (χ^2_1>20.0, P<0.0001) alone.

6.3.2 Predictive model

S3 and S4 blocks were used to build a predictive model because ample data was only available from those two lines. The cumulative wind vector over the first 3 days after block release, combined with the distance from shore, were the best predictive models to estimate onshore recoveries of S3 and S4 blocks (Table 6.2, 6.3). To determine wind

Table 6.2. Model fitting results for how well wind direction vectors and distance of drop locations from shore explain recovery rates of blocks from S3 drops. Only 10 best models are shown. The number after wind refers to the number of days included after the drop. Stars (*) indicate interactions, pluses (+) indicate combined effects.

Model	ΔAICc	Deviance[1]	# parameters
*Wind3*distance*	0.00	980.46	4
Wind3	14.18	998.64	2
Wind3+distance	14.18	996.64	3
Wind14+distance	15.36	997.82	3
Wind14	16.88	1001.34	2
*Wind14*distance*	17.36	997.82	4
*Wind1*distance*	25.80	1006.26	4
Wind2	47.56	1032.02	2
Wind4	47.56	1032.02	2
Wind7	47.56	1032.02	2

[1] DEV=-2*(log-likelihood)

Table 6.3 Model fitting results for how well wind direction vectors and distance of drop locations from shore explained recovery rates for blocks from S4 drops. Only 10 best models are shown. The number after wind refers to the number of days included after the drop. Stars (*) indicate interactions, pluses (+) indicate combined effects.

Model	ΔAICc	Deviance[1]	# parameters
Wind3+distance	0.00	742.04	3
Wind2+distance	0.00	742.04	3
*Wind3*distance*	0.10	740.14	4
Wind2+distance*	0.10	740.14	4
*Wind4*distance*	46.66	786.70	4
*Wind7*distance*	46.66	786.70	4
*Wind6*distance*	46.66	786.70	4
*Wind5*distance*	46.66	786.70	4
Wind3	360.34	1104.38	2
Wind2	360.34	1104.38	2

[1] DEV=-2*(log-likelihood)

specific recovery rates and associated areas of ocean from which blocks originate (catchment areas, a_w), I investigated S3 drops in more detail, because all blocks found on the Avalon Peninsula, where systematic beached bird surveys are carried out (chapter 2), originated from these drops. As models with 3-day winds were the best predictive models to explain recovery rates (Table 6.2) I calculated 3-day winds for each S3 drop and related them to observed recovery rates under those conditions. Recovery rates increased as 3-day winds switched from N to NW to W (Table 6.4). No other wind vectors during the three days following each trial were observed.

6.3.3 Defining catchment areas

I derived catchment areas (a_w) by determining the origin of blocks found during each of the calculated 3-day wind directions, and for W winds, assumed that drop locations were the along outer boundary of those areas (Table 6.4, Fig. 6.4). NW winds were only observed during trials carried out along the S3 line on February 2000 and January 2001 (Fig. 6.1, 6.2). Combined, 7.8 % of blocks were found, 95 % of these from within 15 nm of the coast. I assumed this distance to be the outer limit, with a recovery rate of 0.058 ± 0.009 (Table 6.4). A catchment area, however, could not empirically be defined as only recoveries from that line existed. Therefore, three scenarios based on the knowledge gained from the 9 March 2002 trial were considered (Fig. 6.9): 1) a small area from the 15 nm drop straight to Cape St. Mary's and to Point Verde (432 km^2), 2) the distance from which recoveries occur were the same along the coast and the northern and southern boundary have a parallel angle towards shore (1100 km^2), and 3) the catchment area is

Table 6.4. Overall 3-day wind vectors following S3 drops, related recovery rates and catchment areas (a_w). See results and Fig. 6.4 and 6.9 for details. Three scenarios are presented for NW winds.

3-day wind vector	Recovery rate	Catchment area (a_w) km^2
N	0	0
NW	0.058 +/- 0.009	432
	0.058 +/- 0.009	1100
	0.058 +/- 0.009	2200
W	0.122 +/- 0.011	2066

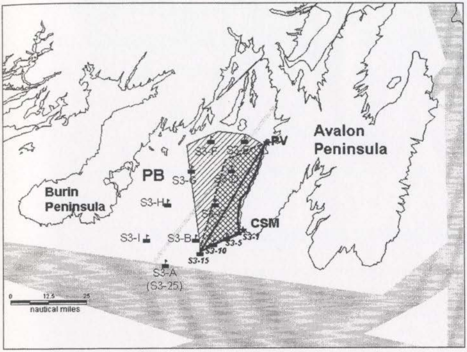


Fig. 6.9 Three scenarios for catchment areas (a_w; 432, 1100, 2200 km^2) during NW winds. Locations of drops carried out in the area shown.

enlarged towards the NW (2200 km^2), similar to the catchment area for west winds which is enlarged to the W (Fig. 6.4). Recovery rates for all three proposed NW wind catchment areas were held constant because recovery rates could not be directly inferred from changes in the area or distance from shore.

6.3.4. Drift rates

Only blocks of known origin and recovery time (within 3 days) were used. Based on observed wind speeds and the timing of block recoveries, blocks drifted at an average speed of 4.2 ± 0.9 % of the wind speed, but this rate ranged between 0.001 and 30.9 % (Table 6.5). Average drift rates differed among wind directions and were highest during NW winds.

6.4 DISCUSSION

6.4.1 Physical factors affecting drift

The likelihood of dead birds floating at sea washing ashore is dependent on the speed and direction of winds, currents and tides. How these interact and influence the velocity and direction of drifting bodies can be very complex, and is poorly understood for the south coast of Newfoundland. The region is located at the southern end of the Labrador Current, which carries cold, low salinity water southward, and at the northern boundary of the Gulf Stream, which carries warm, high salinity water northward (Fig. 6.10). The

Table 6.5 Estimated average drift rates (% of average wind speed) per drop of blocks recovered onshore within 14 days of the drop, and for which the drop origin was known. Drop dates, number of blocks recovered (n), mean number of days afloat, mean distance traveled (km) assuming linear travel, average winds and block (km/h) speeds during drift period are indicated.

Drop date	# blocks	Mean days afloat	Mean distance traveled (km)	Average wind speed (km/h)	Total wind vector	Average block speed (km/h)	Drift rate (% wind speed)
8 Jan 01	41	4.7 ± 7.7	42.8 ± 7.3	18.7 ± 6.1	NW	0.76 ± 1.36	4.3 ±3.2
12 Jan 01	388	2.5 ± 0.8	30.2 ± 2.3	10.4 ± 1.6	NW	0.61 ± 0.40	5.1 ± 1.2
27 Feb 01	12	12.8 ± 0.5	1.9 ± 0.0	27.7 ± 0.3	N	0.006 ± 0.0002	0.02 ± 0.0005
17 Mar 01	16	11.9 ± 3.0	44.7 ± 12.9	11.3 ± 0.09	N	0.21 ± 0.24	1.8 ± 2.2
9 Mar 02	100	6.6 ± 2.7	60.9 ± 15.1	29.7 ± 2.1	W	0.42 ± 0.11	1.44 ± 0.43
TOTAL	**557**						**4.2 ± 0.9**

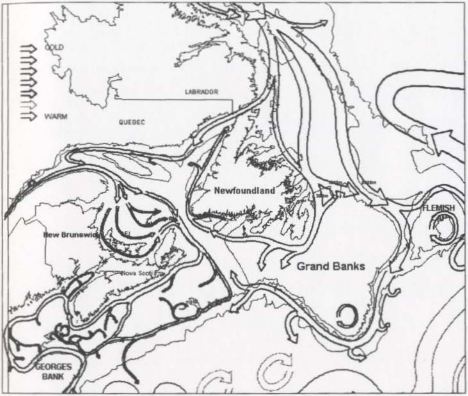


Fig. 6.10. Map of Atlantic Canada showing the major current patterns (Lock et al. 1994).

Labrador Current turns southwest over the Grand Banks, and a branch current flows west along the south coast of the island through the Cabot Strait into the Gulf of St. Lawrence. A weak coastal counter current flows east along the south coast of Newfoundland (McManus and Wood 1991, Greenberg and Petrie 1988). Residual surface current patterns in the area are not well known. Counter-clock wise primary tidal currents have been documented in Placentia Bay, and a strong tidal current flows back and forth through the strait between the islands of St. Pierre and Miquelon. Apart from these observations, however, no specific data on surface current patterns along the south coast of Newfoundland exists and currently presents a major gap needed to fully understand the interactions of winds and currents on drift patterns. Predominant strong winds between October and March come from the west and secondarily from the northwest (MacLaren Plansearch 1991). The combined effect of wind induced surface currents with the predominant large ocean current patterns would carry carcasses either parallel to, or away from, the south shore, although there is local knowledge of an onshore current that set towards Cape Race (Tuck 1961). On the other hand, blocks drifted in almost opposite directions of the wind during periods low wind speeds, suggesting a predominant influence of ocean currents over the wind. Clearly, quantitative data on surface currents, and deflection patterns due to the effects of the Coriolis force in the area are needed. But as Burger (1993b) found off the west coast of Canada, the low overall recovery rates of 8.8 % indicates that even birds that die within 1-5 nm of the shore may not be deposited on beaches, unless there are strong localized tidal currents and/or periods of strong onshore winds inducing onshore surface currents.

6.4.2 Proportion of birds lost at sea

As an apparent reflection of the predominant wind and current patterns, only 0.4 % of blocks dropped off the east and southeast coast (east of Cape Race, i.e. E drops and S1, S2 drops) were recovered, suggesting that very few birds that die at sea in this area east off Newfoundland would drift ashore. Most birds found along these stretches of coastline during winter probably died very close to shore, drifted in during occasional east winds, or arrived alive by swimming or flying. Observations from systematic beached bird surveys and detailed deposition studies on the Avalon Peninsula (chapter 2 and 3) indicate that between 0-15% of oiled birds found on beaches may arrive alive. Similarly, 15 % of oiled birds found on beaches during large oil spills appears to arrive alive (G. Ford, pers. comm.; see chapter 7). Like birds off the east coast, birds that die west of St. Pierre et Miquelon (S5) have little chance of ever reaching shore or being counted, as none of the 1200 blocks dropped in this region were reported within 14 days of the drops. The coastline in this area is mostly uninhabited and only accessible by boat. Although no recoveries were expected, it is possible that blocks washed up on isolated beaches given the wind conditions measured during trials. Regardless, it appears that most seabird mortality due to oil in these areas would go unnoticed.

The likelihood that seabirds which die at sea between Cape St. Mary's and St. Pierre et Miquelon (S3 and S4, Fig. 3) due to spilled or discharged oil will be recovered, is higher than for other regions, and can be quite substantial for birds that perish close to St. Pierre et Miquelon. As in other studies (Hlady and Burger 1993, Flint and Fowler 1998), I believe that an additive interaction of wind-driven and tidal-currents drives blocks ashore.

When winds favourable to onshore deposition apparently interacted with tides to create onshore surface currents, blocks were generally driven ashore within a few days (Table 6.5), such as during the S4 drop in January 2001.

The rate of reported block recoveries indicate the maximum proportion of seabird carcasses floating at sea likely to be recovered, because blocks were painted bright orange, do not get scavenged, and survey effort was more intensive than during weekly systematic beached bird surveys. On the other hand, the total number of birds that arrive on shore may differ from these rates, as birds actively swimming or flying to land are not considered here.

Differences in recovery rates from drops at different distances from the coast were comparable to previous findings (Hlady and Burger 1993). Brief periods of south or southeast wind induced surface currents at the onset were important in keeping blocks in the vicinity of shore or driving them close enough to shore for tidal currents to take effect and deposit them on beaches. Generally, it appears that if these conditions are not present soon after a seabird dies at sea, it will not be recovered. However, in a few instances, blocks were found after more than 10 days. Presumably, several days of winds parallel to the coast kept them close enough so that a few hours of onshore winds carried them ashore or at least in reach of local tidal currents that ultimately deposited them on a beach.

6.4.3 Drift rates

Reported average rates of seabird carcass drifts range between 2.2 % and 4.6 % of the wind speed measured (Hope Jones et al. 1970, Bibby and Lloyd 1977, Bibby 1981), although it may be as high as 12 % (Bibby 1981). Although drift blocks of blocks could be expected to be higher than that of birds due to water-logging and a reduction of wind-forcing over time on birds, average drift rates of block movement in this experiment of 4.2 ± 0.9 % are comparable to these earlier findings and confirm that the blocks used accurately mimic seabird carcass drift (Wiese and Jones 2001), although drift rates up to 30 % were estimated in a few instances. Nevertheless, these average drift rates are considered a minimum, due to lag times of several hours to several days between the arrival of blocks and their recovery on shore, and because blocks most likely did not travel in a straight line between drop and recovery points.

6.4.4 Predictive model of recovery rates

Blocks were found on all beach types representative of the area: open and enclosed rocky beaches on the Avalon Peninsula, and on the open sandy beaches typical for the western Burin Peninsula and St. Pierre et Miquelon. Beach accessibility and exposure to onshore winds likely explains why recovery rates in St. Pierre et Miquelon were substantially higher than in Placentia Bay and elsewhere. Recoveries on the west coast of the southern Avalon Peninsula were observed on most beaches, including ones that are, and are not, regularly visited during systematic beached bird surveys.

Oiled seabirds have washed ashore for decades in southeastern Newfoundland (Tuck 1961), and have often been the only indication that oil had been discharged at sea (Montevecchi and Tuck 1987). Systematic beached bird surveys have been conducted on selected beaches on the Avalon Peninsula since 1984, and results have shown some of the highest oiling rates in the world (chapter 2). This experiment indicates that most birds found on the beaches of southeast Newfoundland originate only from a small adjacent marine area, and that this catchment area and the proportion of onshore recoveries of birds that die within it, fluctuates with different winds.

Wind vectors over the first three days after block release were the best predictors for the likelihood of onshore recoveries of S3 blocks, the majority of which occurred on the west coast of the southern Avalon Peninsula, which is also where periodic beached bird surveys are carried out. Three different 3-day wind directions were observed after S3 drops: north, northwest and west. North winds occurred only once, during drops in March 2001, when no blocks were recovered. Based on the predominantly west-facing orientation of beaches in eastern Placentia Bay, it may also be assumed that no birds on the southern edge or outside of Placentia Bay would wash ashore under prevailing winds from NE, E, or SE. It is possible, however, that some birds that die far inside the bay may be recovered even under N winds, as they are driven towards the open sea parallel to the coast. However, we do not have any information to support or refute such an assumption.

Northwest winds occurred twice, in February 2000, and January 2001. During these periods, where 5.8 ± 0.9 % of blocks were recovered, 95 % of them within 15 nm from shore. It appeared that 15 nm may be close to the outer boundary for the majority of birds that die at the edge of the bay and are recovered. It is unclear how many birds that die inside the bay end up on the shore, and from which areas they might come, and hence the definition of a catchment area under NW winds is difficult. In the absence of more information, we assumed three possible scenarios for a catchment area, that range in size and location from 432 to 2200 km^2 (Fig. 6.9). Due to the interaction of winds and distance, it is difficult to determine how recovery rates may fluctuate if a catchment area that reaches further than 15 nm from shore is assumed. Until more data from drift block experiments can be collected under these wind conditions, it seems reasonable to use the mid-sized catchment area (Table 6.4, see chapter 7).

Three-day winds from the west occurred during drops on 9 March 2002, in which a grid of drops was laid out to determine such a catchment area more carefully. The catchment area for west winds was estimated at 2066 km^2 (Fig. 4), although this is most likely a minimum. In the absence of drops further away from the coast, we assumed drop locations to be along the outer boundary of this area. It is thus possible that the catchment area is larger, but no data exists to determine what the recovery rate in such an area may be. We thus assume a catchment area during west winds of 2066 km^2 and a recovery rate of 12.2 ± 1.1 %. Similarly, we may assume that prevailing winds from the SW and S have similar catchment areas and recovery rates, although they may be shifted towards the south and have a larger catchment area inside than outside the bay. Until more data can be collected, there is no evidence to support such an assumption other what can be inferred from recoveries during known winds and beach orientations.

6.5 CONCLUSIONS

Wind direction and speeds have a strong influence on seabird carcass drift on the open ocean as they push the bird directly or indirectly (through induced surface currents) through the water, while tidal currents most likely have a substantial additional influence deposition on local beaches. Ocean surface currents may become a predominant factor influencing seabird carcass drift patterns during periods of mild winds and it appears imperative the gap of knowledge regarding surface currents in the area be addressed. Only a small proportion of birds that die at sea reach the shore. The area from where birds originate and subsequent recovery rates is highly dependent on wind conditions present during the first three days after death and proximity to land.

Birds drifted at net speeds of about 4 % of the wind speed, although they may drift as fast as 30 % and cover large distances in a few days. However, to accurately assess these speeds and to investigate what the relative contributions of wind driven currents, ocean currents and tidal currents are to these drift rates, future experiments should be carried out using similar drift blocks equipped with satellite transmitters. Detailed seabird carcass trajectory models could then be constructed to forecast and hindcast seabird trajectories and aid in seabird mortality estimates, spill response and enforcement.

It is significant that these experiments demonstrate the regional importance of St. Pierre et Miquelon in monitoring chronic oil pollution. Although some blocks from the S4 line were found on the Burin Peninsula, most recoveries were from these two small islands.

Clearly the open exposure of beaches on these islands, their relatively high population density, and their proximity to shipping lanes, make St. Pierre et Miquelon an ideal monitoring location for chronic oil pollution, and possibly better than the south coast of insular Newfoundland.

These experiments confirm that the great majority of seabird mortality due to oil at sea is never detected. In addition to oil discharges from ships, threats to seabirds also exist around offshore oil platforms (Wiese et al 2001). Because regional oil exploration and production sites are all east of Newfoundland, seabird mortality associated with these activities can not be assessed through beached bird surveys. It is therefore essential that beached surveys be combined with surveys by dedicated, trained, independent observers on platforms to ensure that all impacts of oil and other sources of industry-related seabird mortality are fully and accurately assessed.

The information collected in this study illustrates the value of using beached bird survey results to assess total seabird mortality due to chronic oil pollution, and the importance of integrating wind-specific recovery rates and catchment areas into such a mortality model. Combining this information with processes that determine the fate of stranded birds on beaches should provide more accurate estimates of current seabird mortality due to chronic oil pollution in Atlantic Canada.

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CHAPTER 7 – OILED SEABIRD MORTALITY MODEL: ASSESSING IMPACTS OF CHRONIC, ILLEGAL DISCHARGES OF OIL AT SEA ON SEABIRDS IN EASTERN CANADA

7.1. INTRODUCTION

Oil found on beached seabirds has been identified as heavy fuel oil, the type typically found in bilges of large ocean going vessels, such as tankers, cargo and container ships (A. R. Lock, personal communication, Averbeck et al. 1992). Regulations set by the International Convention for the Protection of Pollution from Ships (MARPOL), which came into force in Canada in 1993, state that oil cannot be discharged into the marine environment at more than 15 ppm, a concentration at which no oil sheen is visible on the water surface. But oiled birds have washed up for decades in many regions of the world where heavy ship traffic overlaps with dense seabird concentrations (Chardine 1990, Camphuysen and Heubeck 2001, chapter 2). Despite the obvious impact, it is extremely difficult to accurately assess seabird mortality, even when oil spills are large and well documented (Ford et al. 1987, Page et al. 1990, Piatt et al. 1990, Burger 1993a, Van Pelt and Piatt 1995, Fowler and Flint 1997). Evidence suggests that effects from numerous chronic oil spills may be more important to long-term seabird population stability than occasional large spills (Hunt 1987, Burger 1992, Nur et al. 1997). An accurate estimate of seabird mortality is needed, especially in areas where chronic oil pollution levels are high, and critical for effective seabird conservation and management.

Oiled beached birds are often the only indications that oil has been illegally spilled into the marine environment, because they are the most visible victims of chronic marine oil spills (Furness and Camphuysen 1997). The internationally accepted index for the severity of chronic marine oil pollution is the proportion of birds found which are oiled (Camphuysen 1989, Camphuysen and Franeker 1992, Camphuysen and Heubeck 2001, chapter 2). This index can be directly derived from the number of birds counted on systematic beached bird surveys and compared among species and regions. The determination of the actual number of birds that die at sea due to chronically spilled oil, however, is a separate and more complex issue, and has much in common with estimating seabird mortality resulting from large catastrophic spills.

Only a small proportion of birds that die at sea wash up on shore, because most birds sink, are scavenged or drift away from shore (Ford et al 1987, Page et al 1990, Hlady and Burger 1993b, chapters 3,5). Of those that arrive on a beach, many are not tallied during regular beached bird surveys, because they are not detected (Fowler and Flint 1997, chapter 3), are eaten by scavengers, or are covered by the beach substrate through wave action (Ford et al. 1987, Page et al. 1990, Hlady and Burger 1993, chapter 3). Detailed and geographically-specific, information is needed on these parameters to accurately estimate seabird mortality due to chronic oil pollution in a given area.

The southeast coast of Newfoundland, Canada (Fig. 7.1) has been subject to some of the highest rates of chronic oil pollution in the world (chapter 2), and records of oiled birds in this region date back to the late 1950s (Tuck 1961). It has been suggested that 20,000-

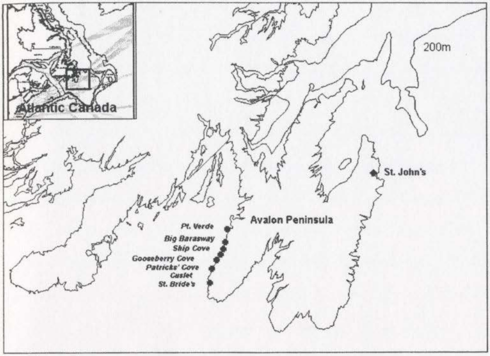

Fig. 7.1. Study area on the southeast coast of Newfoundland, Canada. Location of periodic beached bird surveys conducted on the Cape Shore, and major shipping routes are shown (grey bars in inset).

500,000 seabirds die annually in Newfoundland due to this anthropogenic impact (Tuck 1961, Piatt et al. 1985). However, little information was available on many of the necessary parameters to create more accurate estimates. In addition, calculations used to arrive at the estimates included extrapolating the density of beached carcasses/km, determined from monthly beached bird surveys, to the entire coastline around the island of Newfoundland considered suitable for deposition to occur. It was also assumed that 10-30 % of oiled birds that die at sea arrive on shore and were counted on the monthly surveys. As they point out, several very large uncertainties existed with these estimates, and recent results indicate that there are additional and critical aspects, including beach survey interval (chapter 3), drift block design (chapter 4), carcass sinking rates (chapter 5), and the lack of relationship between at-sea seabird mortality and the length of coastline considered suitable for bird deposition to occur, that need to be considered.

As a result of the remaining uncertainties in these past estimates, much discussion and confusion remained on the extent of the mortality due to chronic oil pollution in Newfoundland. To accomplish changes in the interpretation and implementation of regional, national and international policies, conventions and laws, a more reliable estimate than 20,000 – 500,000 seabirds killed was needed. In light of new research findings, the purpose of this study was to develop a general mathematical model to estimate seabird mortality due to chronic oil pollution by ships along a given stretch of coastline, and to apply this model to an area within Atlantic Canada where chronic oil pollution is very high (chapter 2), and where the necessary parameters for such an estimate have been determined.

7.2 METHODS

7.2.1 Oiled Seabird Mortality Model (OSMM)

The process of estimating seabird mortality at sea due to chronic oil pollution by ships in a given area can be divided into two components, and is the reverse of the natural process that occurs after seabirds encounter oil on the water (Fig. 7.2). During the onshore component, the number of birds found on beaches is extrapolated through a sequence of four steps to estimate the number of oiled birds that arrived along a defined stretch of coastline during the survey period (N_o). During the at-sea component, N_o is extrapolated out to sea in a sequence of four steps to determine total seabird mortality per survey period due to spilled oil in a defined area of water (M_o).

7.2.1.1 Onshore component

First, the number of birds found on a beach during a regular beached bird survey (N_{obs}) is corrected for the probability that birds are detected on the beach (d). This is dependent on the beach substrate and the observer, and data should be segregated by beach type and observer accuracy if more than one person performs the surveys. Second, this number is corrected using a persistence coefficient (s) to account for birds that arrived on a beach following the last survey, but disappeared in the interim period due to scavenging or substrate turnover. Together, this will result in an estimate of the actual number of birds that arrived on a particular beach between surveys (N_{bs}). Third, the total number of oiled birds that arrived on surveyed beaches during each survey period (e.g. winter months,

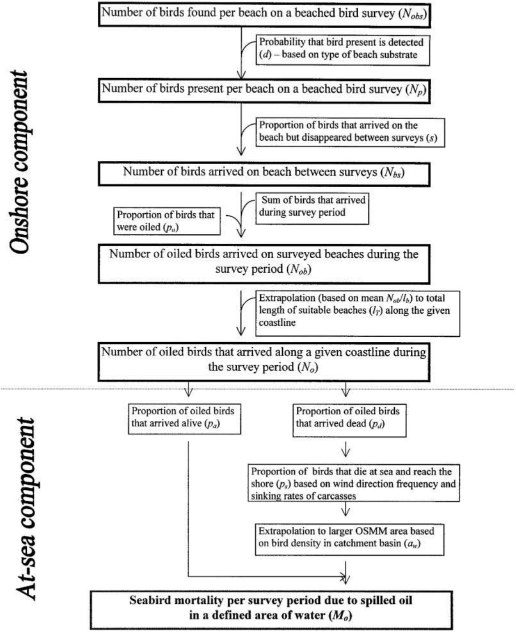

Fig. 7.2. Schematic diagram of the Oiled Seabird Mortality Model

N_{ob}), is calculated by summing N_{bs} for all beaches over the entire survey period and multiplying it by the overall proportion of oiled birds found (p_o) during the survey period. Finally, an estimate of the total number of oiled birds that arrived along the portion of a given coastline where deposition can occur (l_T) during that survey period (N_o) is calculated by extrapolating the number of oiled birds that arrived on a particular beach between surveys (N_{ob}) to include any beaches not surveyed between the limits of the coastal survey area (e.g. due to inaccessibility), based on the mean number of oiled birds found per kilometer of beach surveyed (l_b), and summing each of these estimates over the entire survey period. This step is critical for heterogeneous coastlines, where not all beaches are surveyed, and where stretches of coastline exist where deposition cannot occur (e.g. cliffs). The assumption for this extrapolation is that all parts along the coastline considered suitable for deposition have equal probabilities to collect birds. In areas where this assumption is not met, this extrapolation should be modified accordingly.

The number of oiled birds that arrive along a given coastline during a survey period (N_o) is calculated as:

$$N_o = \left[\sum_t \left[\sum_{beaches} \frac{N_{obs}}{d*s} \right] \right] * p_o * \frac{l_b}{l_T} \qquad [1]$$

where:

t = total survey period

beaches = beaches surveyed

N_{obs} = number of birds found per beach on a given survey

d = probability that a bird present is detected

s = persistence coefficient

p_o = proportion of birds found that are oiled

l_b = length of surveyed beaches

l_T = total length of coastline between survey limits where deposition can occur

7.2.1.2 At-sea component

The proportions of birds that swim or fly to shore and die on the beach (p_a), or die at sea and passively drift (p_d) must be considered. The former group is included in beach surveys, and is an actual count that does not have to be further corrected. The latter, represents only the fraction of birds that die at sea and reach the shore, and must be extrapolated to a defined area of water. In all likelihood this extrapolation is wind dependent, because the proportion of birds that die at sea and reach the shore (p_s), as well as the area from which birds found on the beach originate (a_w), is variable and directly dependent on wind conditions immediately after their death (chapter 6). Finally, if desirable, this estimate can be extrapolated to a larger area of ocean, based on knowledge of seabird distributions, ship traffic densities and known chronic oil spill locations. When combined, these factors will result in an estimate of seabird mortality in a defined area of water during the total survey period (M_o).

The total seabird mortality due to spilled oil per survey period in a defined area of water (M_o) is calculated as:

$$M_o = \frac{N_o * p_d}{p_s * a_w} + (N_o * p_a) \qquad [2]$$

where:

N_o = number of oiled birds that arrive along a given coastline during a survey period

p_d = proportion of birds found on shore that arrive dead

p_s = proportion of birds that die at sea which arrive on shore based on wind direction frequencies and sinking rates of carcasses at sea

a_w = area of water from which birds reaching the shore originate

p_a = proportion of birds found on shore that arrive alive

All correction factors, (d_p, s, p_d) have associated errors that are carried over from one step to the next. Errors should be combined using the Delta Method (Seber 1982), resulting in a final oiled seabird mortality estimate which has an estimate of associated standard error and 95 % confidence intervals.

7.2.2 OSMM applied to Newfoundland

7.2.2.1 Onshore component

Most of the parameters needed to estimate seabird mortality due to chronic oil discharges are geographically distinctive and can vary within and among years. In southern Newfoundland, a series of studies have determined beached bird detection rates (*d*) and persistence coefficients (*s*, chapter 3, Table 7.1). In chapter 3, it was determined that 10

Table 7.1. Parameter estimates or equations used in the OSMM for Newfoundland, and compiled from previous chapters. Standard errors are given where applicable.

Parameter	**Value**
Substrate-dependent probability that a bird present is detected (*d*)	Sand: 0.792 ± 0.029 Cobble: 0.855 ± 0.022 Boulder: 0.879 ± 0.019
Time dependent persistence coefficient (*s*)	$0.9718e^{-0.4329x} + 1.369^{e-2.86x*10-10}$ where x = survey interval in days
Length of coastline between survey limits on the Cape Shore where deposition can occur (l_T)	14.8 km
Mean length of shoreline surveyed per survey period (l_b)	5.6 – 6.8 km
OSMM extrapolation area[1]	56,670 km^2

[1] see Methods for details

days or less is the preferred survey interval to permit accurate interpretation of the number of beached birds found. As a result, the Canadian Wildlife Service conducted periodic surveys along a well monitored stretch of coastline in southeastern Newfoundland (Fig. 7.1) between October and March during the years 1998-1999, 1999-2000 and 2000-2001. These surveys provided the complimentary information needed to apply the OSMM to Newfoundland. Species composition and oiling rates were determined using international beached bird survey protocols (chapter 2), and the ratio of live versus dead oiled birds was recorded during all surveys. In chapter 3, a model was presented to extrapolate from the number of birds found during a survey to the number of birds that had been present on the beach between periodic surveys and data was presented for 7 day intervals. In instances where the interval between surveys was more or less than seven days, the technique described in chapter 3 was applied to determine correction factors. Prior to extrapolating to the entire coastline, the number of oiled birds on surveyed beaches during each winter survey period (N_{ob}) was calculated by summing the number of birds that arrived on all beaches through the survey period and multiplying that sum by the weighted mean proportion of oiled birds found (p_o) for the species considered. Because birds found on beach surveys included species that are only coastal (e.g. eiders, guillemots, gulls, Lock et al. 1994), only that proportion of N_{bs} that was made up of species found in both coastal and offshore areas, namely Common Murres (*Uria aalge*), Thick-billed Murres (*U. lomvia*) and Dovekies (*Alle alle*), was used to calculate p_o and considered for to extrapolate out to sea.

The total stretch of coastline that was considered suitable for carcass deposition on the Cape Shore (l_T), including the most southerly and northerly surveyed beaches, was determined using Environment Canada's Sensitivity Mapping Program (Table 7.1). The total number of oiled birds that arrived along the defined stretch of coastline during the winter survey period (N_o) was calculated by extrapolating the mean number of oiled birds on surveyed beaches during each winter survey period (N_{ob}) to beaches that were not surveyed due to their relative inaccessibility. This extrapolation was linear and assumed no birds are deposited on cliff edges, or if they are get eventually dislodged. It also assumed equal probabilities among beaches to collect birds (see below).

7.2.2.2 At-sea component

Previous studies determined wind-dependent catchment areas (a_w) and subsequent proportions of onshore depositions of birds that die at sea (p_a, chapter 6, Table 7.2). Wind information for the area for all three winters was obtained from the weather station in Argentia (Fig. 7.3), maintained by the Atlantic Climate Centre in Fredericton, NB, Canada. The best predictor for the number of recoveries on shore is the cumulative wind direction vector for the first three days after bird carcasses start passively floating at sea (chapter 6). It was not possible to determine the wind vector for birds found during each beach survey, as birds may drift for 10 days at sea (chapter 5) and remain on the beach for an average of 3 days (chapter 3). It is therefore unknown which winds each bird found may have been subjected to, as on average these may included any combination of winds during the 13 days before a survey was conducted.

Table 7.2. Wind vector frequencies for 3-day periods throughout winters (October to April) 1998-2001 in the study area, rounded to the nearest percent.

Wind direction	**1998/99**	**1999/2000**	**2000/01**	**Catchment area a_w (km^2)**	**Proportion of birds recovered (p_s)**
N	10	8	16	0	0
NW	37	32	44	432	0.058 ± 0.009
				1100**	0.058 ± 0.009
				2200	0.058 ± 0.009
W	40	29	18	2066	0.122 ± 0.011
SW	2	17	7	2066*	0.122 ± 0.011*
S	3	3	0	2066*	0.122 ± 0.011*
SE	2	3	0	0*	0*
E	3	3	2	0*	0*
NE	5	5	14	0*	0*

* assumed values (see Table 6.4)

** assumed most reasonable (chapter 6)

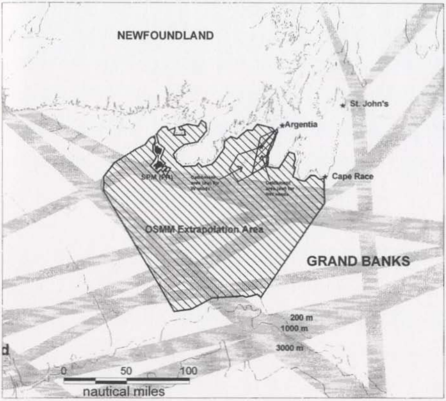

Fig. 7.3. Two wind-dependent catchment areas (a_w) for birds found on beaches along the Cape Shore of southeastern Newfoundland, Canada. Only one catchment area is shown for NW winds for clarity (see Fig. 6.9 and Table 7.2 for details). The overall Oiled Seabird Mortality Model extrapolation area and the location of the weather station in Argentia are indicated. Major shipping lanes are shown as grey bars.

As a consequence, wind data for each of the three winter survey seasons was partitioned into 3-day blocks. Wind directions were classed into eight directions (N, NW, W, SW, S, SE, E, NE) of 45 degrees each. The cumulative wind vector was calculated for each 3-day block, and the frequency of wind directions of these 3-day blocks season was determined throughout the winter survey. The proportion of (N_o) that arrived dead was apportioned based on the wind direction frequencies during which onshore recoveries can be expected, namely NW, W, SW and S winds (Table 7.2). Winds during which no recoveries have been observed, or none can be expected, were assigned no birds.

Each calculated fraction of N_0 was then extrapolated to the wind specific catchment area (a_w) using the proportion of birds recovered from that area based on drift block experiments (chapter 6, Table 7.2). For NW winds the most reasonable catchment area (a_w, chapter 6) was used in the first instance, although all were considered (see below). The number of dead oiled birds in the catchment area was then extrapolated into the larger OSMM extrapolation area (Fig. 7.3), based on the ratio of the catchment area to the defined OSMM area. We assumed average uniform density of dead oiled birds over the entire area in this extrapolation (see below).

The OSMM area selected was the southwest portion of the Grand Banks, including coastal areas around southeastern Newfoundland out to the shelf edge, west to the edge of the Bank, and east to Cape Race, the southeast corner of Newfoundland (Figure 7.3). Although exact boundaries were somewhat arbitrary, they were based on the knowledge that murres and dovekies occur from the coastal areas out to the shelf edge (Tuck 1961, Lock et al. 1994), that ship tracks cover the entire shelf (Fig 2.2, chapter 2), and that mystery oil spills of ship origin have been sighted in the entire area, with most ships discharging their bilges west of Cape Race (Lock et al. 1994, Environment Canada unpublished data).

Finally, the proportion of N_o that arrived alive was added to the extrapolated estimate for the OSMM area to produce the total seabird mortality estimate due to spilled oil per survey period in the defined area of water (M_o). Live birds were thus not extrapolated and were assumed to arrive from the entire OSMM area. This entire process was repeated for each winter.

7.2.2.3 Monte Carlo model

As an alternative to the deterministic model structure, a Monte Carlos procedure was used to simulate the effects of uncertainty on model outcomes (Ford et al. 1996). The model was run 1 million times for each year. For each run, a value was randomly selected from an appropriate normal probability distribution for each of the parameters that had associated measured errors.

7.2.2.4 Assumptions and perturbation analysis

The following critical assumptions were made when applying the general model to Newfoundland: 1) the probability of birds arriving on surveyed beaches and non-surveyed beaches is equal, 2) determined catchment areas (a_w) and recovery rates during southerly and southwesterly winds are accurate and comparable to the one estimated during westerly winds, and southeasterly, easterly, and northeasterly winds lead to no recoveries on the surveyed shoreline (Table 7.2, see chapter 6 for details), 3) the proportion of oiled birds found alive during weekly surveys is representative for the entire winter and area, 4) the OSMM area chosen is an accurate representation of the area in which the majority of seabird mortality due to oil occurs, and 5) seabird mortality due to oil is uniform overall in the OSMM area.

To test the sensitivity of some these assumptions, the following alternative scenarios were considered for 1999, and the effects of changing assumption sources on M_o was investigated: 1) the probability of birds arriving on surveyed beaches is twice as high than that of non-surveyed beaches, 2) alternative catchment areas (a_w) for NW winds are used (Table 7.2), 3) the proportion of oiled birds that arrive alive is 10 %, and 4) the OSMM area is 10% smaller than initially assumed. Oil risk zones have been determined based on the available information of seabird distribution and ship traffic densities (Environment Canada 1998, Figure 7.4). We thus also assumed an additional OSMM areas that included only the main extreme and high risk seabird oiling zones on the Grand Banks and along the South coast of Newfoundland (Fig. 7.4). Based on the modeling of the oil risk zones, it can be assumed that at least in these areas, oiled seabird mortality is about equal, and that the oiled seabird density determined for the catchment area is representative, or even conservative, for these zones, as catchment areas are not defined in the modeling as an oiling risk zone. All means are presented as ±1 SE.

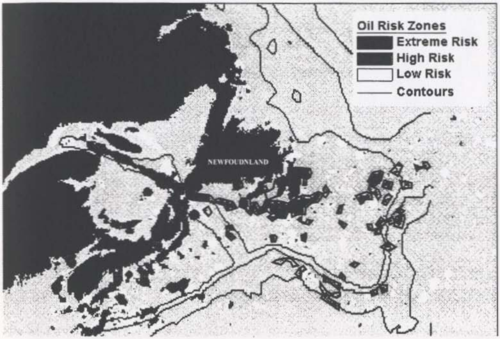

Fig. 7.4. Alternative area (black) used for large OSMM extrapolation on the Grand Banks based on Seabird Oiling Risk Zones (adapted from Environment Canada 1998).

7.3 RESULTS

7.3.1 OSMM applied to Newfoundland

7.3.1.1 Onshore component

A total of 67, 151, and 66 oiled and unoiled birds was found on survey beaches during the seven month winter periods (October to April inclusive) 1998/99, 1999/2000, and 2000/01, respectively. 15 % of all birds found were alive. During the first two winters, the survey period only lasted 20 and 21 weeks, compared to 27 weeks in 2000 (Table 7.3). Survey intervals varied among years due to weather-dependent accessibility and other logistical considerations. In 1998/99, the mean survey interval was 12 days (range 6-49), in 1999/2000 it was 9 days (range 6-27), and in 2000/01 it was 8 days (range 4-15). In order to facilitate direct comparison, all years were standardized to 27 weeks based on the median number of birds found per beach, per week and per winter. Correcting for detection probabilities and time dependent persistence coefficients, resulted in estimates that 336, 409 and 363 birds arrived on these beaches throughout the 27 week winter period, respectively (Table 7.3).

Species composition of birds found varied substantially among years (Table 7.4). Between 25 – 50 % of murres found could not be identified to species because only parts of birds were found. These were partitioned into Common and Thick-billed Murres based on the frequency of these species in the rest of the dataset. Common Murres were four times more frequent in 1998/1999 than in 1999/2000, and were totally absent in 2000/2001, while Dovekies were three to four times less frequent in 1998/1999 than in

Table 7.3. Summary information of the number of birds found per winter 1998-2001 along selected beaches on the Cape Shore in southeastern Newfoundland, Canada, and the result of subsequent extrapolations to the oiled seabird mortality area. Where applicable, standard errors are indicated.

Winter	**Survey period (weeks)**	**# surveys**	**Mean survey interval (days)**	**# birds found (N_{obs})**	**Estimated # birds on beaches (N_{bs} : standardized to 27 weeks)**	**Proportion of murres and dovekies**	**Proportion of oiled murres and dovekies (p_o)**	**Proportion of shoreline surveyed considered suitable for deposition (l_b / l_T)**	**Estimated # oiled murres and dovekies on Cape Shore (N_o)**	**Estimated # dead oiled murres and dovekies in catchment area**	**OSMM estimate (M_o)**
1998/99	20	14	12 ± 13	67	336 ± 6.5	0.821	0.872	0.44 ± 0.18	561 ± 147	5845 ± 1239	308,500 ± 66,216
1999/2000	21	19	9 ± 5	151	409 ± 6.6	0.689	0.955	0.46 ± 0.14	586 ± 41	5840 ± 598	298,200 ± 32,900
2000/01	27	26	8 ± 3	66	363 ± 2.7	0.788	0.605	0.38 ± 0.07	420 ± 10	4986 ± 608	338,800 ± 37,800
Mean				**95**	**360 ± 5.3**	**0.766**	**0.804**	**0.42 ± 0.13**	**522 ± 118**	**5557 ± 1028**	**315,200 ± 45,600**

Table 7.4. Species composition (% and number) of oiled and unoiled birds found during periodic beach bird surveys along selected beaches on the Cape Shore of Newfoundland, Canada, during winters (October-April) 1998-2001. Mean over the three year period is indicated.

Species	**1998**	**1999**	**2000**	**1998-2000**
Thick-billed Murres (*Uria lomvia*)	50.8 (34)	36.4 (55)	50.0 (33)	43.0 (122)
Common Murres (*Uria aalge*)	23.9 (24)	6.6 (10)	0.0	12.0 (34)
Dovekies (*Alle alle*)	7.5 (5)	25.8 (39)	28.8 (19)	22.2 (63)
Herring Gulls (*Larus argentatus*)	6.0 (4)	0.7 (1)	0.0	1.8 (5)
Gull species (*Larus* spp.)	4.5 (3)	0.7 (1)	0.0	1.4 (4)
Black-legged Kittiwake (*Rissa tridactyla*)	3.0 (2)	6.0 (9)	0.0	3.9 (11)
Black Guillemot (*Cepphus grylle*)	1.5 (1)	3.3 (5)	3.0 (2)	2.8 (8)
Common Loon (*Gavia immer*)	1.5 (1)	0.0	0.0	0.4 (1)
Great Black-backed Gull (*Larus marinus*)	1.5 (1)	1.3 (2)	0.0	1.1 (3)
Common Eider (*Somateria mollissima*)	0.0	3.3 (5)	3.0 (2)	2.5 (7)
Atlantic Puffin (*Fratercula arctica*)	0.0	0.0	1.5 (1)	0.4 (1)
Northern Gannet (*Sula bassana*)	0.0	0.7 (1)	1.5 (1)	0.7 (2)
Northern Fulmar (*Fulmarus glacialis*)	0.0	2.0 (3)	0.0	1.1 (3)
Glaucous Gull (*Larus hyperboreus*)	0.0	0.7 (1)	0.0	0.4 (1)
Storm-Petrels (*Oceanodroma* spp.)	0.0	1.3 (2)	0.0	0.7 (2)
Long-tailed Duck (*Clangula hyemalis*)	0.0	0.0	3.0 (2)	0.7 (2)
Common Merganser (*Mergus merganser*)	0.0	0.0	1.5 (1)	0.4 (1)
Cormorants (*Phalacrocorax* spp.)	0.0	0.0	1.5 (1)	0.4 (1)
Loons (*Gavia* spp.)	0.0	0.0	1.5 (1)	0.4 (1)
Shearwaters (*Puffinus* spp.)	0.0	0.0	1.5 (1)	0.4 (1)
Unidentified land bird	0.0	0.0	3.0 (2)	0.7 (2)
Unidentified bird	0.0	11.3 (17)	0.0	6.0 (17)

the following two years. By far the most abundant species was the Thick-billed Murre, accounting on average for 45 % of all birds found. Combined, these three species accounted for 82.1 %, 68.9 %, and 78.8 % of all birds found during the three winter periods. Because the distribution of most other species in winter tends to be coastal or is poorly understood (Lock et al. 1994), only birds from these three species were used for subsequent extrapolations of the number of birds dying at sea. Hence the total number of birds found on all surveyed beaches in each winter period were multiplied by these proportions, as well as by the weighted average proportion of oiled birds for these three species (Table 7.3), to arrive at the number of oiled murres and dovekies that arrived on surveyed beaches during each winter period (N_{ob}).

The mean proportion of shoreline surveyed (l_b) that was considered suitable to collect birds was 0.42, but varied among years (Table 7.3). Based on equation [1], we estimate that the total number of oiled murres and dovekies that arrived on the shoreline (N_0) between mid October and mid April was 561, 586, and 420, respectively (Table 7.3).

7.3.1.2 At-sea component

The distribution of three-day wind direction frequencies differed among years (Table 7.2). Most importantly, periods of wind with easterly components, where no recoveries take place (chapter 6), occurred over 31% of the time in 2000/01 compared to about 19 % in the two previous years. In addition, northwesterly winds (for which the proportion of birds recovered is lower, and the catchment area smaller, than during southerly to westerly winds; Table 7.2, Fig. 7.3), were 29 – 39 % less frequent in 1998/99 and

1999/2000, as compared to 2000/01. The large OSMM area that contained essentially the southwest part of the Grand Banks from coastal areas to the shelf edge and east to Cape Race, comprised 56,670 km^2 (Fig. 7.3).

Despite differences among the initial number of birds found, accounting for differences in species composition, proportion of oiled birds found and wind direction, resulted in total seabird mortality estimates within the large OSMM extrapolation area for the three winter periods within 41,000 birds of each other (Table 7.2), averaging 315,200 (95 % C.I. 225,800 – 404,600) oiled murres and dovekies per year within the defined area of water (Fig. 7.3). Of these, 68 % (214,600 ± 31,000) were Thick-billed Murres, 10 % (31,700 ± 5,900) were Common Murres, and 22 % (69,900 ± 8,700) were Dovekies.

7.3.2 Monte Carlo model

Results of the Monte Carlo model showed that drawing uncertainty parameters from random distributions skewed the results towards higher estimates of mortality (Fig. 7.5, Table 7.5), most likely because the interactions of the model processes are multiplicative. For all three years combined, the overall yearly mortality estimate due to oil averaged 321,900 (95 % C.I. 217,800 – 458,600) murres and dovekies within the defined area of water (Table 7.5).

Figure 7.5. Distribution of oiled seabird mortality estimates within the OSMM extrapolation area based on Monte Carlo model outcomes based on 1 million interations. Input parameters were randomly selected for each iteration based on a range of possible values.

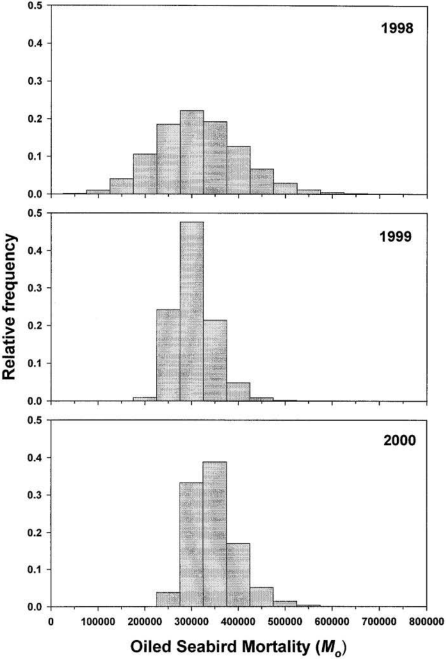
0.5
0.4
0.3
0.2
0.1
0.0
1998
1999
2000
Relative frequency
0
100000
200000
300000
400000
500000
600000
700000
800000
Oiled Seabird Mortality (M_o)

Table 7.5. Mean estimates with 95 % Confidence Intervals of seabird mortality due to oil (M_o) within the OSMM extrapolation area based on Monte Carlo simulations. Input parameters were randomly selected for each iteration (1 million iterations per year) based on a range of possible values.

Year	**Mean M_o**	**Lower 95% C.I.**	**Upper 95% C.I.**
1998	314,800	147,700	509,800
1999	304,300	235,700	400,670
2000	346,600	270,000	465,440
1998-2000	**321,900**	**217,800**	**458,600**

7.3.3 Perturbation analysis

Assuming that 50 % more birds arrived on surveyed beaches than on non-surveyed beaches resulted in a 41% reduction in the estimate of M_o. Varying the proportion of birds that arrive on the beach alive and varying the large OSMM area, resulted in linear changes in the final estimate (Table. 7.6), as expected from Eqn [2]. Finally, changes in the catchment area (a_w) lead to inverse proportional changes in M_o. Using high and extreme seabird oiling risk zones on the Grand Banks as the OSMM extrapolation resulted in a 22 % larger area than the one defined in Figure 7.3 (69,096 km^2). Applied to all years, this resulted in an average M_o of 384,300 ± 55,600.

7.4 DISCUSSION

To our knowledge, this is the first report of a mathematical model to assess seabird mortality from chronic spills, and the first empirically-based seabird mortality estimate due to chronic oil pollution within a given area. Although many of the same parameters are required (Ford et al. 1996), the process of estimating seabird mortality due to chronically spilled oil over the entire winter differs from the approaches to estimate mortality caused by single large events. The main difference lies in ways to extrapolate counts of beached birds to estimates of numbers dying at sea. Unlike large oil spill events, the timing and location where chronic spills occur is unknown, and hence no overlays of slick trajectories, seabird distribution patterns and carcass trajectories can be performed

Table 7.6. Results of 6 perturbation scenarios using the Oiled Seabird Mortality Model estimates with data for data collected in southeast Newfoundland in the winter of 1999/2000. Standard errors are shown where appropriate. The six scenarios compared to the initial assumptions used to derive estimates in Table 7.3 (base) are: assuming that birds are 50 % more likely to wash ashore on surveyed than on non-surveyed beaches (Δ length), assuming that only 10% of birds found on beaches arrive alive (Δ live birds), assuming that the catchment area for NW winds is 60 % smaller or 22 % larger (Δ a_w; Table 7.2), and assuming that the OSMM area is 10 % smaller (Δ OSMM area). Parameters varied in the model are bolded for clarity.

Perturbation	Estimated # birds on beaches	Proportion of shoreline surveyed considered suitable for deposition (l_b / l_T)	Estimated # oiled murres and dovekies on Cape Shore (N_o)	Proportion alive (p_a) and proportion dead (p_d)	Estimated # dead oiled murres and dovekies in catchment area	Catchment area (a_w) for NW winds (km^2)	OSMM area (km^2)	OSMM estimate (M_o)
Base	409 ± 7	0.46 ± 0.14	586 ± 41	0.15	5840 ± 598	1100	56,670	298,200 ± 32,900
				0.85				
Δ length	409 ± 7	**0.60 ± 0.18**	343 ± 34	0.15	3418 ± 384	1100	56,670	174,600 ± 21,000
				0.85				
Δ live birds	409 ± 7	0.46 ± 0.14	586 ± 41	**0.10**	6183 ± 633	1100	56,670	315,700 ± 34,800
				0.90				
Δ a_w	409 ± 7	0.46 ± 0.14	586 ± 41	0.15	5840 ± 598	**432**	56,670	766,800 ± 100,300
				0.85		**2200**		232,800 ± 21,100
Δ OSMM area	409 ± 7	0.46 ± 0.14	586 ± 41	0.15	5840 ± 598	1100	**51,003**	268,400 ± 29,600
				0.85				

(Ford et al. 1987, Ford et al. 1996). Nevertheless, it is possible to derive accurate estimates of seabird mortality with associated measures of error due to chronic spills throughout a survey season, and the mathematical model presented here provides the means by which the time and location specific data can be interpreted and applied.

Applied to Newfoundland, we estimate, that, on average, about 315,000 ± 45,600 seabirds die due to illegal discharges of machinery waste oils from tankers, cargo and container vessels each winter off the coast of southern Newfoundland. If model parameters are chosen randomly based on their degree of uncertainty, the estimate is only slightly higher at 322,000 ± 56,000 seabirds. Regardless, these new more refined estimate lie well within the two previous estimates of seabird mortality due to chronic oil pollution of 20,000-500,000 that exist for Newfoundland (Tuck 1961, Piatt et al. 1985). Although the final 95% C.I. only ranges between 226,000 - 405,000 (or 218,000 – 459,000) birds, it is noteworthy that the standard error of the estimate for 1998/99 was two to three times as large as in the following years. As discussed in chapter 3, the accuracy of estimates for the total number of birds that arrive on a beach between surveys is related to the interval between surveys, which was also greatest and most variable in 1998/99. More importantly, the error increased substantially in 1998/99 as we extrapolated to the entire shoreline. Particularly in one instance, the proportion of shoreline surveyed was below 10 %, and the error of this estimate contributed over 90 % of the resulting error in N_0. These differences highlight the necessity to conduct beach surveys within 8 days of each other and to survey as much as possible of the available shoreline, if estimates are to be used to calculate total seabird mortality with reasonably restricted confidence intervals.

7.4.1 Assumptions and perturbation analysis

Parameters that lead to the greatest uncertainties in the final estimate was the proportion of the total extrapolation area comprised of the catchment area (a_w), the proportion of depositable beaches searched, and the proportion of birds on surveyed versus unsurveyed beaches. While the latter parameter is most likely not as skewed as the scenario presented (see below), the importance of surveying as many of the beaches where deposition can occur cannot be overstated, and coincides with findings for mortality models applied after large oil spills (Ford et al. 1996). Also similar to the model presented by Ford et al. (1996) is the source of uncertainty and sensitivity of our model to the catchment area and its recovery rate. As discussed below, more drift block studies to reduce this uncertainty are desirable.

Several assumptions were made to arrive at a final estimate of about 315,000 murres and dovekies. When systematic beached bird surveys were initiated by the Canadian Wildlife Service in 1984, beaches were chosen based on accessibility, but also on the knowledge that those beaches collected many birds (R. D. Elliot, pers. comm.). If these selected beaches are more likely to collect birds than other sections of the coastline that are considered suitable for carcasses to wash ashore, but that were not surveyed, we have over-estimated total seabird mortality. Assuming that it is 50 % more likely for birds to wash ashore on surveyed beaches than on non-surveyed beaches, we would have overestimated total seabird mortality due to oil by 41 % (Table 7.5). However, a difference in deposition rates between surveyed and non-surveyed beaches is not supported by recent observations. Drift block experiments conducted along the south coast of Newfoundland appear to indicate that several non-surveyed beaches collected as many blocks as those surveyed regularly (chapter 6). This is further supported by recent observations by beach surveyors that indicate that several areas outside the survey beaches frequently collect birds (P. C. Ryan pers. comm.). Long-term daily surveys are needed to quantify possible differences in deposition rates among beaches. Yet given recent observations, and in the absence of current data supporting inequality between stretches of suitable shoreline capable of collecting birds floating at sea, we assume that all beaches have equal probability of collecting birds.

It was also assumed that the catchment area during southerly and southwesterly winds is comparable to the one measured during westerly winds, and that southeasterly, easterly, and northeasterly winds lead to no recoveries along the surveyed shoreline (see chapter 6 for details). This assumption was made in the absence of any drift block recovery data during such winds, but was based on the geography of the area and what was learned during observed conditions. Although this assumptions requires further testing, it is believed to be reasonable and likely not a large source of error. The exact determination of the catchment areas, however, has a large influence on the final mortality estimate and extensive drift blocks experiments laid out in a grid in the area of interest are needed to assess these as best as possible (chapter 6). Although a catchment area of 1100 km^2 during NW winds was considered to be the best approximation with the data available, it is likely this area is underestimated and probably more similar in size to the catchment area during W winds but shifted to the north (Fig. 6.9). In this case, we may have overestimated M_0 by about 20 %.

We are not aware of any published values for the proportion of live oiled birds found during periodic surveys, designed to monitor chronic oil pollution, for comparison with this study. No live oiled birds, although many dead oiled birds were detected during 70 daily beach surveys spread over three winters conducted for carcass persistence studies (chapter 3). Information gathered during emergency responses on the west coast of the US after large catastrophic spills documented that an average of 15% of all oiled birds found are alive (R. G. Ford, pers. comm.). In these instances, however, the oil is often close to shore and birds are heavily oiled. These factors would presumably increase the likelihood for oiled birds to come to shore to exit the water and avoid acute hypothermia. Over 75 % of birds found affected by chronic oil pollution have 25 % or less of their bodies oiled and are highly emaciated (chapter 2). In contrast to large spills, chronic spills often occur offshore, and live oiled birds on beaches usually die within hours of arriving onshore. This suggests that oiled birds lived for periods of time at sea after oiling and only attempt to leave the water as a last resort after all energy reserves have been depleted. As a result, we consider a rate of 15 % oiled live birds on beaches in areas of chronic oil pollution a maximum estimate. Varying the proportion of live oiled birds in the model lead to an inverse linear response in the final mortality estimate. If we assume that only 10 % of oiled birds come ashore alive, we may have underestimated seabird mortality due to chronic oil pollution by up to 5 %. In contrast, it could be argued that birds oiled to a small extent retain the ability to fly for a period of time after oiling and that they fly or swim into the catchment area and die there. These birds would consequently be counted as dead birds on beaches and extrapolated accordingly, leading to an overestimate in seabird mortality due to oil in the OSMM area. However, there are no data to support these assumption; clearly the behaviour of oiled birds at sea is still poorly understood and requires further investigation.

Finally, assumptions were made about the size of the OSMM extrapolation area and the uniformity of seabird mortality due to oil within it. The ocean area used, essentially the southwest portion of the Grand Banks, was based on three factors: 1) the knowledge that murres and dovekies occur from the coastal areas out to the shelf edge (Tuck 1961, Lock et al. 1994), 2) that ship tracks cover the entire shelf throughout the year (chapter 2), and 3) that mystery oil spills of ship origin have been sighted in the entire area (Lock et al. 1994, Environment Canada unpublished data). Thus, the area was chosen to include coastal areas and offshore areas south and west to the shelf edge and east to the level of Cape Race, the southeast corner of Newfoundland (Figure 7.3), because most sightings of mystery slicks occur in this area (Environment Canada, unpubl. data) and it appears from regional differences in oiling rates that the discharge of oily waste from ship bilges is higher west of Cape Race (Wiese 1999). Although the general area is based on these three factors, the exact boundary is somewhat arbitrary. The perturbation analysis showed that varying the size of the area by 10 % results in a direct 10 % change in the final estimate. Hence alternative scenarios should be considered based on limited information collected elsewhere. Oil risk zones were determined based on the available information of seabird distribution and ship traffic densities (Environment Canada 1998, Figure 7.4). Unfortunately, a large number of ships that cross the area in question are not reported to local authorities (chapter 2), and as a result, ship density is most likely underestimated for many parts of the continental shelf frequented by birds. In addition, detailed information on seabird distribution and density during the winter months is very sparse on the Grand Banks and south of Newfoundland (Lock et al. 1994), and not detailed enough to define species densities and specific areas that should be included in, or excluded from, an extrapolation area. Several areas on the Grand Banks with high risk oiling may not be coded as oil risk zones, because no seabird distribution data exist for those areas. Regardless, we present an alternative scenario to define the OSMM extrapolation area that includes only the main extreme and high risk seabird oiling zones on the Grand Banks and along the south coast of Newfoundland (Fig. 7.4). Based on the modeling of the oil risk zones, we assumed that within these areas, oiled seabird mortality is about equal to that estimated in the catchment area, and that the oiled seabird density determined for the catchment area is thus representative for these zones. Indeed, the catchment area used in this study is not even defined in the modeling as an oiling risk zone, but clearly birds are oiled in this area. Using the oil risk zones on the Grand Banks to define the OSMM area resulted in an extrapolation area that is about 22 % larger than the one defined in Figure 7.3 (69,096 km^2), and applied to all years lead to an average estimate of M_o of 384,300 ± 55,600 birds.

Apart from defining areas where seabird most likely get oiled, we assumed uniformity in the number of oiled murre and dovekies within the OSMM area and when compared to the catchment areas. This implies, however, the Murres and Dovekies are uniformly distributed throughout the area, which is most likely not the case, as they can be expected to be associated with prey densities (Cairns and Schneider 1990, Montevecchi and Berruti 1990, Davoren 2001), and are therefore patchily distributed. However, offshore surveys conducted between 1967 and the late 1980's indicate the murres and dovekies occur on the coast and on the shelf edge, as well as on other areas on the Grand Banks where surveys have taken place during winter (Lock et al. 1994). Concentrations of Thick-billed Murres in the Bering Sea were also observed more than 200 km offshore during the non-breeding season (Hatch et al. 2000). On the other hand, local observations of hunters in Newfoundland indicate large concentrations of Thick-billed Murres inshore as opposed to 50 km away from the coast (R. D. Elliot), although no comparable observations exist beyond that distance. Dovekies, on the hand, appear to be more frequent offshore than inshore, as they are rarely seen in high concentrations by hunters, yet millions of dovekies migrate onto the Grand Banks from Iceland and Greenland during winter (Chardine 1995). As earlier, it is impossible to quantitatively assess to what degree densities of murres and dovekies differ throughout the region based on the available data. In its absence, we assume a uniform distribution, and recognize that a bias may exist, that for certain species and areas, but perhaps not overall, may lead to slightly different mortality estimates due to chronic oil pollution.

7.4.2 Seabird mortality due to chronic oil pollution in Atlantic Canada

In spite of the limitations in the data used to calculate these oiling risk zones, the range of estimates that result depending on the areas that are chosen (174,600 – 766,800), and uncertainties in other model parameters discussed above, we consider a level of mortality due to chronic oil pollution of about 300,000 birds per winter period a reasonable estimate. We even consider our estimate to be a conservative representation of yearly seabird mortality inflicted by illegally spilled oil in Atlantic Canada for several reasons. First, the most conservative side of each measured parameter that would yield the lowest overall mortality estimate was used in the model, i.e. we considered blocks that were found onshore until day 14, even though on average birds sink within 8 days, and 95% sink within 10 days (chapter 5), we assumed the proportion of live birds to originate from the entire OSMM area, and it was assumed that recovery rates of blocks are equal to birds, even though survey effort for blocks was more intense than regular periodic surveys and blocks were painted bright orange. Second, the estimate only include the winter months, even though oil pollution does occur in the summer (chapter 2), when small numbers of oiled birds are found on beaches (Wiese and Ryan 1999). Third, this estimate includes only murres and dovekies, even though other seabird species were found oiled on beaches (chapter 2, Table 4). Fourth, it is often believed that the southeast coast of Newfoundland is the most affected area, as many of the major shipping lanes between North America and Europe converge in this region (chapter 2, Chardine 1990), and it is that where seabird concentrations of the most vulnerable species are highest (Lock et al. 1994, Chardine 1995). However, reports exist of oiled birds from other parts of Atlantic Canada (Fig. 7.6, chapter 2, Lock 1992, Wiese et al. 2001, Environment Canada unpublished data), that are not reflected in the current estimate. Finally, there is ample evidence of long-term effects of oil on seabirds, such as decreased reproductive success (Hartung 1965, Ainley et al. 1981, Albers 1983, Leighton 1995), or decreased

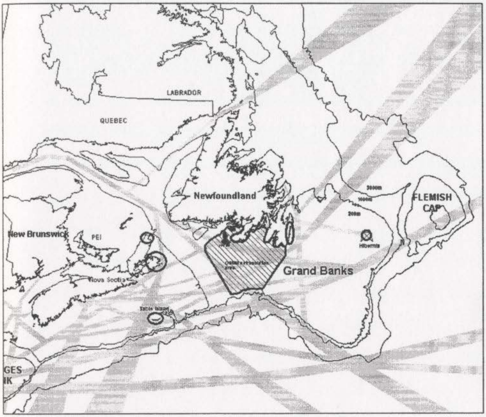

Fig. 7.6. The Oiled Seabird Mortality Model extrapolation area on the southeast coast of Newfoundland, Canada and other locations of known oiled bird reports outside the estimated mortality area (black circles). Grey bars are shipping lanes.

long-term survival (Peakall et al. 1980, Esler et al. 2000). However, the amount of additional mortality these long-term effects may cause is unknown and cannot currently be estimated.

7.5 CONCLUSIONS

The Oiled Seabird Mortality Model presented here is the first of its kind. It is applicable to any area where the required geographically-specific information is available. Given that the presence of chronic oil pollution is already well documented in many parts of the world where seabirds occur (Camphuysen and Heubeck 2001), we hope that comparable analysis can take place.

The absence of a strong relationship between spill volume and seabird mortality points to the necessity to thoroughly investigate impacts of smaller spills (Burger 1993a). Concerns have been raised that chronic small spills may kill as many seabirds as do more widely publicized large spills, and be equally, or more detrimental to long-term population stability (Hunt 1987, Camphuysen 1989, Burger and Fry 1993, Oka et al. 1999). Our results confirm this first concern. The level of annual mortality documented here is comparable to the *Exxon Valdez* kill in 1989 (Ford et al. 1996, Piatt and Ford 1996). The crucial difference is that this degree of mortality due to chronic spills occurs every year of the coast of Newfoundland, and very little of it is visible on beaches.

The most critical element in the model that allows an accurate interpretation of the number of birds found on beaches, is the wind direction specific extrapolation for catchment area and proportion of birds lost at sea. Without this approach, a constant value for the proportion of birds lost at sea, and a uniform catchment area, would have to be assumed. With such an approach, final estimates would be just as unrepresentative and variable as the number of birds found on beaches.

For Newfoundland, several estimates in the model need to be further investigated to refine the final estimate. First, accurate ship traffic densities and seabird-at-sea densities are needed to determine species-specific risk zone and better define the OSMM area. Second, catchment areas need to be better resolved with additional drift block experiments in Placentia Bay (chapter 6). Third, sinking rates of carcasses at sea and the occurrence of scavenging should be investigated, ideally using fresh free floating radio-tagged carcasses (Ford et al. 1996). In conjunction with detailed surface current studies, a better understanding of the movement of dead birds relative to winds and currents could be gained. Fourth, the behaviour of live oiled birds needs to be investigated to more accurately determine the proportion of oiled birds that arrives onshore alive and from how far, and fifth, additional beached bird deposition studies should be carried out to determine whether surveyed-beaches have equal or different deposition rates than unsurveyed stretches of coastline suitable to birds to wash ashore.

Despite remaining uncertainties in some parameter estimates, we believe that an estimated annual seabird kill of 300,000 due to chronic oil pollution in Atlantic Canada is conservative, and such a high level of mortality unacceptable. It is caused by illegal activities that could be avoided. International conventions such as MARPOL, and national laws such as the Canada Shipping Act, the Environmental Protection Act and the Oceans Act, prohibit these discharges. Furthermore, the Migratory Birds Convention Act protects these species by law. With a scientifically defensible mortality estimate now in hand, we look to national and international authorities to increase efforts to reduce this problem regionally and internationally (see section 9.5 and Wiese 2002 for a detailed discussion on how to reduce chronic oil pollution).

Finally, we estimate that over 65 %, or roughly 200,000, of the birds killed annually due to these illegal activities are Thick-billed Murres. This species is already subject to extensive summer and winter hunting in Greenland (Kampp 1991, Evans and Kampp 1991, Christensen 2001), as well as to a traditional murre hunt in Newfoundland, which kills an additional 250,000 – 300,000 birds a year (Chardine et al. 1999). The combined and sustained magnitude of these anthropogenic mortalities raises serious conservation concerns for the Northwest Atlantic populations of this species, as related populations in the North Pacific Ocean showed long-term impacts after a single *Exxon Valdez* kill in Alaska (Lance et al. 2001). We recommend that a population model be constructed and the population-level impacts be investigated for species such as Thick-billed Murres, based on these mortality estimates.

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CHAPTER 8 – IMPACTS OF CHRONIC MARINE OIL POLLUTION AND THE MURRE HUNT IN NEWFOUNDLAND ON THICK-BILLED MURRE (*URIA LOMVIA*) POPULATIONS IN THE EASTERN CANADIAN ARCTIC

8.1 INTRODUCTION

Much concern has been voiced over the potential impacts of chronic oil pollution on seabird populations (Bourne 1968, Croxall 1975, Camphuysen 1989, Dunnet 1982, Albers 1995, Newton 1998), and several authors have suggested that it may be more detrimental to long-term population stability than occasional large spills (Hunt 1987, Burger and Fry 1993, Oka 1999). Unequivocal demonstration of a direct cause and effect relationship between oil pollution and fluctuations in the size of populations is difficult for many reasons, including: 1) estimates of the magnitude of mortality from acute or chronic spills are often too imprecise; 2) seabird colonies are difficult to census and they are often widely dispersed or sensitive to disturbance; 3) it is often difficult to ascribe changes in abundance to specific causes in light of other anthropogenic pressures (e.g. hunt, by-catch), normal year-to-year variations in populations, the presence of large non-breeding cohorts, and the occurrence of natural phenomena (e.g. El Niño); 4) series of colony censuses and estimates of population trends prior to oil spills often do not exist; 5) colony census only count breeding adults; and 6) most mortality from oil occurs in winter, when breeding populations from different regions may be impacted simultaneously (Piatt et al. 1991, Burger and Fry 1993).

Despite the difficulties, attempts have been made to link large oil spills to observed seabird population declines (Hope-Jones et al. 1970, Baillie and Mead 1982, Stowe 1982, Hunt 1987, Page et al. 1990, Piatt et al 1991, Takekawa et al. 1990, Edwards and White 1999, Irons et al. 2000). Many of these studies have accumulated evidence that immediate, and in some cases long-term, population effects occur after large oil spills. Regardless, some debates continue (Baker et al. 1990, Piatt et al. 1991, Parrish and Boersma 1995, Piatt 1996, Wiens 1996, Murphy et al. 1997, Lance et al. 2001), perhaps because the focus often lies only on decreasing populations (Hunt 1987, Burger 1993, Burger and Fry 1993) or on the disappearance of small local colonies (Piatt et al. 1991). Yet more subtle effects can occur, such as reductions in population growth (Hunt 1987), declines in numbers of non-breeders, reduced adult survival (Esler et al. 2000) or reduced reproductive success (Ainley et al. 1981), that do not necessarily lead to detectable population declines.

These more subtle effects of oil mortality on populations are important in assessing impacts of chronic oil spills. Yet with the exception of Nur et al. (1997), no attempts have been made to measure or model the impacts of mortality caused by chronic oil pollution on seabird populations. Clearly, one of the limitations has been the lack of good estimates of seabird mortality caused by these events, and the possibility to link them to monitored populations.

In Newfoundland, Canada, it has been estimated that on average, about 300,000 seabirds (murres and dovekies) are killed every winter due to the illegal discharges of oily waste from ships (chapter 7). One of most numerous seabirds in the northern hemisphere (Gaston and Hipfner 2000), the Thick-billed Murre (*Uria lomvia*) is also the most frequent and vulnerable victim of chronic marine oil spills in Atlantic Canada (chapter 2). In addition to the mortality caused by oil pollution, Thick-billed Murres are killed every year during the traditional Newfoundland winter murre, or turr, hunt (Chardine et al. 1999). This hunt, estimated at 600,000 – 900,000 in the 1980s (Elliot 1991), was markedly reduced after restrictive hunting regulations came into effect in 1993, and is now believed to range between 173,000 and 287,000 per year (Chardine et al. 1999).

Fluctuations in numbers of breeders at two Thick-billed Murre colonies at opposite ends of their range in the eastern Arctic, suggests that population changes may be determined by events on the common wintering grounds (Gaston *in review*). Although some Thick-billed Murre populations in the eastern Canadian Arctic appear to be stable or increasing (Gaston 2002), it is important that the impacts of these anthropogenic causes of mortality on Thick-billed Murre populations be examined. We developed a novel Thick-billed Murre population model to examine the relative and cumulative effects of mortality due to oil and harvest that occur in Newfoundland waters on the population of Thick-billed Murres that breed in the eastern Canadian Arctic.

8.2 METHODS

8.2.1 Demography

Thick-billed Murres, like all auks, are characterized by a low fecundity – high survival life history strategy: clutch size is one, age of first breeding is 4-6 years, adult survival is 90 % or higher, and individuals live up to 30 years of age (Gaston and Hipfner 2000). In the 1980s there were an estimated 1.5 million breeding pairs distributed among 10 colonies in Canada's eastern Arctic (Nettleship and Evans 1984). However, this number underestimates the current population size by about 30% (Gaston and Jones 1998, A. J. Gaston, pers. comm.). Demographic parameters of Thick-billed Murres in the Canadian Arctic were reported by Nettleship and Birkhead (1985) but are best known from Coats Island, northern Hudson Bay (Gaston et al. 1994, Donaldson 1995, Gaston 2002). However, in some cases these estimates are considered to be biased low, due to limitations in monitoring methodology (Gaston 2002) and the inability to detect emigration (Gaston et al. 1994). As a result, demographic parameters used for the model (Table 8.1) are in some instances slightly different from those published previously.

8.2.2 Migration

In the North Atlantic, Thick-billed Murres breed in the eastern Canadian Arctic, Greenland, Iceland, Spitsbergen and several other locations in Northern Europe (Gaston and Hipfner 2000, Fig. 8.1). In August, chicks and adults leave their colonies and begin their migration to wintering areas (Fig. 8.1). The relative proportions from each breeding area that make up the winter population encountered off Newfoundland have not

Table 8.1. Summary of demographic parameters used for the pre-breeding population model for Thick-billed Murres breeding in the eastern Canadian Arctic.

Parameter	Value	Source
Population size N (breeding pairs)	1,950,000	Nettleship and Evans 1984, Gaston and Jones 1998, A. J. Gaston pers. comm.
Survival of juveniles S_0	0.52	Donaldson 1995
Adult survival (2+ years old) S_a	0.91	Nettleship and Birkhead 1985, Gaston et al. 1994, Gaston 2002, Gaston and Hipfner 2000, A. J. Gaston pers. comm.
Proportion breeders Pb_a		Gaston et al. 1994, Gaston unpubl. data
1 year-old	0.000	
2 year-old	0.000	
3 year-old	0.025	
4 year-old	0.367	
5 year-old	0.700	
6 year-old	0.931	
> 6 year-old	0.980	
Fecundity m_a		Gaston et al. 1994, Gaston unpubl. data
1 year-old	0	
2 year-old	0	
3 year-old	0.17	
4 year-old	0.20	
5 year-old	0.33	
6 year-old	0.46	
7 year-old	0.41	
8 year-old	0.54	
9 year-old	0.52	
10 year-old	0.58	
11 year-old	0.65	
> 11 year-old	0.73	

Fertilities calculated as $F_a=Pb_a*m_a*S_0*0.5$

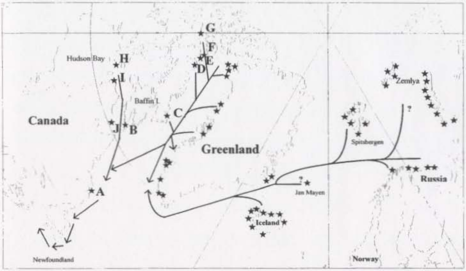

Figure 8.1. Location of Thick-billed Murre colonies in the North Atlantic and eastern Canadian Arctic. Canadian colonies are (A) Gannet Islands, (B) Hantzsch Island, (C) Reid Bay (The Minarets), (D) Cape Graham Moore, (E) Cape Haye, (F) Coburg Island, (G) Prince Leopold Island, (H) Coats Island, (I) Digges Island, and (J) Akpatok Island. Approximate direction of fall migration is shown. Adapted from Gaston (1980), Kampp (1988), and Donaldson et al. (1997).

been clearly quantified. However, reports from banding studies of Thick-billed Murres at colonies in the eastern Canadian Arctic and Greenland, indicate that roughly 80 % of Thick-billed Murres that overwinter off Newfoundland and Labrador originate from the eastern Canadian Arctic and about 20 % from western Greenland (Tuck 1961, Gaston 1980, Kampp 1988, Donaldson et al.1997). Band recoveries confirm that some originate from as far as Iceland and Spitsbergen (Gaston and Hipfner 2000), although their contributions to the overall winter population are unknown and probably minimal. Some Thick-billed Murres, mostly non-breeders and perhaps a small number of breeders, can be found in Newfoundland year-round (Tuck 1971, Montevecchi and Tuck 1987), although large numbers are only present between November and April, with a clear peak in January and February (Tuck 1961, Brown et al. 1975, Gaston 1980).

8.2.3 Mortality due to chronic oil pollution

Thick-billed Murres are the most frequent and most vulnerable victim of chronic oil pollution in Newfoundland (chapter 2), and comprised between 56.3 % and 87.5 % (160,735 - 274,877) of the estimated annual kill over three winters 1998/99-2000/01 (chapter 7). To assess the impacts of this kill, it is critical to examine whether birds of different age classes have different probabilities to encounter oil slicks, and when they do, if their probability of getting oiled differs. Because an oiled bird is a dead bird (chapter 2), we are not aware of any mechanism that causes one age class to be more likely to get oiled once they encounter a slick. Analysis of banding data indicates that the spatial distribution of different age classes varies throughout the winter around Newfoundland (Donaldson et al. 1997). Limited data from beached bird surveys conducted in

Newfoundland between 1984-2000 indicated that 69 % (n=59) of birds which could be aged were adults (>1 year), which would imply that birds 2 years and older are affected less by oil than their abundance would infer from a stable age distribution (90%, Eqn 1). However, 80 % (n=368) of oiled Thick-billed Murres found were not aged, making exact inferences on the age distribution of oiled Thick-billed Murres carcasses difficult. We are not aware of any conclusive evidence that indicates that chronic oil pollution affects certain age classes differentially, and assume oil mortality is not age-specific.

8.2.4 Mortality due to hunt

The most recent estimates of the magnitude of the murre hunt in Newfoundland and Labrador average 227,000 birds annually, and ranged between 173,000 and 287,000 during the period 1995/96 to 1997/98 (Chardine et al. 1999). Thick-billed Murres make up 95 % of these birds, while the remainder are Common Murres (*Uria aalge,* Elliot 1991). First-year birds, which comprise about 10 % of the population (Eqn 1) appear to be at a greater risk of being shot than older birds. Gaston et al. (1983) reported that 53 % of murres killed were first-year birds, while Elliot (1991) documented 50 %. From bands recovered in the hunt, Donaldson et al. (1997) determined that 58 % were one-year olds, 25 % two-year olds, 8% three-year olds, 6 % forth-year olds, and 2 % five year-olds. Recent studies (Gaston 2002) indicate no significant changes in the proportion of one-year olds killed in the hunt over time, but significant decreases for older birds. Most markedly, recoveries of birds banded as breeders virtually ceased after 1990. Based on these findings, it was assumed that 50% of Thick-billed Murres killed in the Newfoundland hunt are in their first year, and that the rest of the harvest is spread among the other age classes according to their abundance. We did not use age-class specific return rates from banding data (Donaldson et al. 1997) because information exists only for 5 age-classes, and the model was based on 12 (see below).

8.2.5 Population model

We developed a stochastic (demographic and environmental), age-structured, pre-breeding, Lefkovitch population projection matrix to model population dynamics (Lefkovitch 1965, McDonald and Caswell 1993, Caswell 2001). This model was based mainly on information from the breeding population at Coats Island (Gaston et al. 1994, Gaston and Hipfner 2000). The computer model written in Matlab consisted of 12 age classes, and only females were considered. We assumed a sex ratio of 1:1 (Tuck 1961), so fecundity parameters from Table 8.1 were divided by 0.5 for the model. We define breeding birds as those that are 5 years and older (Gaston and Hipfner 2000), although a few 3 and 4 year olds breed. Stochasticity in survival rates, as well as in fecundity, was assumed to be ± 5 % of the observed values for each age class (Gaston et al. 1994, Donaldson 1995). No estimates of variance were available for the proportion breeding per age class, so these values were held constant. Stochasticity in mortality due to oil and harvest was based on the observed range of values described above. Harvest values were multiplied by 0.95 in order to eliminate Common Murres from the hunt, by 0.8 to take into account only birds from Canadian colonies, and by 0.5 to consider only females. This resulted in a range of 65,705 - 109,010 female Thick-billed Murres from Canadian colonies killed annually in the hunt. Similarly, oil mortality values were multiplied by 0.8 and 0.5 to give an effective range of 64,294 - 109,951 female Thick-billed Murres from Canadian colonies killed by chronic oil pollution. For all runs, random uniform numbers within each range were drawn for each projection. A uniform distribution was chosen because mortality estimates come only from three years of data, and in order to increase variance.

The effects of anthropogenic mortality on a population are most easily determined by comparing survival rates between affected and unaffected populations. In the case of Thick-billed Murres, however, we are not aware of any populations that are not subjected to anthropogenic mortality to some extent, and we were interested in quantifying the relative effects on this particular population. Therefore a comparison of survival rates of impacted and non-impacted Thick-billed Murre populations breeding in the eastern Canadian Arctic is simply not possible. As a result, we evaluated the cumulative and individual effects of anthropogenic mortality due to oil and hunting on population dynamics occurred in three phases. Phase one determined baseline growth rates from vital rates obtained from Coats Island (an impacted population). In phase two, birds killed by explicit anthropogenic mortality due to oil and hunting (actual number of birds killed) were removed from the population, and the reduction this caused in the population growth rate was determined. In essence in this phase, birds were killed twice: once explicitly due to anthropogenic causes, and once implicitly by using observed survival rates of a population that is already impacted by anthropogenic mortality. In phase three, we corrected the survival rates to compensate for the anthropogenic mortality (oil and hunting) by increasing the survival rates sufficiently to return to the population growth rate calculated in phase one. In essence we translated the anthropogenic mortality expressed as an absolute number into a reduction in survival rate.

In detail, in phase one, we determined the stable age distribution (**w**), the reproductive value vector (**v**), the elasticity matrix (e), and the intrinsic growth rate of the deterministic matrix (λ_d; Caswell 2001). Elasticity analysis of matrix projection models examines the effects of proportional changes in demographic transitions (survival, growth, and reproductive parameters) on the asymptotic population growth rate λ (Heppell et al. 2000). Elasticities are useful in identifying vital rates that have large impacts on population growth rates and have thus been used to evaluate the relative importance that each parameters has for management and research. We calculated the initial population vector (**n**) by distributing the estimated female breeding population according to the stable age distribution (**w**) and based on the known proportion of breeders (Pb_a) in each age class. The number of non-breeders were calculated once the number of breeders in each age-class was assigned.

These calculations resulted in the base values used for comparison during phase three. We also used the Heyde-Cohen equation to calculate the growth rate (with 95% confidence intervals) for the stochastic matrix (λ_s), based on a 20 year projection period (Heyde and Cohen 1985), and compared our model to observed values for λ. A 20 year projection time was chosen because we considered a relevant time frame for management purposes, and because we did not want to consider density dependent processes that may apply with a long-term projection and population trends.

In phase two, using **n** as the initial vector, we projected the stochastic population matrix over 20 years in half-year stages (Fig. 8.2). The first stage was the breeding season to the middle of winter, and second stage was the middle of winter to pre-breeding phase at the colony. During stage one, young were produced, and individuals from each age class were removed based on the square root of observed annual survival rates ($\sqrt{S}$). Between stages, birds dying from stochastic, but density-independent, age-class specific, hunt and oil mortality were explicitly removed from the population. During stage two, this population was further reduced based on the square root of observed annual survival rates to model winter and spring mortalities. As described above, this model killed the same birds twice: once explicitly between stages due to anthropogenic causes, and once implicitly throughout the projection by using observed survival rates of a population that is already impacted by anthropogenic mortality. The overall growth rate after these two projections was calculated by running 10,000 simulations, or until the mean and median stochastic growth rates were equal.

In phase three, we examined the relative impact of anthropogenic mortality on population dynamics by adding different percentages to the juvenile survival rate, to the adult survival rate, and to both survival rates simultaneously, until the growth rate again matched the baseline λ_s determined during phase one. In order to estimate population growth in the absence of the hunt and mortality due to oil, the explicit anthropogenic mortality introduced between stages in phase two was removed, and the stochastic matrix was projected with the increased survival rates. Phases two and three were carried out

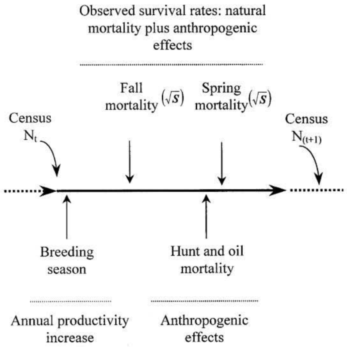

Figure 8.2. Schematic of two-staged population projection during phase two (see methods for details) for Thick-billed Murres breeding in the eastern Canadian Arctic.

examining only mortality due to oil, examining only the hunt, and with hunt and mortality due to oil combined. Relative decreases in potential population growth caused by each mortality factor were calculated, as were cumulative effects of both impacts.

8.3 RESULTS

8.3.1 Baseline values

The intrinsic growth rate of the deterministic matrix was $\lambda_d = 1.0102$, while $\lambda_s = 1.0098$ (95 % C.I. 0.9969-1.0226). Assuming stable age structure (**w**), 90% of individuals in the populations are more than one year old, while 55 % are breeding age, taken as five years or older (Eqn 1). Reproductive value vector (**v**, Eqn 2) and initial population vector (**n**, Eqn 3) are shown below. The elasticity matrix (e) showed clearly that proportional changes in survival (>1 year) have the largest influence on population trends (93.7 %), whereas proportional changes in juvenile survival and fecundity had weaker influence on population growth (6.3 %, Eqn 4). Proportional changes in survival rates of breeding age birds (>4 years) contributed 68.8 % of the growth rate.

$$\mathbf{w} = \begin{bmatrix} 0.0992 \\ 0.0894 \\ 0.0805 \\ 0.0725 \\ 0.0653 \\ 0.0588 \\ 0.0530 \\ 0.0477 \\ 0.0430 \\ 0.0387 \\ 0.0349 \\ 0.3169 \end{bmatrix} \quad [1] \qquad \mathbf{v} = \begin{bmatrix} 0.2469 \\ 0.2741 \\ 0.3042 \\ 0.3375 \\ 0.3694 \\ 0.3938 \\ 0.4070 \\ 0.4234 \\ 0.4327 \\ 0.4444 \\ 0.4533 \\ 0.4583 \end{bmatrix} \quad [2] \qquad \mathbf{n} = \begin{bmatrix} 296330 \\ 266930 \\ 240460 \\ 216600 \\ 195120 \\ 175760 \\ 158330 \\ 142630 \\ 128480 \\ 115730 \\ 104250 \\ 946780 \end{bmatrix} \quad [3]$$

$$\mathrm{e} = \begin{bmatrix} 0 & 0 & 0.000 & 0.001 & 0.002 & 0.004 & 0.003 & 0.004 & 0.004 & 0.004 & 0.004 & 0.037 \\ 0.063 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0.063 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0.063 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0.062 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0.059 & 0 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0.055 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0.052 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0.048 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0.046 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0.041 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0 & 0.037 & 0.350 \end{bmatrix} \quad [4]$$

8.3.2 Mortality and survival rates

Proportional increases in survival rates, which were required to offset the explicitly-added mortality in order to return to the baseline level of λ_s, were greatest for juveniles (Table 8.2). Increases for adults, and for adults and juveniles combined, were almost equal. The

Table 8.2. Stochastic growth rates for three different models projected over 20 years. Model descriptions refer to explicitly added anthropogenic mortalities. Survival rates (S) are proportional increases needed to offset added mortality back to baseline levels λ_s = 1.010 (0.997-1.023). S_a=adult survival, S_0=first year survival. Potential population growth is defined as λ_s in the absence of specified additional mortality, but with added survival. Effect is defined as the estimated absolute effect of the specified anthropogenic mortality on the potential population growth rate. Where applicable, 95% confidence intervals are given in parentheses.

Explicit mortality model	**% increase in S to compensate**	**Potential population growth (λ_s)**	**Effect of anthropogenic mortality on λ_s**
Oil	S_a: 2.8		
	S_0: 41.0		
	S_a+S_0: 2.6	1.035 (1.009-1.062)	-0.025 (0.012-0.039)
Hunt	S_a: 2.0		
	S_0: 34.0		
	S_a+S_0: 2.0	1.030 (1.004-1.057)	-0.020 (0.007-0.340)
Hunt and oil	S_a: 5.1		
	S_0: 77.5		
	S_a+S_0: 4.7	1.057 (1.030-1.084)	-0.047 (0.033-0.610)

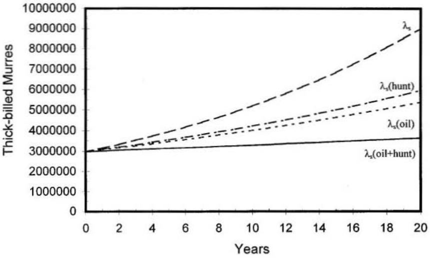

Figure 8.3. Mean stochastic, density independent growth curves during a 20 year projection for Thick-billed Murre populations breeding in the eastern Canadian Arctic affected by both oil and hunt ($\lambda_s(oil+hunt)$), for the same population affected only by harvest ($\lambda_s(hunt)$), affected only by chronic oil pollution ($\lambda_s(oil)$), and in the absence of both harvest and mortality due to oil (λ_s). Population size relates to females only.

8.4 DISCUSSION

Chronic marine oil pollution and murre winter hunt off the Newfoundland coast have a significant effect on Thick-billed Murre populations breeding in the eastern Canadian Arctic. Our study supports the interpretation that events on the common wintering grounds may cause population changes in Thick-billed Murres (Gaston *in review*). Despite the absence of observed population declines, our results indicate that the anthropogenic sources of winter mortality we examined have decreased annual survival rates and have lead to slower population growth. As expected, the cumulative effect of chronic oil pollution and harvest together had the largest impact, lowering adult survival by more than 0.043, and population growth by 4.7 %. As predicted by the elasticity analysis, changes in adult survival, through increased adult mortality, had the largest impact on Thick-billed Murre population growth rates. Although similar numbers may be killed by both sources of mortality, population growth and survival rates are decreased by a correspondingly larger extent by oil pollution, because it affects age classes according to their abundance, while the hunt preferentially impacts juveniles, whose survival has less importance than adults in determining population growth.

8.4.1 Model assumptions

Recent surveys at colonies in the Canadian Arctic indicate that Thick-billed Murre populations are stable to moderately growing, with average growth rates between 0.4 - 2.1 % (Gaston 2002). Our baseline λ of 1.01 falls in the mid-range of these observed conditions. The models which estimate population impacts and potential population growth assume that 1) no density dependent changes occur in demographic parameters, 2) mortality is additive, 3) hunt and oil mortality are density independent, and 4) age-specific mortality in the hunt, and non age-specific vulnerability from mortality due to oil.

Thick-billed Murre colonies in the Canadian Arctic declined between the 1950s and 1970s (Evans and Nettleship 1985, Gaston 1988). Records of annual influxes of first-year Thick-billed Murres to the Great Lakes and northeastern United States common during the early part of the 20th century, have not occurred since 1952 (Gaston 1988). Despite overall population increases since the 1980s, a return to pre-1950 population levels has not occurred (Gaston *in review*). These observations support the assumption that Thick-billed Murre populations are not currently at carrying capacity, and that density-dependent process are not at play, at least not in the short-term,.

Anthropogenic causes of mortality must be, at least in part, additive, and not simply compensatory to natural mortality to have an effect on populations (Fox 2000). Compensatory mortality assumes that anthropogenic causes kill the "doomed surplus", the annual proportion of the population that would have died due to natural causes (Singer et al. 1997, Banks 1999, Boyce et al. 1999). In general, it is assumed that all or part of the "doomed-surplus" is killed when density-dependent factors are present, which is most likely to occur when populations are near carrying capacity (Allen et al. 1998, Boyce et al. 1999). As described above, this does not seem to be the case for Thick-billed

Murres. In addition, recent studies provide strong evidence that hunting mortality in geese was additive to natural mortality, because the relatively constant natural mortality of long-lived species in general does not allow compensation (see below, Gauthier et al. 2001). We therefore consider the Thick-billed Murre harvest and the mortality of Thick-billed Murres due to chronic oil pollution, which affects a greater proportion of adults that the hunt and would therefore even less likely be compensated for, to be additive to natural mortality (Boyd 2000).

It is reasonable to assume that at some level, absolute hunt and oil mortality will decrease with decreasing population size, as overlaps between dense seabird concentrations and ship traffic become less frequent, and as it becomes more difficult for hunters to encounter birds. Conversely, mortality due to chronic oil pollution may increase if the current problem of oil dumping from ships remains constant, while murre populations increase. Due to current hunting regulations (Chardine et al. 1999), an increase in hunting mortality does not seem probable. Given current population sizes and trends, it seems most reasonable to assume density independent mortality for the time periods considered in the model.

8.4.2 Murre hunt

Murres have been hunted in Newfoundland and Labrador for centuries, and surveys in the late 1970s and early 1980s, after the introduction of speed boats and modern shotguns, resulted in estimates that between 600,000 - 900,000 birds were killed every winter

(Elliot 1991). Simple population models indicated that those levels of harvest were unsustainable and needed to be reduced by 50 % to maintain a stable population (Chardine 1988). Although no declines at colonies in Canada had been detected at the time (Gaston 1999), restrictive hunting regulations (shorter season, new daily bag limit) were introduced in 1993 that effectively reduced the harvest to its current levels of approximately 250,000 –300,000 per winter season (Chardine et al 1999).

Despite the predictions, it is difficult to detect whether this reduction in the hunt had an effect on populations in the eastern Canadian Arctic. The Thick-billed Murre population at Coats Island was increasing between 1991 and 1993 (Gaston 2002), and continued to increase as these hunting restrictions were implemented. At the same time, unfavourable environmental conditions on Thick-billed Murre wintering grounds between 1989 and 1990 in the form of an abrupt onset of cold water, slowly returned to pre-1989 conditions (Montevecchi and Myers 1996, 1997, Carscadden et al. 2001). However, an improvement in the rate of recruitment of 4- and 5-year-olds was observed at Coats Island after 1996, cohorts which were reared subsequent to the implementations of hunting regulations in 1993 (Gaston 2002). This suggests a positive effect of reduced hunting pressure on birds in their second year.

Gaston (2002) reported similar adult survival rates at Coats Island before and after hunting restriction came into place. However, no banded breeding-age birds were recovered in the hunt during the 1990s, suggesting that older birds now comprise a small proportion of the hunt. If this pattern holds true, we could have overestimated the impact of the hunt on Thick-billed Murre populations by overestimating the impact on older (breeding) age-classes. Sufficient data to measure impacts on adult and juvenile survival rates and recruitment only exists for Coats Island, which comprises only 1% of the population (Gaston 1999). There are also no data to assess whether changes in age structure may have taken place, while leaving population sizes unaffected. Clearly more long-term data are needed from Coats Island, as well as from some of the larger colonies or those colonies located on the edges of the breeding range.

In contrast, there are clear impacts of the murre hunt on populations in Greenland. Most west Greenland colonies declined over the last 50 years by 20 – 100 %, and average adult survival rates there are estimated at only 85 % (Kampp 1991, Evans and Kampp 1991, Christensen 2001), These declines have been attributed to fisheries by-catch (1965-1975), disturbance by boats at the colonies, egging, and primarily to hunting during summer and winter (Evans and Kampp 1991, Kampp 1991, Kampp et al. 1994, Christensen 2001). The relative contributions of these sources of mortality to Thick-billed Murre population dynamics in Greenland have not been investigated, although it is believed that the summer hunt, which takes the highest proportion of Greenlandic breeders, is the major cause (Falk and Durinck 1992). Until 1991, 40% of all winter band recoveries from birds banded in Greenland came from Newfoundland (Kampp 1991). Although these were mostly juveniles and the number may have been inflated by active solicitation for bands in Newfoundland in the 1980s (R. D. Elliot pers comm.), this suggests that mortality of Thick-billed Murres due to the hunting and chronic oil pollution off Newfoundland may have been an important contributing factor to population declines in Greenland.

8.4.3 Oil pollution

Millions of seabirds have died from oil pollution in recent decades, the majority recorded only after large catastrophic spills (Newton 1998). The effects of chronic marine oil pollution on adult and juvenile survival and reproductive success has gained little emphasis in comparison, despite the fact that several studies have indicated that it may be equally or more detrimental to long-term population stability than occasional large spills (Hunt 1987, Burger and Fry 1993, Oka 1999). Although controversies remain (Paine et al. 1996, Piatt 1996, Wiens 1996, Irons et al. 2000), studies following large oil spills have associated observed short-term declines and long-term effects on seabird populations, with direct oil induced mortality (Hope-Jones et al. 1970, Page et al. 1990, Piatt et al. 1991, Burger and Fry 1993, Edwards and White 1999, Irons et al. 2000, Lance et al. 2001), with effects of oil on food sources and habitat (Suchanek 1993, Day et al. 1997, Edwards and White 1999, Irons et al. 2000) and with decreased winter survival rates (Esler et al. 2000).

Chronic oil pollution off the Newfoundland coast is among the highest recorded in the world (chapter 2). The best estimate of annual seabird mortality due to this pollution is equivalent to an *Exxon Valdez*-sized spill every year (chapter 7). Our study estimates that these current levels of mortality have reduced population growth of Thick-billed Murres by 2.5 % (1.3 - 3.9 %). These values are higher than those estimated for Common Murres in Britain (0.9 – 1.5 %) after a winter of severe chronic spills (Baillie and Mead 1982), are comparable to reduced growth rates calculated by Nur et al. (1997) for Common Murres exposed to chronic oil pollution on the Farallon Islands (3 %), and are lower than the estimated impacts of a winter of severe chronic spills on Razorbill populations in Britain (3.4 - 5.6 %, Baillie and Mead 1982).

As predicted, decreased adult survival has the largest effect on population trends, and we estimate the annual adult survival has been reduced by sustained mortality from chronic oil pollution by 2.75 %. This value is higher than calculated for Common Murres (1.7 %) by Nur et al. (1997), but lower than that for female Harlequin Ducks (*Histrionicus histrionicus*, 5.7 %) after continued chronic exposure to oil residues from the *Exxon Valdez* oil spill (Esler et al. 2000), which also seems to be the determinant of the annual 5.4 % decrease in that population.

Our estimates of a potential adult survival rate of 0.933 in the absence of mortality due to chronic oil pollution, or 0.953 in the absence of mortality due to both oil and harvest, is well within survival rates recorded elsewhere for other auks, especially Common Murres, (Table 8.3). However, our estimate of the impacts of chronic oil pollution on this population could be biased for several reasons. Effects could have been over-estimated

Table 8.3. Adult survival rates (S_a) for different auk species in Europe and North America

Species	**Location**	S_a	**Reference**
Common Murre *(Uria aalge)*	Isle of May, Scotland	0.93-0.97	Harris and Bailey 1992, Harris and Wanless 1995
	Skomer, Wales	0.94	Hatchwell and Birkhead 1991
	United Kingdom	0.94	Mead 1974
	Farallon I., California	0.94	Sydemann 1993
	Gannet I., Labrador	0.94	Birkhead et al. 1985
	Great I., Newfoundland	0.98	Wilhelm and Storey (unpubl. data)
Razorbill	Skomer, Wales	0.92	Lloyd and Perrins 1977
(*Alca torda*)	United Kingdom	0.91	Mead 1974
	Quebec, Canada	0.90	Chapdelaine 1997
Atlantic Puffin	Isle of May, Scotland	0.89-0.96	Harris 1983, Harris et al. 1997
(*Fratercula arctica*)	Skomer, Wales	0.95	Ashcroft 1979
Thick-billed Murre	Coats I., Canada	0.90	Gaston (*in review*)
(*Uria lomvia*)	Greenland	0.85	Kampp 1991

if, during an average winter, young birds are more affected than adults than their abundance would suggest. Also, because the fecundity estimates for birds >11 years old of 0.73 (Table 8.1) is generally lower than those measured at High Arctic colonies (0.8, Gaston and Nettleship 1981, Birkhead and Nettleship 1985) we may be underestimating the global productivity of Thick-billed Murres in the eastern Canada Arctic. On the other hand, we could have under-estimated the impact of oil mortality because these absolute mortality estimates are considered conservative (chapter 7), and because no chronic, sub-lethal effects of oil to Thick-billed Murres populations were considered, in areas where prey organisms may also be contaminated (Croxall 1977, Peakall et al. 1981, Culik et al. 1991, Holmes 1984, Leighton 1995). Overall, the estimated impact of these two sources of anthropogenic mortality on Thick-billed Murres appears to be reasonable, and substantial enough to raise conservation concerns.

8.4.4 Conservation implications

Murres have been hunted in Newfoundland and Greenland for centuries and are considered an important food source. Hunting regulations are now in place in Canada which have reduced the mortality by 50 %, and the hunt that kills mostly juveniles. We are thus more concerned about the degree of mortality caused by chronic oil pollution off the Newfoundland coast (chapter 7), and its implications for the long-term population trends of Thick-billed Murres that breed in the eastern Canadian Arctic. The high annual kill due to negligent, illegal behaviour by ship operators is unacceptable in itself.

In addition to oil pollution and hunt, by-catch of murres in commercial fisheries is also a potential threat to Thick-billed Murre populations from the Canadian Arctic, and one that was not considered here. Very large numbers of Thick-billed Murres died in a salmon drift-net fishery in 1960s in western Greenland (Tull et al. 1972, Falk and Durnick 1992,). Common Murres were substantially affected by inshore gill-net fisheries in Newfoundland in the 1970s and 1980s (up to 20 % of the local breeding population drowned in one year, Piatt et al. 1984). Since a groundfish moratorium sharply reduced fishing effort in 1992, seabird by-catch is thought to have plummeted, although a limited fishery was re-opened in Atlantic Canada in 1997 (Chardine 1998). Seabird by-catch is not regularly monitored in Canada. Some data suggest that by-catch mortality also occurs offshore at murre wintering areas (Piatt et al. 1984, Chardine 1998) and other observations indicate that current inshore by-catch incidents are under-reported (Davoren 2001). At present levels of hunting and oil mortality, our model indicates that Thick-billed Murre populations overall are stable or increasing at an average rate of only 1 % per annum. If by-catch or other levels of anthropogenic mortality reach that level of mortality, population increases would most likely stop or change to declines. We concur with Chardine (1998), that seabird by-catch monitoring should become an integral part of fisheries management activities in Canadian waters.

Compensatory mechanisms exist that could dampen the effects of anthropogenic mortality on populations, such as increased reproductive success, decreased natural mortality, and decreased age of first breeding. It is not clear whether such mechanisms could be at work in Thick-billed Murre populations (Gaston et al. 1994), but as mentioned long-lived seabird species have limited compensatory capabilities (Hellgren et al. 1995, Boyce et al. 1999). This strongly suggests that anthropogenic mortality such as that from hunting and oil is additive (Gauthier et al. 2001). If compensatory mechanisms do operate to some degree in Thick-billed Murre populations, we could have overestimated the overall impacts of anthropogenic mortality. However, compensatory mechanisms are generally only effective to counter-balance temporary disturbances, and rarely effective enough to withstand high sustained increases in adult and juvenile mortality (Weimerskirch et al. 1997). Additional work is needed to determine the presence and plasticity of compensatory mechanisms in Thick-billed Murre populations.

8.5 CONCLUSIONS

We present a model that allows the quantification of effects on seabird populations of sources of anthropogenic mortality. The model is especially useful in the absence of detectable population declines or where logistical considerations do not allow sufficient data collection to measure potential impacts. This is especially critical in light of natural variation in demographic parameters, and confounding environmental impacts that may make it difficult to detect populations changes in the field, or to attribute them to a specific cause (Eppley 1992). In essence, however, such models cannot be built in the absence of data from long-term demographic monitoring studies, such as that ongoing at Coats Island since 1984.

There is no doubt that sustained adult mortality inflicted by human activities can cause severe population declines in seabirds (Moloney et al. 1994, Weimerskirch et al. 1997, Tasker et al. 2000, Tuck et al. 2001, Ainley et al. 2001), and that these sustained impacts are generally more detrimental to populations in the long-term than single large kills. Even some very abundant bird species have not escaped these impacts (e.g. Great Auk *Pinguinus impennis*, Montevecchi and Kirk 1996; Passenger Pigeon *Ectopistes migratorius*), and unsustainable harvest levels are known to have caused substantial declines of Thick-billed Murre populations in Greenland. Although current Thick-billed Murre populations in eastern Canada seem stable (Gaston 2002), some stochastic projections predict declining populations. Declines observed in Thick-billed Murre populations on Coats Island during the late 1980s (Gaston 2002, and similar examples elsewhere (Duffy 1983), show that the coincidence of unfavourable environmental conditions with large sustained anthropogenic impacts can easily cause population declines, even when species are abundant. At the very least, decreased population growth, even when no declines occur, causes an increased vulnerability in these populations to changes in their environment and other pulse perturbations (Dunnet 1982, Ford et al. 1982, Takekawa et al. 1990). Our model points out these sensitivities.

Nevertheless, long-term monitoring of colonies in the eastern Canadian Arctic continues to be necessary to assess these changes, and we recommend that monitoring should be expanded to include some of the major Thick-billed Murre colonies in the eastern Canadian Arctic such as Akpatok Island. It would also be valuable to increase efforts to assess the age of oiled beached birds whenever possible, to more accurately assess the impact of chronic oil pollution on populations. Finally, we strongly recommend that actions be taken to reduce seabird mortality due to chronic oil pollution (Wiese 2002, section 9.5), before Thick-billed Murre populations that breed in the eastern Canadian Arctic follow the pattern of their declining Greenlandic counterparts.

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CHAPTER 9 – GENERAL DISCUSSION

In this thesis, I examined levels of chronic oil pollution off the southeast coast of Newfoundland, as assessed through beached bird surveys from 1984-1999. In addition, results from experiments designed to determine the origin and fate of oiled and unoiled birds at sea and on beaches generated a novel Oiled Seabird Mortality Model, that estimated the number of seabirds killed annually by chronic oil pollution within a defined area of ocean. Finally, I constructed a sophisticated population model to determine the impacts of mortality due to oil pollution and hunting on Thick-billed Murre populations. This species is the most affected by chronic oil pollution in Newfoundland, and the target of a traditional winter hunt.

The following sections synthesize results and conclusions from chapters 2, 7 and 8, and is followed by a list of future research needs.

9.1 INDEX OF CHRONIC OIL POLLUTION

Levels of chronic oil pollution in Newfoundland are among the highest recorded in the world. Sixty percent of birds found on beaches over the last 16 years, and 74 % of birds found over the last five years perished due to oil. Although it is to be expected that birds found in northern latitudes and colder oceans have high oiling rates, because thermal stress after oiling is high (Hartung 1967, Jenssen et al. 1985, Culik et al. 1991, Doerffer 1992), Newfoundland's overall oiling rates are disproportionately high. As found in studies elsewhere (Camphuysen and Heubeck 2001), auks (murres, dovekies) and diving ducks (mostly eiders) are the avian groups most affected by oiling in Newfoundland, likely because they spend most of their time on or under water, and dive rather than fly as an escape response. The much higher number of murres compared to all other species certainly reflects their high abundance during the winter months and influences the overall oiling rates. But the overall species-specific oiling rate for murres of 72 %, (80 % during the last 5 years), also ranks among the highest in the world (3-82 %, Camphuysen 1998), and clearly reflects regional differences in oiling risk.

In addition to the high proportion of oiled birds found in Newfoundland, the average density of oiled birds on studied beaches during winter was also markedly higher than in other regions of the world during a comparable period (Burger 1993, Vaitakus et al. 1994, Heubeck 1995) and has remained at these high levels for the last 16 years. As in the North Sea (Camphuysen and Heubeck 2001), oiling rates in Newfoundland are significantly higher in winter than summer (Wiese and Ryan 1999). Yet given the strong relationship between weather and oiling rates, it is doubtful that chronic oil pollution only occurs in the winter. Rather, lower oiling rates in the summer could reflect 1) reduced abundance of highly vulnerable birds such as Thick-billed Murres, dovekies and eiders, 2) reduced overlap of nesting Common Murres with shipping lanes (Lock et al. 1994), 3) reduced susceptibility of seabirds to oil in warmer ambient and sea surface temperatures,

4) faster evaporation of refined oils at higher sea surface temperatures, and 5) faster loss of volatile and soluble toxic components of crude oils components, and 6) faster reduction of these oils to biologically inert solids such as tar balls, at higher sea surface temperatures (Bourne and Bibby 1975, Kennish 1997). Observations of body conditions and the degree of oiling of beached birds furthers strengthens this interpretation. Small quantities of oil on a carcass, combined with high emaciation, indicates that the bird lived for a while after contacting the oil before it likely perished from hypothermia and starvation during harsh weather (Camphuysen and Franeker 1992).

9.2 MORTALITY CAUSED BY CHRONIC MARINE OIL POLLUTION

To determine total mortality of seabirds due to chronic oil pollution, several parameters related to the fate of birds at sea and on the beach need to be known. First, I determined that carcasses persisted on average for only 3.3 ± 0.1 days on beaches in southeastern Newfoundland (chapter 3); after which they become undetectable because they are scavenged or buried in the beach substrate. In addition, no differences were found in persistence rates between oiled and unoiled birds. I also determined deposition rates and detection probabilities of bird carcasses, and developed a model which allowed an accurate estimation of the number of birds arriving on a beach between periodic surveys. This model only performs well if survey intervals are kept below 10 days. Second, I designed a drift block that accurately mimics a seabird carcass drifting at sea (chapter 4).

Drift blocks commonly used in past studies were shown to not drift like seabird carcasses because they were overly influenced by wind. A more realistic drift block was needed to accurately interpret the number of birds that are found dead. Third, I measured murre carcass sinking rates and found that intact birds only float 8.2 ± 5.2 days before sinking (chapter 5), but that scavenging is important. Fourth, to determine the proportion of birds that die at sea and reach the shore, I carried out extensive drift block experiments using the new block design (chapter 6). Recovery rates of blocks dropped from various locations varied, and the best predictor for the proportion of blocks lost at sea was the distance from shore where they were dropped, combined with the cumulative wind direction for the first three days after the blocks were set adrift. Based on wind patterns observed during the experiment, I was able to estimate wind specific recovery rates and catchment areas for birds that die at sea.

A framework for a novel Oiled Seabird Mortality Model was built and applied to Newfoundland based on results from beached bird persistence studies, buoyancy loss experiments, and drift block recovery trials (chapter 7). This model uses wind-specific extrapolation factors to assess the number of birds that arrive onshore, and was used to estimate the number of seabird killed due to chronic oil pollution that occurred over a survey season, and within a defined area of ocean. The best estimate indicated that on average, about 300,000 seabirds are killed each year in the waters of Atlantic Canada due to illegal activities of ship operators; a yearly seabird mortality impact equal to *the Exxon Valdez* disaster in Alaska in 1989 (Ford et al. 1996). A detailed examination of the model assumptions and perturbation analysis revealed several key factors that need further investigation to increase confidence in several parameter estimates (chapter 7, see section 9.4 below).

The framework for this Oiled Seabird Mortality Model is applicable to any area where the required geographically-specific information is available. Given that the presence of chronic oil pollution is already well documented in many parts of the world (Camphuysen and Heubeck 2001), I hope comparable analysis can take place.

9.3 IMPACTS ON POPULATIONS

Breeding populations of murres vary naturally and depend mostly on a variety of environmental conditions encountered on the winter grounds (Gaston *in review*). In face of these natural fluctuations and given the presence of non-breeding 'floater' populations, population declines can be difficult to detect. When detectable, however, such declines are usually quite severe and immediate mitigative actions are required. However, where colonies are monitored intensively, and where modern modeling techniques are employed, it could now be possible to have earlier indications of population impacts prior to detectable population declines. One such sign is a partial or total reduction in the rate population growth or the slow depletion of the 'floater' population. Under such conditions, a population may appear stable with no apparent effects caused by anthropogenic activities, when indeed the impact could be quite substantial.

The species most affected by spilled oil along the south coast of Newfoundland, both in numbers (>60 %) and in the proportion oiled (>80 %), were Thick-billed Murres. This species is characterized by late sexual maturity, by low fecundity (laying only one egg per year), by a long life span of up to 30 years, and by high adult survival. About 80 % of the Thick-billed Murres that overwinter in the waters of Atlantic Canada, breed in the eastern Canadian Arctic, and most of the remainder breed in Greenland (Tuck 1961, Gaston 1980, Kampp 1988, Donaldson et al.1997). Numbers breeding in most West Greenland colonies have declined during the past 50 years by 20 –100 %, and average adult survival rates there are estimated to be only 85% (Kampp 1991, Evans and Kampp 1991, Christensen 2001), These declines have been attributed to fisheries by-catch (1965-1975), disturbance by boats at colonies, egging, and hunting during summer and winter (Evans and Kampp 1991, Kampp 1991, Kampp et al. 1994, Christensen 2001). The relative contributions of these mortalities on Thick-billed Murre population dynamics in Greenland have not been investigated, although it is believed that the summer harvest is the major cause (Falk and Durinck 1992).

Populations from the eastern Canadian Arctic are subject to heavy winter mortality due to oil and hunting. In addition to the above calculated mortality due to oil, there is a traditional winter murre hunt in Newfoundland, where between 130,000 – 210,000 Thick-billed Murres from Canadian colonies are killed annually (Elliot 1991, Chardine et al. 1999).

Thick-billed Murre populations in the Canadian Arctic have increased since the 1980s but appear stable since about 1997 (Gaston 2002). Based on demographic information collected at Coats Island, Hudson Bay over the last 15 years, we constructed a stochastic (demographic and mortality), age-based, density-independent, pre-breeding projection matrix population model for Thick-billed Murres breeding in the eastern Canadian Arctic. The model matched the observed trends that Thick-billed Murre populations are stable or increasing slightly, but is also showed that these populations have a yearly potential for growth of about 5.7 % in the absence of oil and hunting mortality. Due to the sustained adult and juvenile annual mortality resulting from illegal discharges of oil at sea, and to a lesser degree from the murre hunt, Thick-billed Murre populations are now extremely sensitive to other changes in their ecosystem (such as changes in food availability resulting from regime shifts in the ocean, climate change) as they have no "growth buffer" from which to draw on, and possibly a decreased 'floater' population to compensate for higher than normal mortality of breeders, to avoid declines. If these high levels of anthropogenic mortality continue or worsen, and if these circumstances were to coincide with periods of unfavorable environmental conditions for murres and their prey (more ice, colder water), Thick-billed Murre populations in the eastern Canadian Arctic could decline steeply as have their Greenland counterparts.

9.4 FUTURE RESEARCH NEEDS

I identified several future research needs to increase our understanding of the behaviour of oiled seabird at sea, to further increase our confidence in the seabird mortality estimate due to chronic oil pollution off the coast of Newfoundland, to collect additional evidence on effects of chronic oil pollution on other parts of the Grand Bank marine ecosystem, and to be able to detect potential impacts on Thick-billed Murre populations.

Despite the knowledge gained trough this research, we still do not have a good understanding about the behaviour of birds in relation to oil slicks, or how birds, once oiled, behave. Experiments should thus be designed to investigate the possible attraction of seabird to oil slicks based on visual and olfactory mechanisms (Bourne 1968, Kerley et al. 1987, Nevitt et al. 1995, Nevitt 1999). Observations regarding the behaviour of oiled birds at sea are needed. Do oiled birds fly to shore as soon as possible or do they remain at sea as long as they can as levels of emaciation and quick death after arriving onshore suggest? Does this depend on the level of oiling? If birds die at sea, are they scavenged or do they remain intact until they come close to shore where density of gulls may be higher, and do different species sink at different rates? Observations collected during large oil indicate that there is a strong relationship between the number of dead and live oiled birds found and a ratio of 85:15 seems common (Ford et al. 1996, R. G. Ford, pers. comm.). In addition, experiment conducted in Alaska suggest that sinking rates of small and large alcids may be comparable, and hence current findings may also be applicable to dovekies (Ford et al. 1996).

Furthermore, I have not investigated sub-lethal effects of oil pollution on birds or other marine organisms. How high are the levels of chronic oil pollution in the area? Apart from the index created from beached birds, there are other indices that could be examined to determine impacts of other marine organisms and bioaccumulation, such as levels of oil degrading bacteria at sea (Rosenberg 1992), Polycyclic Aromatic Hydrocarbon (PAH) concentration in sessile invertebrates along the shore (Jones et al. 1998), and concentrations of fossil fuel hydrocarbons in the gut contents of Procellariiformes (Boersma 1986).

There is also a clear need for more detailed seabird-at-sea distribution data to better identify regions of increased seabird vulnerability to oil pollution, especially in the winter. Existent information is sparse and dates back more than 20 years (Lock et al. 1994), during which many oceanographic processes have fluctuated (Davoren 2001).

To further increase our confidence in seabird mortality estimations, more drift block experiments need to be carried out, especially in the Placentia Bay area. These should be laid out in a grid pattern inside and outside the bay and performed during several different wind regimes, so that estimated catchment areas and recovery rates can be determined more precisely and integrated into OSMM. Also, as mentioned, it would be useful to determine if sinking rates for other species commonly found during beached bird surveys such as Dovekies, as indeed comparable to murres. As well, it is necessary to further explore differences in sinking rates between birds affected by different amounts of oil.

In addition, beached bird surveys in Newfoundland should continue on a periodic basis, ideally with survey intervals below 10 days so that overall seabird mortality estimates can continue, and so that the effectiveness of mitigative strategies can be assessed. In this respect, detailed deposition studies would be insightful to quantify how representative surveyed beaches are for the overall region. Surveys should hence be expanded geographically throughout Atlantic Canada to better determine the overall impact for the region. During all beached bird surveys, carcasses should be aged whenever possible to determine whether there is an age-specific mortality due to oil.

It is critical that long-term seabird monitoring in the Eastern Canadian Arctic continue, and include some of the large colonies of Thick-billed Murres. Although modeling can reveal vulnerabilities and relative effects of anthropogenic mortalities on populations, much remains to be learned of possible compensatory mechanisms and resultant changes in demographic parameters. Clearly this can only be assessed through long-term monitoring schemes. In this regard it is critical to investigate the origin of Thick-billed Murres over-wintering off the coast of Newfoundland more closely. This could be achieved through satellite telemetry. Finally, the potential impacts of oil pollution on other species should not be ignored. Although Thick-billed Murres were the most common victims, the kill of Common Murres or Dovekies relative to their population size may imply potential impacts that should be investigated further through population modeling.

9.5 CONCLUSIONS

The lack of sufficient surveillance and enforcement in Atlantic Canada encourages careless and/or deliberate dumping of oil and oily wastes (CCG 2002, Wiese 2002). Ships are able to flout national legislation and international conventions while dumping toxic hydrocarbons into Canadian waters almost at will. An estimated 300,000 seabirds are killed on average each year in Atlantic Canada, the equivalent of the *Exxon Valdez* kill in 1989. Local mortality caused by careless ship operators have impacted Thick-billed Murre populations from the Canadian Arctic, and similar effects may be felt in places as far away as Greenland. The need for action in this region is Canada's international responsibility.

Enforcement agencies have several tools in hand to ensure compliance, many of which have proven powerful and effective in other parts of the world (CCG 2002, Wiese 2002). In the Netherlands, for example, oiling rates were significantly reduced after aerial surveillance and imposed fines for polluters were increased (Camphuysen 1998).

Given the high levels of chronic oil pollution in Atlantic Canada and its impacts on Thick-billed Murres, aerial surveillance and ship inspections regimes need to be intensified. To be an economic deterrent, fines imposed under the current legal regime need to be increased, and to be representative of the severity of the impact be a multiple of the costs expected to be saved by the polluter, incurred by investigating agencies, and of damages caused to the environment. A minimum rather than a maximum fine regime should be considered. Environmental Damage Assessment should become more common, even for small chronic spills, and research is needed to increase assessment capabilities. Convenient and accessible on-land oil-disposal facilities should be used and established, vessel monitoring, enforcement efforts should increase, and education and awareness programs should be launched among the public, industry and government (CCG 2002, Wiese 2002).

Many of the facts surrounding the problem of chronic oil pollution are unknown to most Canadians. It is important to stress that this damage is not necessary, and that important steps have been identified to reduce these illegal activities. The attention that chronic oil pollution has garnered by the media, government, industry and the scientific community has been small in comparison to that for large catastrophic spills. It is imperative that more efforts are diverted into the prevention of chronic oil pollution events that collectively have clear detrimental impacts on our marine ecosystems.

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APPENDIX 1. Winter Oil Vulnerability Indices

Winter Oil Vulnerability Indices (WOVI) for the species most commonly found on beached bird surveys in southeastern Newfoundland. Following methods suggested by King and Sanger 1979 and adapted from Camphuysen 1998. Maximum score per criteria is 5, overall maxima is 40.

AOU species codes	**Behaviour**					**Range**			
	Roosting	Foraging	Escape	Flocking	Exposure	Migration	Wintering	Marine orientation	**WOVI**
GBBG	3	3	3	3	2	1	1	3	**19**
HEGU	3	3	3	3	2	1	1	3	**19**
NOFU	5	3	1	3	1	1	1	5	**20**
NOGA	5	3	3	3	1	1	1	5	**22**
LTDU	3	5	1	5	2	3	3	1	**23**
ATPU	5	5	5	3	1	1	1	5	**26**
BLGI	5	5	5	3	2	1	3	3	**27**
COEI	5	5	1	5	4	3	3	5	**31**
COMU	5	5	5	3	5	5	1	5	**34**
DOVE	5	5	5	3	5	5	1	5	**34**
MUSP	5	5	5	3	5	5	1	5	**34**
TBMU	5	5	5	3	5	5	1	5	**34**