Evaluation Report for the 5-Year Status Review of the Marbled Murrelet in Washington, Oregon, and California

Prepared for:

U.S. Fish and Wildlife Service, Region 1

March 2004

Contract No: 101813C046

Prepared by:

C. McShane, T. Hamer, H. Carter, G. Swartzman, V. Friesen, D. Ainley, R. Tressler, K. Nelson, A. Burger, L. Spear, T. Mohagen, R. Martin, L. Henkel, K. Prindle, C. Strong, & J. Keany

EDAW

HAMER ENVIRONMENTAL

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Cover photographs by T. Hamer (tree climber, forest stand, nest, egg, and chick) and R. MacIntosh (murrelet at-sea)

EXECUTIVE SUMMARY

The marbled murrelet (Brachyramphus marmoratus) is a small diving seabird that nests mainly in coniferous forests generally within 30 miles (50 km) of the coast and forages in near-shore marine habitats. Its range includes southern Alaska (including the Aleutian Islands, northern Gulf of Alaska, and Southeast Alaska regions), British Columbia, Washington, Oregon, and California. In 1992, the Oregon, Washington, and California population of this species was listed as threatened under the Endangered Species Act (ESA) due to the loss of nesting habitat from logging and urbanization, as well as mortality associated with gill-net fisheries and oil pollution. In 2002, the U.S. Fish and Wildlife Service (Service) was sued, in part over failure to conduct a 5-year status review of the marbled murrelet, as required by the ESA. In 2003 the Service agreed to conduct the status review, which consists of a review of available scientific information plus a regulatory review.

To conduct the scientific review, the Service sought proposals from qualified groups and awarded a contract to EDAW, Inc. (Contract 101813C046) on September 29, 2003. EDAW, with the assistance of Hamer Environmental, assembled a panel of experts to review available scientific information obtained since the marbled murrelet was listed, including documents received from 2 Federal Register requests by the Service in 2003. Working over a 5-month period (October 2003- February 2004), the panel reviewed over 500 documents and prepared this Evaluation Report which summarizes, evaluates, and interprets the biological, ecological, and population information on the marbled murrelet. The report also provides an evaluation of current threats to the species (excluding inadequate regulatory mechanisms) and how these threats may have changed since the listing. Information provided in the Evaluation Report will be used during a separate regulatory review of the murrelet listing status that is being conducted by the Service.

Panelists were assigned various topic areas associated with their expertise and instructed to review all relevant research studies on marbled murrelets. Where information specific to the marbled murrelet was lacking or inadequate, the panelists referred to data from studies on other seabirds, if possible, with clarification on applicability to the marbled murrelet. In general, information on murrelet breeding biology, population size, terrestrial habitat use, and marine habitat use is the most comprehensive, allowing many conclusions to be drawn with a high degree of certainty. Reproductive success, diet, and variation in prey resources are much less well known and more difficult to interpret, requiring greater use of professional judgment to

assess murrelet and other seabird studies within and outside the listed range. Genetics data are available for the Aleutian Islands, Alaska, and California but not for Oregon or Washington, creating some uncertainty in how boundaries between subpopulations should be defined. To estimate future population trends and extinction probabilities, several panelists developed a new demographic model. Future population projections have an inherent degree of uncertainty due to the model's assumptions and the obviously insufficient knowledge of future conditions. However, rates of decline were similar to more simplistic models with fewer assumptions. This new model served as the best possible method of integrating and evaluating available information on current and future population status and trends. The panelists met twice over the course of the review, communicated frequently, and often challenged each other to provide more information, logic, and rationale. Although differing opinions were expressed on certain topics, the content and conclusions presented in this Evaluation Report are supported by the entire expert panel.

The world population size of marbled murrelets is recently estimated at 947,500 birds, with 91% in Alaska, 7% in British Columbia, and 2% in the listed range. While murrelets within the listed range make up only a small fraction of current world population size, this area represents 18% of the linear range of the species and probably supported greater populations historically. Available data on genetic and ecological differences suggest at least 3 primary populations of the species: (1) Aleutian Islands; (2) Alaska Peninsula to Puget Sound; and (3) western Washington to California. Demographic modeling suggests that the population within the listed range will decline over the next 40 years, with largest relative declines in California. While a major decline in near-shore densities of murrelets has been validated only in Oregon since 1992, only very small populations of poorly reproducing birds currently occur in central California (Mendocino and San Mateo/Santa Cruz counties), where local extinction maybe a concern. Population decline within the listed range appears related primarily to the effects of historic and ongoing loss of breeding habitat in old-growth forests, combined with poor reproductive success from relatively high levels of corvid nest predation in remaining forest patches, especially those near human settlements. However, the annual rate of breeding habitat loss and loss of occupied sites due to survey error has been reduced since 1992. Mortality from oil pollution has continued, but mortality from gill-net fishing has been reduced in Washington and eliminated in California. While the murrelet population has continued to decline, rates of decline have likely been reduced since 1992, which greatly benefit the species in the short term. From the available information, long-term survival of the marbled murrelet in Washington, Oregon and California is not certain.

Preparers

This report was prepared primarily by a panel of scientists with expertise in marine ecology, terrestrial ecology, seabird biology, genetics, and seabird population modeling, with contributions from several other scientists. The overall scientific review process, including team coordination, consultation with the Service, and document production, was managed by EDAW, with assistance from Hamer Environmental. Authors and their project roles are listed in the table below.

Name	Affiliation	Role
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Other Contributors

Preparation of this report would not have been possible without the contributions and support of the following staff from EDAW, Inc.:

- Peter Carr – Production editor and overall document coordination
- Liza MacKinnon – Graphics, web design and coordination, and Administrative Record maintenance
- Emily Inkpen – Word processing
- Chris Stoll – GIS support

U.S. Fish and Wildlife Service

Preparation of this report was greatly facilitated by staff from the Service (Region 1). Paul Phifer, Ph.D., Brian Cox, and Barry Mulder acted as Service project managers and provided project direction, support, and review. Lee Folliard, Deanna Lynch, and Lynn Roberts obtained and summarized available data on estimated amounts of suitable and occupied murrelet habitat, and reviewed sections of this report. Other Service reviewers included: Don Compton, Gary Falxa, John Grettenberger, and Vince Hawke. The panelists and contributing authors wish to express their appreciation to the Service staff for their prompt critical review and assistance.

External Reviewers

Several sections of the Evaluation Report were reviewed by outside experts. Their review and comment helped clarify and strengthen these sections and greatly contributed to the quality of the report. The panelists and contributing authors wish to express their appreciation to the following individuals for their timely review and constructive comments:

Reviewer	Affiliation	Sections Reviewed
James Baldwin (M.S.)	Pacific Southwest Research Station, U.S. Forest Service Albany, CA USA	Survey effectiveness and error (Sections 4.2.2 and 4.2.3)
George Barrowclough (Ph.D.)	American Museum of Natural History Washington D.C. USA	Morphological and genetic variation and genetic threats (Sections 3.4 and 3.6)
Steve Beissinger (Ph.D.)	University of California, Berkeley, CA USA	Demographic modeling (Section 3.5)
Glenn Ford (Ph.D.)	Independent Consultant Portland, OR USA	Effects of oil spills and demographic modeling (Sections 5.4.2 and 3.5)

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ACRONYMS AND ABBREVIATIONS

ADFG	Alaska Department of Fish and Game
AIC	Akaike's Information Criterion
AOU	American Ornithological Union
AWA	Administratively Withdrawn Areas
BA	Biological Assessment
BC	British Columbia
BEC	Biogeoclimatic Ecosystem Classification
BLM	Bureau of Land Management
BO	Biological Opinion
bp	base pair
CA	California
CDC	Center for Disease Control
CDF	Coastal Douglas-Fir
CDFG	California Department of Fish and Game
CESA	California Endangered Species Act
CHU	Critical Habitat Unit
CI	Confidence Interval
COASST	Coastal Observation and Seabird Survey Team
CWA	Congressionally Withdrawn Area
CWH	Coastal Western Hemlock
CZ	Conservation Zone
dbh	diameter at breast height
EM	Effectiveness Monitoring
ESA	Endangered Species Act
ESU	Evolutionarily Significant Unit
FEMAT	Forest Ecosystem Management Team
FR	Federal Register
GIS	geographical information system
GPNF	Gifford Pinchot National Forest
HAG	Harmful Algal Blooms
HCP	Habitat Conservation Plan
kJ	kilojoule
LIMBS	landscape-level individual murrelet based simulation
LSR	Late Successional Reserves
MBF	Million board feet
MH	Mountain Hemlock
MMCA	Marbled Murrelet Conservation Area
MOF	Ministry of Forestry
mtDNA	mitochondrial DNA
MU	management unit
NF	National Forest
NHP	National Historic Park
NM	National Monument
NP	National Park
NPS	National Park Service

ACRONYMS AND ABBREVIATIONS (cont.)

NRA	National Recreation Area
NWFP	Northwest Forest Plan
NWHC	National Wildlife Health Center
ODF	Oregon Department of Forestry
ODFW	Oregon Department of Fish and Wildlife
OR	Oregon
PALCO	Pacific Lumber Company
PCB	polychlorinated biphenyls
PCDD	polychlorierte dibenzo-dioxine
PCDF	polychloro-dibenzo-furannes
PDO	Pacific Decadal Oscillation
PSG	Pacific Seabird Group
PVA	Population Viability Analysis
RFP	Request for Proposal
SD	standard deviation
SEIS	Supplemental Environmental Impact STatement
SEPA	State Environmental Policy Act
Service	U.S. Fish and Wildlife Service
TFW	Timber, Fish, and Wildlife
USFS	U.S. Forest Service
USGS	U.S. Geological Survey
WA	Washington
WDFW	Washington Department of Fish and Wildlife
WDNR	Washington Department of Natural Resources
WFPA	Washington Forest Protection Association
WNV	West Nile Virus

1.0 INTRODUCTION

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1.1 Background

The marbled murrelet (*Brachyramphus marmoratus*) is a small seabird that nests in coastal coniferous forests and forages in near-shore marine habitats along the Pacific coast of North America. In September 1992, the Oregon, Washington, and California population of this species was listed as threatened under the Endangered Species Act (ESA) of 1973, as amended (57 Federal Register [FR] 45328). The listing decision was based on the determination that the marbled murrelet was threatened from: (1) loss and modification of nesting habitat (older forests), primarily due to commercial timber harvesting; (2) mortality associated with gill-net fisheries off the Washington coast; and (3) mortality resulting from oil pollution. A recovery team was formed in 1993, with a recovery plan produced in 1997 (Department of the Interior, U.S. Fish and Wildlife Service [Service] 1997).

In 2002, the Service was sued by the American Forest Resources Council and others (*American Forest Resources Council, et al. vs. Secretary of the Interior*) over failure to comply with Section 4(c)(2) of the ESA relative to the marbled murrelet. This section of the ESA requires a 5-year review, based upon the "best available scientific and commercial information," to determine whether a change in listing status is warranted. The Secretary of Interior is ultimately responsible for conducting the 5-year reviews of listed species, but this responsibility has been stepped down to the Service for species under their jurisdiction. On January 13 and 14, 2003 (as amended, June 30, 2003), the Service agreed to conduct a 5-Year Status Review for the marbled murrelet in connection with settlement of the lawsuit.

The Service initiated the review process for the marbled murrelet in April 2003 by issuing a request for the best available scientific and commercial information on the species since its original listing in 1992 (68 FR 19569, 21 April 2003). A second request was released in July 2003 (68 FR 44093, 25 July 2003). Data were requested on the following topics:

- Species biology, including but not limited to population trends, distribution, abundance, demographics, and genetics;
- Habitat conditions, including but not limited to amount, distribution, and suitability;

- Conservation measures that have been implemented that benefit the species;
- Threat status and trends; and
- Other new information, data, or corrections, including but not limited to taxonomic or nomenclatural changes or improved analytical methods.

In response to these 2 requests, the Service in Portland received more than 450 documents, reports, datasets, and comment letters. These materials form the basis of the Administrative Record for the 5-Year Status Review and are available as public information upon request. The Service has created a database of all the available information in the Administrative Record.

In July 2003, the Service decided to seek contractor assistance in gathering and synthesizing information for the 5-Year Status Review of the marbled murrelet and issued a Request for Proposal (RFP). The contract was awarded to EDAW, Inc., an environmental consulting firm with an office in Seattle, Washington, on September 29, 2003.

In the RFP for the 5-Year Status Review, the Service asked the potential contractors to identify and organize a panel of scientific experts. The panelists assembled by EDAW, with the assistance of subconsultant Hamer Environmental, L.P., include the following scientists with expertise in seabird biology, genetics, and marine and forest ecology:

- Dr. David Ainley, H.T. Harvey and Associates; San Jose, California
- Dr. Alan Burger, University of Victoria; Victoria, British Columbia
- Mr. Harry Carter, Independent Consultant; Richmond, British Columbia
- Dr. Vicki Friesen, Queens University; Kingston, Ontario
- Mr. Thomas Hamer, Hamer Environmental; Mt. Vernon, Washington
- Ms. Kim Nelson, Oregon State University; Corvallis, Oregon
- Dr. Gordy Swartzman, University of Washington; Seattle, Washington

Upon contract award, the Service provided EDAW with all of the documents and datasets received from the information requests in April and July 2003, as well as the Administrative Record database. After reviewing the database of available information, the panelists requested copies of

documents relevant to their areas of expertise. The panelists then spent 2 months reviewing and assessing this information, as well as other relevant articles and data on marbled murrelets. The panelists evaluated the quality of all available relevant information and the validity of the conclusions drawn from the information. If no conclusions were drawn, the panelists determined what, if any, conclusions may be appropriate. And finally, the panelists, in coordination with EDAW, prepared this Evaluation Report, which summarizes and interprets the information on marbled murrelets since listing and includes an assessment of the threats to the listed population.

1.2 Objectives of the 5-Year Status Review

As required by Section 4(c)(2) of the ESA, the purpose of the 5-Year Status Review is to assess the following for a given listed threatened or endangered species:

- Whether new information suggests that the species population is increasing, declining, or stable;
- Whether existing threats are increasing, the same, reduced, or eliminated;
- If there are any new threats; and
- If new information or analysis calls into question any of the conclusions in the original listing determinations as to the species status (68 FR 44093, 25 July 2003).

Information on the marbled murrelet prior to listing in 1992, through 1994-1995, was complied and summarized by Ralph et al. (1995). The Expert Panel process for the 5-Year Status Review focused on new information since 1994-1995. Earlier information on the murrelet is incorporated into this Evaluation Report as needed to provide background and context. Similarly, this Evaluation Report focuses on the 3-state area where the marbled murrelet is currently listed as threatened (California, Oregon, and Washington), but information on the species in Alaska and British Columbia is presented where appropriate, due to the extensive research in these areas.

The 5-Year Status Review process for the marbled murrelet consists of 2 distinct tasks: a scientific review and a regulatory review. This Evaluation Report is the result of the scientific review conducted by the Expert Panel. It provides an evaluation, synthesis, and interpretation of the information related to 4 of the 5 listing factors in the ESA 4(a)(1)(A-E) for the marbled murrelet. These 4 factors are:

(1) Present or threatened destruction, modification, or curtailment of habitat or range;

(2) Over-utilization for commercial, recreational, scientific, or educational purposes;

(3) Disease or predation; and

(4) Other natural or manmade factors affecting its continued existence.

An evaluation of the fifth listing factor, adequacy of existing regulatory mechanisms in protecting the species, will be the responsibility of the Service staff.

This report does not make any recommendations regarding changing or maintaining the listing status of the marbled murrelet. The statutory review regarding the listing status of the species remains the responsibility of the Service. In addition, the Service will also consider the application, if appropriate, of the Policy Regarding the Recognition of Distinct Vertebrate Population Segments (61 FR 4722, 7 February 1996) to the marbled murrelet.

The Service will use the information in this Evaluation Report in its review of murrelet listing status. In addition to this introductory section (Chapter 1), the Evaluation Report consists of an additional 5 chapters, which cover the biology and ecology of the marbled murrelet (Chapter 2); population and demographics (Chapter 3); terrestrial habitat (Chapter 4); and marine habitat (Chapter 5). Chapter 6 provides a summary and conclusion, focusing on threats to the species.

2.0 BIOLOGY AND ECOLOGY

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The following sections provide information on the basic biology and ecology of the marbled murrelet. Relevant data on murrelet breeding biology, movements, diet, and mortality are summarized, with a specific emphasis on findings confirmed after listing in 1992, and, in particular, recent information on marbled murrelet biology gained through research conducted since 1994-1995.

2.1 Species Description and Taxonomy

The marbled murrelet is a small, dove-sized seabird that inhabits the coastal forests and nearshore marine environment along the Pacific coast of North America from southern California to southern Alaska and the Aleutian Islands (Carter and Morrison 1992, Ralph et al. 1995, Nelson 1997). The long-billed murrelet (*B. perdix*) and Kittlitz's murrelet (*B. brevirostris*), which are mostly restricted to northeastern Asia and Alaska, respectively, are the only other species in the *Brachyramphus* genus worldwide. Other murrelet species belong to the genus *Synthliboramphus*, and include Xantus' (*S. hypoleucus*), Craveri's, (*S. craveri*), ancient (*S. antiquus*), and Japanese (*S. wumizusume*) murrelets. All murrelets are wing-propelled pursuit divers in the marine bird family Alcidae, which also includes other genera such as murres (*Uria* spp.), guillemots (*Cepphus* spp.), auklets (*Ptychoramphus* sp., *Cyclorrhynchus* sp., *Aethia* spp, and *Cerorhinca* sp.), puffins (*Fratercula* spp.), razorbill (*Alca torda*), dovekie (*Alle alle*), and the extinct great auk (*Alca impennis*) (Gaston and Jones 1998).

Until recently, long-billed and marbled murrelets were long considered to be 2 races of the same species, despite several morphological differences and original description in the 19th century of the long-billed murrelet as a separate species ("partridge murrelet"). In the mid-1990s, 2 research groups documented major differences in mitochondrial DNA (mtDNA) sequences of a small number of marbled murrelets sampled from Magadan (Russia) and Alaska (Friesen et al. 1996a). In 1996, Friesen et al. (1996b) conducted a more comprehensive analysis of the 2 subspecies, including comparisons of 1,045 base pairs (bp) of the mitochondrial cytochrome *b* gene and 37 allozyme loci between 33 representatives of marbled murrelets (sampled between Attu Island and Oregon), 4 samples of long-billed murrelets, and 7 Kittlitz's murrelets. Results indicated that marbled and long-billed murrelets are genetically distinct, and have probably been reproductively isolated for 5-6 million years. Subsequent analyses of nuclear introns supported the genetic

distinctiveness of these 2 forms (Friesen et al. 1997). Although all of these analyses involved a limited sampling of Asian murrelets (maximum 4 birds, all from Magadan), the American Ornithologists' Union (AOU) recognized the marbled and long-billed murrelets as separate species in 1997.

The marbled murrelet is about 9.25 to 9.5 in. (23-24 cm) long from bill to tail; its body is relatively short compared to wing length. It is similar in appearance to both the Kittlitz's and long-billed murrelets. The marbled murrelet has a dark bill that is longer than the bill of Kittlitz's murrelet, and is heavier in terms of body mass (Friesen et al. 1996a). Compared to the long-billed murrelet, the marbled murrelet has a slightly shorter bill, is lighter, and lacks a white eye ring (Friesen et al. 1996a). The tail of the marbled murrelet is dark brown with white on overlapping underrtail coverts. In breeding plumage, the bird is all dark above and heavily mottled below. In the winter, it is distinguished from most other murrelets by the white on the scapulars. Juvenile plumage resembles that of winter adults but is dusky-mottled below; by the first winter, the underparts are mostly white (National Geographic Society 1987).

Marbled murrelet wings are more pointed in comparison with other alcids. But like other alcids, they are adapted for both underwater and aerial flights. They have reduced wing surface area relative to their body size to reduce drag while underwater and well-developed flight muscles. Consequently, they are relatively stocky birds with high-wing loading (ratio of body mass to wing surface area) (Burger 2002). Birds with high wing-loading require rapid flight speeds to maintain lift (Pennycuick 1987); marbled murrelets fly at 43 miles/hour (70 km/hr) or faster (Burger 2002) and can rise directly off the water without first running across the surface (Audubon Society 1983).

2.2 General Geographic Distribution

The breeding range of the marbled murrelet extends from the Aleutian Islands through central California (Figure 2.2-1). Small numbers have been reported (mostly in the non-breeding season) as far north as the Chukchi Sea and as far south as northwestern Baja California, Mexico. The marbled murrelet is thought to be more or less continuously distributed from southern Alaska to southern British Columbia, but gaps occur in its distribution farther south and along the Aleutian Islands. The current geographic and numeric center of the population is found from Prince William Sound and the Kodiak Island area to southern British Columbia (Ralph et al. 1995). The species is rare or absent from most of southeast Vancouver Island, off lower mainland British Columbia, and parts of the

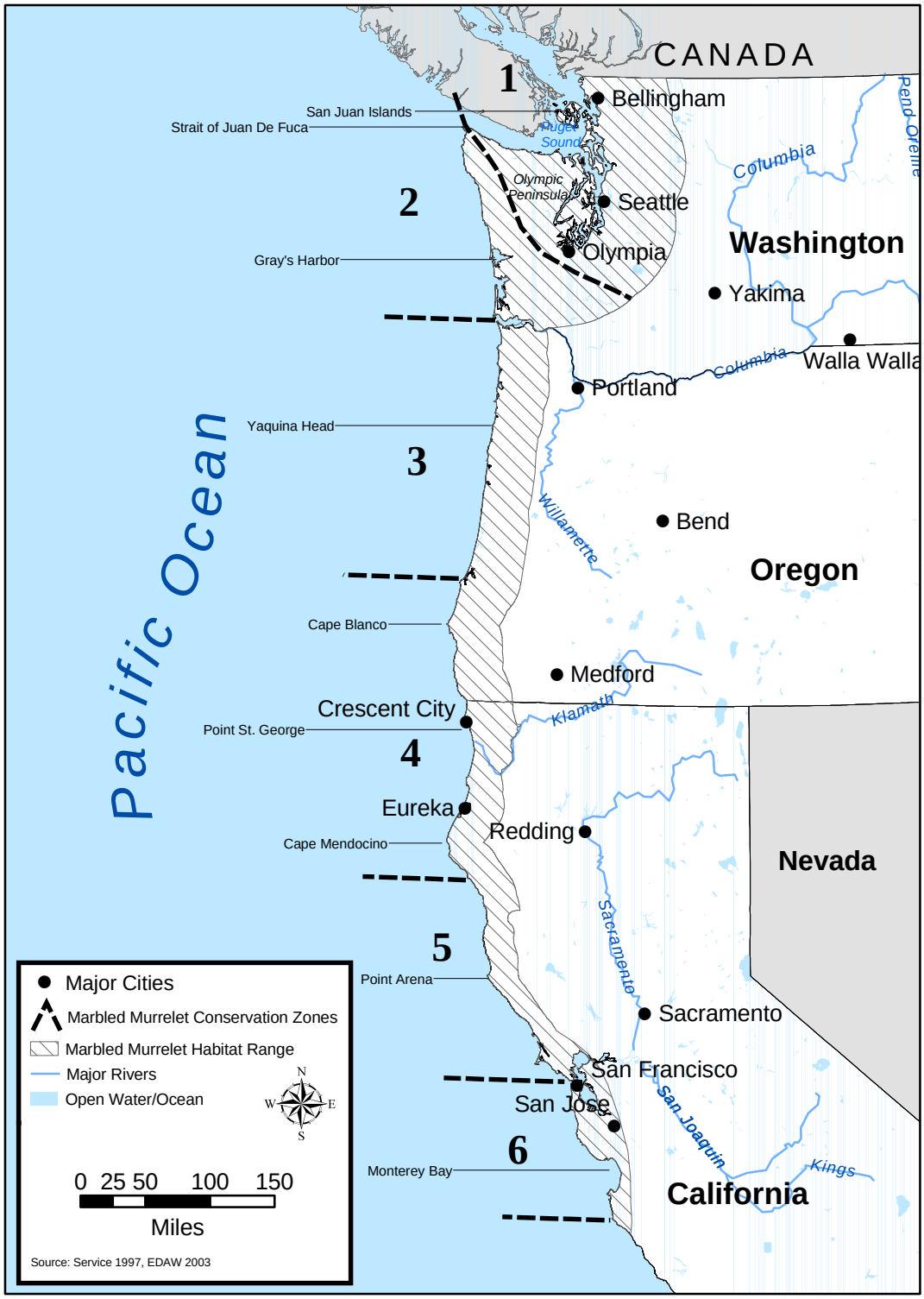


Figure 2.2-1. Range of the Marbled Murrelet in the 3-State Area.

southern Oregon and northern California coasts. The largest gap in distribution occurs in California between Humboldt and San Mateo counties, where only small scattered numbers (<100-300) of murrelets occur (Huff et al. 2003, Lank et al. 2003).

Throughout most of its breeding range, the marbled murrelet uses old-growth coniferous forest habitat for nesting and forages in the nearshore marine environments. At the north end of the range, ground-nesting occurs extensively in the Aleutian Islands and parts of southern Alaska where coastal old-growth forests do not occur. Both ground- and tree-nesting occur in parts of southern and southeastern Alaska area where both tree- and ground-nesting habitats occur widely. In British Columbia, tree-nesting predominates, but ground-nesting was recently documented. In Washington to California, only tree-nesting is known.

The distance inland that marbled murrelets breed is variable and influenced by a number of factors including nesting habitat availability, climate suitability, maximum foraging range, and predation rates. Most murrelets appear to nest within 37 miles (60 km) of the coast (Miller and Ralph 1995); the Service (1997) considers 50 miles (31 km) as the minimum inland distance for determining habitat suitability and amount within the listed range (Figure 2.2-1). Commuting distances are, however, extremely variable, with birds in Washington tending to commute larger distances that those in Oregon and California. In Washington, occupied habitat has been documented 52 miles (84 km) from the coast; a grounded murrelet was found 62 miles (100 km) from the ocean, the maximum inland distance murrelets have been found within the listed range (Hamer 1995). The "Marbled Murrelet Effectiveness Monitoring Plan for the Northwest Forest Plan" (Madsen et al. 1999) considers the primary nesting range of the species to extend inland 40 miles (24 km) in Washington, 35 miles (22 km) in Oregon, and 25 and 10 miles (16 and 6 km) in California, north and south of Fort Bragg, respectively.

Like its inland range, the at-sea range of the marbled murrelet is variable, depending on the extent of available shallow water habitat and the time of year. During the breeding season, murrelets tend to forage in marine waters within 1.2 miles (2 km) of the coast in Washington, Oregon, and California (Strachan et al. 1995). In Alaska, murrelets forage more frequently farther offshore during the breeding season, often as far as 25 miles (40 km) (Piatt and Naslund 1995). Outside the breeding season, birds disperse to varying degrees in different areas and can be less concentrated in nearshore coastal waters (Strachan et al. 1995).

2.3 Food Habits

Like many other alcid species, marbled murrelets are known to be opportunistic feeders (Sanger 1987, Burkett 1995, Nelson 1997), and various studies have documented the broad diversity of the species' diet (summaries by Sealy 1975a, Carter 1984, Vermeer et al. 1987, Burkett 1995, Nelson 1997, Day and Nigro 2000, Becker 2001). In general, small schooling fish and large pelagic crustaceans (euphausiids, mysids, amphipods) represent main prey items for marbled murrelets, with Pacific sand lance (*Ammodytes hexapterus*), northern anchovy (*Engraulis mordax*), immature Pacific herring (*Clupea harengus*), capelin (*Mallotus villosus*), and smelt (Osmeridae) documented as the most common prey species taken. Immature salmon smolts (*Oncorhynchus* spp.), immature rockfish (Scorpaenidae), and eulachon (*Strongylura exilis*) are also taken in some areas. Squid (*Loligo* spp.), euphausiids (principally *Thysanoessa spinifera* and *Euphausia pacifica*), mysid shrimp, and large pelagic amphipods are the main invertebrate prey, frequently ingested by adult murrelets but seldom used to feed chicks. Burkett (1995) summarized many important characteristics of the common prey taken by murrelets. Invertebrates are primarily eaten in the non-breeding season, whereas fish are eaten year round.

Marbled murrelets usually carry a single fish to their chicks and appear to select a relatively large (relative to body size), energy-dense fish for this purpose, typically larger sand lance, immature herring, anchovy, smelt, and occasionally salmon smolts (Carter and Sealy 1987a, Burkett 1995, Nelson 1997, Jones 2001). Older age classes of both sand lance and herring were found to be larger and have higher energy contents than immature classes (Vermeer and Devito 1986, Robards et al. 1999). In California, northern anchovy and Pacific sardine (*Sardinops sagax*) were the most rewarding food items (kilojoules [kJ] per item), although only immature sand lance were considered in that comparison (Becker 2001). Adult sand lance and herring have high energy contents (Burkett 1995). Adult murrelets frequently consume smaller items such as immature sand lance and crustaceans, especially in winter and spring. This pattern of adults taking large prey items back to nest sites but eating large amounts of smaller prey is common among fish-feeding alcid species (Vermeer et al. 1987).

Stable isotope analysis allows non-lethal sampling of murrelet tissues (feathers) to determine prey types. This method can also be applied to museum specimens to compare the trophic levels used by murrelets in the past to those in the present (Hobson 1990, Burkett 1995, Becker 2001). Analyses from Barkley Sound, BC showed isotopes consistent with the dominant prey (sand lance and some marine crustaceans) taken by murrelets

in that area (Carter 1984), although some birds in these samples also showed evidence of significant freshwater prey, likely salmon smolts (Hobson 1990). Results of this study suggest that, depending on regional variation in prey availability, some marbled murrelets may feed extensively on freshwater prey for several weeks, although freshwater protein is not a significant component in the species' diet overall. Feeding on freshwater prey likely occurs mainly in British Columbia and southern Alaska where large coastal lakes with substantial fish populations occur in close proximity to nesting habitats (Carter and Sealy 1986). Marbled murrelets also have been observed on some lakes in Washington but not Oregon or California.

In California, stable isotope analysis of feathers produced during both pre-breeding and post-breeding molts have revealed seasonal and annual variations in diet, some of which may be attributed to oceanographic conditions (Becker 2001). Lower trophic level items (likely crustaceans, juvenile rockfish, sand lance, or herring) were more commonly eaten during the pre-breeding molt in winter than during the post-breeding molt in later summer and early fall. This shift to lower trophic levels was found in non-El Niño years (1999 and 2000) but not in an El Niño year (1998), possibly as a result of reduced prey availability. The analysis also showed some sexual dimorphism in diet: during the pre-breeding molt, all females sampled shifted to lower trophic levels, but only those males that later had brood patches (indicating breeding). This study concluded that adult murrelets were apparently ingesting prey roughly in proportion to their availability, and not specializing in any particular prey species or trophic level. Variations in diets and trophic levels in response to changes in oceanographic conditions are discussed further in Section 5.2.

A comparison of regional studies on the diet of the marbled murrelet reveals an apparent latitudinal gradient in the importance of prey types, with anchovy more important to the south (California, Oregon, and Washington) and sand lance and herring toward the north (British Columbia and Alaska) (Burkett 1995, Derocher et al. 1996, Ostrand et al. 1998). In California, anchovy and, to a lesser extent, sand lance appear to be the major prey items taken during breeding, although other small fish such as immature rockfish are also taken (Burkett 1995, Becker 2001). Museum specimens from birds collected off California reveal that sardines may have been important in the past (Becker 2001). In Oregon and Washington, Burkett (1995) found anchovy, sand lance, and smelt to be the major prey types carried by murrelets to chicks, although sample sizes for this study are notably small. Additional information on geographic variation in the marbled murrelet's diet is provided in Section 5.2.

2.4 Foraging Behavior

Like all alcids, murrelets dive using their wings as the primary source of propulsion. Although occasionally occurring in deeper water, most foraging appears to occur in relatively shallow nearshore water, generally <98 feet (30 m) deep (Sealy 1974, Strachan et al. 1995, Burger 2002). The most common foraging depths are not known. However, marbled murrelets incidentally collected in gill-nets in Barkley Sound, BC were captured 9.8-16.4 feet (3-5 m) below the surface at night (Carter and Sealy 1984), and murrelets are known to feed on small schools of fish within the upper 16.4 feet (5 m) of marine waters (Mahon et al. 1992, A. E. Burger unpubl. data). An alcid the size of a murrelet is expected to have a maximum diving depth of about 154 feet (47 m) (Mathews and Burger 1998), although the deepest incidental collection was recorded in a gill-net at 89 feet (27 m) depth off California (Carter and Erickson 1992). Based on prey remains found in murrelets collected in Alaska, Sanger (1987) suggests that birds may forage in mid-water depths of 59-148 feet (18-45 m). Jodice and Collopy (1999) reported no variation in dive times with depth of water (9.8-118 feet [3-36 m]), with most diving recorded in water <33 feet (10 m) deep. In this study, the authors suggest that murrelet diving effort may increase (shorter pauses between dives) in years with reduced prey availability.

Small juvenile fish (e.g., sand lance and herring) are often found at shallow to moderate depths (likely less than 98 feet [30 m]) requiring dive times between 15-60 seconds (Carter and Sealy 1990, Strachan et al. 1995). At times, however, these fish are caught near the sea surface with very short (<5-10 seconds) and shallow (<3.2-16.4 feet [1-5 m]) dives, and murrelets have been observed in multi-species flocks feeding on near-surface schools of small fish in some coastal areas (Sealy 1973, Mahon et al. 1992). Most often, however, murrelets forage in pairs or less commonly solitarily or in groups of 3 or more (Sealy 1975a, Carter and Sealy 1990, Strachan et al. 1995, Speckmann et al. 2003, McFarlane Tranquilla et al. in press). In areas rich in prey, loose at-sea aggregations of murrelets (tens to thousands of birds) can form and remain for periods of weeks or months (Carter 1984, Sealy and Carter 1984, Carter and Sealy 1990). Large feeding aggregations (hundreds to thousands of birds) are evident in parts of British Columbia and Alaska where larger population sizes occur and where prey can be more concentrated for periods of time through local topography, sandy substrates used by sand lance, and near large herring spawning areas. Small feeding aggregations (tens to hundreds of birds) are found throughout the breeding range where prey are concentrated or available. Within such feeding aggregations, murrelets still mainly feed as pairs and singles.

Paired foraging has been documented as common throughout the year, even during the incubation period (when one bird of each mated pair is on the nest), suggesting that murrelets may temporarily pair up on the water, perhaps to obtain some benefit from feeding together (Strachan et al. 1995, Speckmann et al. 2003). However, since egg laying and incubation are not highly synchronized and high rates of nest failure occur, most paired foraging observed during the incubation period also likely represents mated pairs.

2.5 Reproduction and Nesting Chronology

Substantial portions of murrelet populations have been found to remain near inland nesting areas year-round in the southern parts of their range (e.g., California: Carter and Erickson 1992, Naslund 1993, O'Donnell et al. 1995; Washington: Cross 1992; British Columbia: Carter and Sealy 1986). Higher proportions appear to exhibit year-round residency farther south. In the more northern parts of their range, most murrelets undertake seasonal migrations, moving away from nesting grounds after breeding to overwinter elsewhere, although small numbers can remain during winter in breeding areas (e.g., British Columbia: Burger 1995, 2002; Alaska: Agler et al. 1998, Kuletz and Kendall 1998). Where seasonal migration is common, murrelets return to the breeding grounds in early to mid-April and, in most cases, appear to be already paired (Sealy 1974, 1975b, McFarlane Tranquilla et al. in press). Courtship includes various displays by birds on the water and in flight, and copulation has been observed on the water and on branches of large trees (Nelson 1997).

Breeding is asynchronous and spread over a more prolonged season than for most temperate seabirds (Nelson 1997, Gaston and Jones 1998). Seasonal and regional variation in the breeding biology and chronology of laying for the marbled murrelet are provided in Table 2.5-1.

Marbled murrelets lay a single-egg clutch (Sealy 1974, Nelson 1997). Replacement of a lost egg following early breeding failure has been documented for small numbers of murrelets in northern California (Hebert et al. 2003) and British Columbia (McFarlane Tranquilla et al. 2003a). Studies in Alaska indicate either a low frequency of renesting or little successful renesting, based on circumstantial evidence at 1 nest (Naslund et al. 1995) and patterns of juvenile appearance at Naked Island (Kuletz and Kendall 1998). The species' extended breeding season in comparison to the length of time needed for incubation and chick-rearing (60-70 days; Nelson 1997) suggests that replacement laying is likely to occur throughout the species' range. However, there is no evidence that marbled murrelets lay a second egg after successfully fledging a first chick (i.e., "second brooding"),

Table 2.5-1. Chronology of breeding for the marbled murrelet based on a limited number of known records, showing approximate dates of each phase of breeding in each state or province.

Region	Egg Laying & Incubation	Chicks	Fledglings	Reference
California	late March to mid August	late April to mid September	late May through early October	Hamer et al. 2003
Oregon	late April to late August	late May to late September	late June to early October	Hamer et al. 2003, Hamer & Nelson 1995
Washington	late April to early August	late May to late August	late June to early September	Hamer et al. 2003
BC	late April to late August, peak laying end May to early June	late May to early September	late June to late September	Hamer et al. 2003, Burger 2002, Lougheed et al. 2002
Alaska	mid May to mid August	mid June to mid September	mid July to early October	Hamer et al. 2003

as has been found to occur for Cassin's auklets (*Ptychoramphus aleuticus*) in central California (Manuwal 1974).

Incubation is shared by both sexes, and incubation shifts are generally 1 day, with exchanges occurring at dawn (Nelson 1997, Bradley 2002). Hatchlings appear to be brooded by a parent for 1-2 days and then left alone at the nest for the remainder of the chick period while both parents spend most of their time at sea. Both parents feed the chick – usually a single fish carried in the bill – and a chick typically receives 1-8 meals per day (mean 3.2) (Nelson 1997). About two-thirds of the meals are delivered early in the morning, usually before sunrise, and about a third at dusk with a few meals sometimes scattered through the day (Hamer and Nelson 1995). Bradley et al. (2002) documented significant differences between sexes during chick-rearing; males made 1.3 times more inland trips than females overall, and made 1.8 times as many trips at dusk. During early chick-rearing, nest visitation rates by males and females were found to be similar, but toward the end of chick-rearing female visitation declined while males maintained the same visitation rates. Males therefore provision the chicks more often than females, especially during the last half of chick rearing (Bradley et al. 2002).

Chicks have been found to fledge 27-40 days after hatching, at 58-71% of adult mass (Nelson 1997). Fledging has seldom been documented but appears to occur typically at dusk (Nelson 1997, Jones 2001). Fledged juveniles appear to receive no parental care and are often seen solitarily on marine waters after leaving the nest (Nelson 1997). The first flight of a fledgling is risky, and there are several documented cases of grounded fledglings (see below).

2.6 Variations in Nesting Chronology

The timing of breeding is undoubtedly affected by local ocean conditions and prey availability, but the details of these effects are poorly known (Nelson 1997). In Alaska, Speckmann et al. (2000) noted murrelet breeding to be earlier and more successful in a year with warmer spring temperatures (1993) than in a cooler year (1992). This seasonal variation in reproductive success and chronology was attributed to increased zooplankton productivity and hence growth in forage fish in the warmer year (Speckmann et al. 2000).

In a radio-telemetry study in Desolation Sound, BC, Bradley (2002) found that early nesting murrelets tended to travel farther from foraging areas to nest and used trees on steeper slopes than those nesting later. Although statistically significant, these correlation coefficients were relatively small. Timing of breeding was not found to be correlated with elevation. Using univariate logistic regression models to study nesting success, Bradley (2002) found that early-breeding birds were more successful, and success increased with increasing commuting distance from foraging areas, slope, and elevation. No apparent variation among years was detected, and multivariate analyses, including stepwise models, did not resolve which was the dominant effect. Reduced predation at nests farther inland and at higher elevations was suggested as a possible explanation for these patterns (Bradley 2002).

2.7 Molt Cycles

As in most alcids, adult marbled murrelets molt into alternate (breeding) plumage in the spring (February through May) and molt into basic (winter) plumage in late summer through early fall (July through November) (Carter and Stein 1995, Nelson 1997). The full pre-basic molt takes 2-3 months, with primaries, secondaries, and rectrices (tail) requiring 45-75 days. Molting birds are flightless for up to 2 months during this time (Nelson 1997). Adults and subadults often move away from breeding areas prior to molting and must select areas with predictable prey resources during the flightless period (Carter and Stein 1995, Nelson 1997).

Subadults, 1 to 2 years old, lacking brood patches or raised levels of vitellogenin (a precursor to egg yolks indicating breeding in mature females) also assume alternate plumage in spring and summer and are externally inseparable from adults (Sealy 1974, Carter and Stein 1995, McFarlane Tranquilla et al. 2003b,c). Thus, plumage characteristics are unreliable for demographic analysis. Within local populations, the timing of molt has been found to vary somewhat among individuals, especially during the pre-basic molt (Nelson 1997). Sealy (1975) concluded that failed breeders are likely to

begin molt well before successful breeders. During the pre-basic molt, all feathers are gradually replaced, but during the pre-alternate (spring) molt, only the body contour feathers are replaced and not the flight feathers (Carter and Stein 1995).

Newly fledged juveniles have a similar plumage to adult basic plumage, although variable amounts of fine barring is evident on breast feathers, which are solid white in adults (Carter and Stein 1995). By late fall, all age classes look similar.

2.8 Movement and Dispersal

Marbled murrelet movements within and among seasons are poorly known because of the difficulties of catching and recapturing marked individuals for standard banding studies. Murrelets may exhibit small-scale seasonal migrations; unlike most migrant waterbirds, however, only a portion of the population appears to leave breeding grounds (Nelson 1997). Major differences in movements exist between portions of the annual cycle when murrelets attend nesting habitats and when they undergo the pre-basic molt.

When attending nesting habitats during the breeding season (and much of the non-breeding season in southern parts of the range), adult murrelets are restricted to foraging within commuting distance from the nest site. Daily commutes likely occur only during the breeding season, whereas less frequent visitation likely occurs in the non-breeding season. Daily movements of breeding adults monitored with radio-telemetry showed that mean distances between nest sites and foraging areas averaged 10 miles (16 km) (range 0.6-19 miles [1-31 km]) in Prince William Sound, Alaska (K. Kuletz, pers. comm.) and 24 miles (39 km) (range 7.4-63 miles [12-102 km]) in Desolation Sound, BC (Hull et al. 2001). Much longer-nest-to-foraging site distances (mean 48 miles [78 km], maximum 80 miles [128 km]) were recorded in southeast Alaska where the murrelets traveled down long fjords to forage in more open waters (Whitworth et al. 2000). However, in the latter study, radio-marked birds may not have been actively breeding when longer movements occurred.

During pre-basic molt, adult and subadult murrelets are flightless, and movements are restricted to swimming at sea for 1 to 2 months (Carter and Stein 1995). In regions where most murrelets do not regularly attend nesting habitats in winter (as suspected in most of Alaska and northern British Columbia), birds are no longer restricted to foraging near nest sites and can potentially move great distances. In Alaska and British Columbia, molting and wintering aggregations probably include murrelets from widespread sub-populations (Rodway et al. 1992, Burger 1995), although some birds appear

to molt and remain near breeding areas during winter in certain areas of British Columbia (Carter and Stein 1995, Beauchamp et al. 1999). Many birds breeding on exposed outer shores of Vancouver Island appear to move into more sheltered waters in Puget Sound and the Strait of Georgia, where numbers increase in fall and winter (Burger 1995). Murrelet numbers are also known to decrease markedly during winter in southeast Alaska (Agler et al. 1998). However, most or all of these birds appear to stay within the northern part of the range.

In the 3-state area, murrelets appear to be largely resident throughout the year, although greater residency occurs in California than in Oregon and Washington. In central California (Zone 6), most radio-marked murrelets remained near nesting areas in late summer and during the pre-basic molt (Burkett et al. 1999, Peery et al. 2003). A few birds dispersed as far north as southern Mendocino County (Zone 5) and as far south as San Luis Obispo County (south of Zone 6) prior to pre-basic molt. In early October, greater dispersal also occurred after molt before batteries in the radio transmitters failed. Lower attendance of nesting areas in this area occurs in the August to November period as breeding ends, during pre-basic molt, and for a brief period thereafter (Carter and Erickson 1992, Naslund 1993). Juveniles are not restricted to foraging near nest sites and can disperse widely.

Beauchamp et al. (1999) provided evidence of 1 banded bird moving between British Columbia and Washington. An adult in breeding plumage banded in the summer of 1995 in Theodosia Inlet (Desolation Sound, southern mainland of British Columbia) was caught in the fall of 1996 in the San Juan Islands, Washington, and was then recaptured again during the 1997 breeding season in Desolation Sound. This movement was consistent with short-distance cross-border dispersal within the same marine ecosystem in the Straits of Georgia and northern Puget Sound during the non-breeding season. It was not known if the bird remained in Washington waters for the entire winter of 1996-97 or returned to British Columbia waters shortly after capture. Seven other color-marked murrelets nesting near Desolation Sound, however, appeared to remain there after breeding (Beauchamp et al. 1999).

Throughout this report, we used the term "dispersal" to describe temporary movements of murrelets away from natal nesting areas post breeding (i.e., during the late breeding season and non-breeding season) before later return back to their natal nesting areas before the following breeding season. We used the word "immigration" to describe permanent movements of birds away from natal breeding areas with continued breeding in non-natal nesting areas. To simplify the text, we also used the word "immigration" for both immigration and emigration movements.

2.9 Site Fidelity and Natal Dispersal

Little data are available regarding nest site fidelity by marbled murrelet because of the low number of observed nest sites and difficulty of observing bands on birds attending nest sites. There are 15 records of murrelets using nest sites in the same or adjacent trees in successive years, but it is not clear if they were used by the same birds (Singer et al. 1995, Manley 2000, Nelson and Peck 1995). From the small amount of data available, it appears that re-use of specific or adjacent nest sites in a subsequent season may be more common in areas where large, old-growth trees are rare or predation is limited, but this has not been verified (Nelson and Peck 1995, Singer et al. 1995, Manley 1999). However, recent confirmation of use of the same nest site for 2 consecutive years in Redwoods National Park (Hebert and Golightly 2003), one of the largest remaining areas of murrelet nesting habitat in northern California, lends additional support to this hypothesis. At a larger landscape scale, murrelets do show fidelity to foraging areas and probably to specific watersheds for nesting (Nelson 1997). This is evident from the recaptures of murrelets banded as adults in the same foraging grounds or flight-paths during the breeding season (Cam et al. 2003, Peery et al. 2003).

Marbled murrelets are generally seen in pairs year-round (see Section 2.4), suggesting that pair-bonds are strong and persist beyond a single breeding season. The only empirical evidence for this, however, is the recapture of a single pair in 2 successive seasons in British Columbia (McFarlane Tranquilla et al. in press). This study also showed that pairs that bred successfully were detected together by telemetry for significantly longer periods (mean 55 days, n=6) than pairs that failed (mean 19 days, n=5).

Breeding adults are highly mobile, and radio-marked birds from many nesting areas mingle at productive foraging sites during the breeding season (Whitworth et al. 2000, Hull et al. 2001, Bradley 2002). Radar surveys in Clayoquot Sound (Burger 2001) and on the Olympic Peninsula (Raphael et al. 2002a) suggest some movement of birds among watersheds from year to year. Radio-telemetry has shown that many birds routinely forage in the same general areas, but a few birds have been tracked making substantial changes in foraging sites during the breeding season (Bradley 2002).

The degree of philopatry (proportion of chicks that return to breed at or near the place where they hatched) in marbled murrelets is not known (Divoky and Horton 1995). Most other alcids exhibit fairly strong but not invariable philopatry, and a small proportion of birds can breed at colonies other than their natal colony. Divoky and Horton (1995) surmised that natal dispersal in

marbled murrelets was likely to be high because they are non-colonial and nest in widely dispersed nest sites, which can potentially be located across wide swaths of inland areas (up to 19-50 miles [30-80 km] inland) if habitat is available. Conversely, Swartzman et al. (1997) suggested that natal dispersal in marbled murrelets was likely to be low, like other alcids (including colonial and less colonial species). Only 1 study (Harris and Wanless 1991) has shown extensive natal dispersal by alcids, during a period of rapid population growth and colony saturations.

Actual data on juvenile dispersal and natal philopatry are sparse. Only 2 out of 106 murrelets banded as juveniles in Desolation Sound, BC, have been recaptured there in subsequent years (Lank et al. 2003). This low recapture rate could be the result of poor recapture techniques or a low number of banded birds relative to the population, which reduces the likelihood of recapture. Assuming adequate sample size and recapture techniques, this low rate could suggest that natal philopatry is not strong, survival is very low, or subadults may not return to natal areas to breed for several years. Juveniles that were radio-tagged in Clayoquot Sound, west Vancouver Island, BC, moved >124 miles (200 km) or more to the north after fledging, but their final destinations were not known (Lank et al. 2003).

2.10 Mortality

Predation, particularly during the breeding season, is the most documented cause of mortality, but its demographic importance, relative to other causes of mortality such as starvation and disease, is not known. Known predators of murrelets have been summarized by Nelson (1997) and Burger (2002). More detail on the threats from oil spills and gill-nets are presented in Section 5.4; threats from predation relative to changes in forest fragmentation and edge-effects are summarized in Section 4.5.6.

2.10.1 At-Sea Mortality

At sea, predation on marbled murrelets by bald eagles (*Haliaeetus leucocephalus*), peregrine falcons (*Falco peregrinus*), western gulls (*Larus occidentalis*), and northern fur seals (*Callorhinus ursinus*) has been reported (Campbell et al. 1997, Vermeer and Butler 1989, Rodway et al. 1992, Nelson 1997, Hooper 2001). California sea lions (*Zalophus californianus*), northern sea lions (*Eumetopias jubatus*), and large fish may be occasional predators as well (Burger 2002). Starvation at sea, though poorly documented, is suspected to occur in fall and winter, and little is known about the effects of disease or parasites (Nelson 1997) (see also Section 3.6.2.). Murrelets at sea have been killed by oil spills, gill-nets, and anglers' hooks (see Section 5.4).

Routine beached bird surveys have not provided much information on causes of at-sea mortality, partly because marbled murrelets are seldom found dead on beaches. Stenzel et al. (1988) reported apparent causes of mortality for seabirds found dead on California beaches in 14 years of beached bird surveys (1971-1985). Unfortunately, details of the 23 marbled murrelets found dead were not given, but the species was included in "small alcids." In this group, only 9.6% of carcasses had some cause of death recorded, of which 98.8% were oiled (the highest incidence among any bird group) and 1.1% had a broken wing. The marbled murrelet is among the most vulnerable of seabirds to oil spills because they remain at sea most of their lives in nearshore areas near shipping lanes and other sources of oil (Burger 2002). Major oil spills have killed hundreds of murrelets off British Columbia, Washington (*Nestucca* and *Tenyo Maru*), and Oregon (*New Carissa*) and thousands in Alaska (*Exxon Valdez*). Low-level chronic oil pollution is also likely to kill murrelets (see Section 5.4.3).

Data provided by the Coastal Observation and Seabird Survey Team (COASST) included records for a total of 9 marbled murrelet carcasses from beached bird surveys conducted in Washington and Oregon from July 1, 2001 through June 30, 2003 (T. Hass, pers. comm.). Although the specific causes of mortality are not often determined, oil and gill-net entanglement were not implicated in these particular cases. The level of effort and geographic diversity of beaches covered by COASST include over 100 beaches in Washington and Oregon, which are generally surveyed bi-weekly by over 200 volunteers. Of the 9 marbled murrelets recorded using these surveys, 7 were found at a single site, Hobuck Beach, on the north coast of Washington. Five carcass recoveries were recorded at this site during January and February, and 2 others were documented during 2 separate surveys (COASST unpublished data). It is unclear if this apparent aggregation reflects offshore distribution, stochastic variability, or increased regional mortality risk. However, COASST data suggesting regional variance in the distribution of beached murrelets in the northwest emphasize the need for additional study of mortality risk for marbled murrelets at sea.

2.10.2 Nest-Site Mortality

Marbled murrelets are highly vulnerable to nest site predation. Most active murrelet nests that have been detected and monitored have been found to fail, and most failures appear to be the result of predation (Nelson and Hamer 1995; Hamer and Meekins 1999; Manley 1999; Manley and Nelson 1999; Bradley 2002; Hebert and Golightly 2003; Nelson and Wilson 2002; Manley 2003; Peery et al. in prep.) (see Section 4.5.6 for more detail). Common ravens (*Corvus corax*) and Steller's jays (*Cyanocitta stelleri*) are known

to take both eggs and chicks at the nest, while sharp-shinned hawks (*Accipiter striatus*) have been found to take chicks. Suspected predators at nests include great horned owls (*Bubo virginianus*), barred owls (*Strix varia*), Cooper's hawks (*Accipiter cooperi*), northwestern crows (*Corvus caurinus*), American crows (*C. brachyrhynchos*), and gray jays (*Perisoreus Canadensis*) (Nelson and Hamer 1995, Nelson 1997, Manley 1999).

In Pacific Northwest old-growth forest, several small mammals species, including deer mice (*Peromyscus maniculatus* and *P. keeni*) (Bradley and Marzluff 2003), are known to visit tree canopies, and squirrels have been found to occur in high densities (Carey 1995, 1996). While small mammals have been documented approaching murrelet nests with incubating adults and chicks, they have never been documented attempting to take eggs or kill the chicks (Nelson 1997, Singer et al. 1997). However, experimental work with artificial nests indicates that predation by squirrels and mice on eggs and murrelet chicks cannot be discounted as a possibility (Luginbuhl et al. 2001, Raphael et al. 2002a, Bradley and Marzluff 2003). The northern flying squirrel (*Glaucomys sabrina*), red squirrel (*Tamiasciurus hudsonicus*), Douglas squirrel (*Tamiasciurus douglasi*), deer mouse, bushy-tailed woodrat (*Neotoma cinerea),* and an unidentified mustellid were all recorded attacking artificial (plastic) eggs and pigeon nestlings (*Columba livia*) (Marzluff et al. 1999, Flaherty et al. 2000, Luginbuhl et al. 2001, Bradley and Marzluff 2003). Other experiments, however, have shown that northern flying squirrels are unlikely to be able to break into marbled murrelet eggs (Flaherty et al. 2000). Artificial nest studies also showed that corvids were more important predators on eggs, whereas mammals, given their olfaction, were more adept at depredating simulated nestlings.

Corvids have been implicated as the primary predator of active murrelet nests, and corvids and squirrels were the key predators at artificial nests (Nelson and Hamer 1995, reviewed in Raphael et al. 2002a). At active nests, corvids were equally successful at depredating murrelet chicks and eggs. However, common ravens accounted for a majority of egg depredations as they appear to be the only predator capable of flushing incubating or brooding adults from nests (Nelson and Hamer 1995, Singer et al. 1991, Suddjian 2003). Populations of several corvid species have increased dramatically in western North America as a result of forest fragmentation, increased agriculture, and urbanization (Marzluff et al. 1994) (see also Section 4.5.6).

2.10.3 Inland Adult Mortality

In forest habitat, known predators of adult marbled murrelets include peregrine falcon, sharp-shinned hawk, and common raven (Singer et al. 1991, Marks and Naslund 1994, Nelson and Hamer 1995). In addition, remains of murrelets have been found at nests of northern goshawks (*Accipiter gentilis*), bald eagles, and peregrine falcons (Nelson 1997, Burger 2002).

The specific impacts of predation by northern goshawks are poorly understood but potentially important to the population dynamics of the marbled murrelet. Steven Lewis (ADFG unpublished data) analyzed prey taken by goshawks in southeast Alaska. Marbled murrelets comprised 2.8% (10 of 361) of prey remains collected from 28 nests throughout southeast Alaska, and were found at 7 of the 28 nests. Similarly, 3.1% (12 of 382) of prey identified from pellets were murrelets or unidentified alcids presumed to be murrelets. Prey deliveries were observed at 10 nests throughout southeast Alaska over a period of 2 years. Marbled murrelets made up 0.8% (11 of 1,451) of all provision deliveries and were delivered to 6 of the 10 nests observed. Not unexpectedly, most murrelets were delivered early in the morning (9 between 03:35 and 05:11, 1 at 08:29, and 1 at 20:28). Iverson et al. (1996) reported alcid (mostly murrelet) remains at 20% of 15 goshawk nests investigated in southeast Alaska.

On Vancouver Island, marbled murrelet remains were found in 15% of 90 goshawk pellets, and murrelets were ranked fourth in prey species' occurrence (Ethier 1999). This suggests that predation by goshawks might be more common than previously suspected (Burger 2002).

Interactions among goshawks, murrelets, and murrelet nest predators, themselves potential goshawk prey (e.g., jays and squirrels), are not clearly defined but are potentially important in affecting murrelet nesting success. If goshawk and murrelet densities in remnant old-growth patches are artificially increased as a result of logging practices and continued habitat loss, then goshawk predation risk to murrelets would likely increase considerably (Burger 2002). This interaction is, however, complicated by the fact that goshawks also kill many nest predators, which might therefore indirectly benefit murrelets. In Ethier's (1999) sampling, the most common prey species taken by goshawks was found to be red squirrel (69% occurrence in goshawk pellets). Other known or potential predators of murrelets found were Steller's jay (38%), gray jay (4%), and northern pygmy-owl (*Glaucidium gnoma*; 1%). If the occurrence of species remains in a pellet represents a separate animal, for every 14 marbled murrelets killed, 101 potential predators (squirrels, jays, owls, etc.) were killed by goshawks. Squirrels and

jays are known to be important prey for goshawks in southeast Alaska, and the introduction of red squirrels to some islands might have benefited regional goshawk populations (Iverson et al. 1996).

The absence of goshawks might lead to "mesopredator release," when the decline or absence of larger predators allows an increase in mid-sized predators (such as squirrels and jays), sometimes causing significant detrimental effects on prey populations (Crooks and Soulé 1999). It is not clear whether direct predation of goshawks on murrelet adults has a greater impact on murrelet populations than reductions in nest predation that goshawks might induce by killing squirrels and jays. The loss of an adult bird is thought to have a much greater impact on murrelet populations than the loss of an egg or chick (Beissinger and Nur 1997, Cam et al. 2003), and the relative impacts of losing a few adults or losing a greater number of eggs and chicks have not been analyzed. Clearly, the impacts of goshawks and other inland predators on murrelet populations need to be examined in more detail, within the context of reduction and fragmentation of old-growth forests.

2.10.4 Other Causes of Mortality

Other causes of marbled murrelet mortality are less well documented in comparison to data on murrelet predation. Documented causes of mortality include collisions with vehicles and transmission wires by low flying adults (Nelson 1997). Nelson (1997) reports at least 5 documented instances of marbled murrelet mortality resulting from vehicular collision. Nesting adults are thought to be especially susceptible to vehicular traffic risk where nests are located in the vicinity of roads as birds typically approach nests from below to allow for a controlled stall on the nest limb (Nelson 1997). Murrelet adults are also believed to have been killed through collisions with transmission lines, with 1 mortality reported from Mapleton, Oregon and 2 from around Juneau, Alaska (K. Nelson pers. obs., Nelson 1997).

Nestlings are known to fall from nests (Binford et al. 1975, Manley 1999), and adults and nestlings die when trees are felled (Nelson 1997). There are many records of fully grown fledglings grounded in forests and elsewhere en route to the sea (Carter and Sealy 1987b, Nelson and Hamer 1995, Nelson 1997, Burger 2002), but the proportion of fledglings lost in this way is not known. Nelson and Wilson (1999) documented the death of a nestling from renal failure (dehydration), possibly due to being provisioned by only 1 parent. Another monitored nestling was confirmed to have died of a burst aorta (Nelson 1997).

2.11 Summary and Conclusions

Because of their low reproductive rate, marbled murrelets are sensitive to small changes in adult mortality, fecundity, and other demographic parameters. They are also difficult to monitor, making it hard to effectively detect population declines in a reasonable timeframe. Cryptic coloration of the egg/chick/adult, crepuscular activity patterns, selection of hidden nest sites with high overhead and horizontal cover, retention of down by the chick, fledging just after dusk, and other breeding habits of marbled murrelets likely indicate that the species evolved under immense predation and nest site competition pressure (Nelson and Hamer 1995, Burger 2002). Predation on adults may be especially important to the survival of the species because demographic models indicate that adult mortality may have a greater impact on murrelet population growth than juvenile survival or nesting success (productivity).

3.0 POPULATION AND DEMOGRAPHICS

3.0 POPULATION & DEMOGRAPHICS

Since California, Oregon, and Washington populations of the marbled murrelet were listed in 1992, there has been considerable research on the population characteristics and demographics of the species. Use of standard protocols for at-sea surveys has resulted in more reliable estimates than previous efforts, which were directed toward determining the existing population distribution and density over the 3-state area. Improved demographic information and models provide the best available information on expected future population trends. And finally, additional sampling and new analytical tools have expanded our knowledge of genetic variability between murrelet populations. This chapter summarizes information on murrelet demographics, estimates of population size and trends (both at present and in the future), genetic variability, and potential threats on a population or demographic scale.

3.1 Demographic Characteristics

Burger (2002) and Lank et al. (2003) recently conducted extensive reviews of several marbled murrelet demographic characteristics that are important in relation to population modeling (see Section 3.5). This section briefly summarizes major new findings concerning key demographic parameters or confirms previous research results for the characteristics listed below.

3.1.1 Sex Ratio, Age at First Breeding, and Clutch Size

Information on these characteristics is summarized as follows:

- *Sex Ratio*: The sex ratio is equal for adults and juveniles (and by extension to subadults). This confirms past studies.
- *Age of First Breeding*: Breeding probably begins at 2-5 years, and was previously considered to be 3 years.
- *Clutch Size*: Murrelets lay 1 egg, which is incubated by both members of a breeding pair. Current research confirms past studies.

3.1.2 Replacement Eggs

Laying of replacement eggs is known to occur among marbled murrelets but is difficult to document (Hebert et al. 2003, McFarlane Tranquilla et al. 2003a). This may occur frequently given the high rates of nest failure, but

frequency has not been well measured in most areas. Burger (2002) speculated that 5% of lost first clutches may be replaced each year, but this estimate is likely low. McFarlane Tranquilla et al. (2003a) reported a minimum rate of replacement laying of 14% based on radio telemetry evidence. Higher rates (up to 63%) could be calculated with various assumptions, but these assumptions could not be verified. Replacement eggs will increase breeding success if some are successful. Hebert et al. (2003) reported 2 failed replacement nesting efforts in northern California, whereas McFarlane Tranquilla et al. (2003a) reported 3 successful replacement nesting efforts in southern British Columbia. We suspect that, like other alcids (Gaston and Jones 1998, McFarlane Tranquilla et al. 2003a), replacement nesting efforts tend to be more successful if they occur early in the breeding season. Until recently, replacement eggs were not documented in marbled murrelets. This recent knowledge may indicate that the extended breeding season of murrelets may be related largely to replacement laying, rather than high asynchrony.

3.1.3 Breeding Success

For this review, breeding success is defined as the number of fledglings that depart from the nest site per egg-laying pair per year. Breeding success for marbled murrelets is likely a function of nest predation, timing, foraging conditions, prey availability, and adult survival during the breeding season. Estimates of breeding success are best determined from nest site data, but difficulties in finding nests has led to widespread use of adult:juvenile ratios as an indirect index of breeding success. Various biases affect adult:juvenile ratios (Beissinger 1995a, Carter and Stein 1995, Ralph and Long 1995, Beissinger and Nur 1997, Burger 2002), including:

- Survey timing;
- Timing of breeding;
- Proportion of adults breeding;
- Nesting success;
- The difficulty of determining age during the pre-basic molt period;
- Differences in timing of dispersal;
- Mortality; and
- Variation in at-sea habitats.

Various adjustments to bring adult:juvenile ratios closer to nest-based measures of breeding success are theoretically possible, but these adjustments have not been verified. Certain biases may be reduced in large survey areas on the outer west coasts of California and Oregon, compared to British Columbia, Alaska, and inner Washington waters. Compared to Alaska and British Columbia, unadjusted and adjusted adult:juvenile ratios indicate relatively low breeding success in California, Oregon, and Washington, with lowest ratios in central California and highest ratios in Washington (e.g., 0.02-0.09 chicks per pair; Beissinger and Nur 1997; Beissinger and Peery 2003). Date-corrected adult:juvenile ratios are higher in Washington, Oregon, and northern California (0.080-0.089) and lower in central California (0.038) (Beissinger and Peery 2003). Unadjusted ratios as high as about 0.08 and adjusted ratios as high as about 0.13 have been found in British Columbia and Alaska (Kuletz and Kendall 1998; Burger 2002). Breeding success estimated from nests with known outcome is generally higher (e.g., 0.22-0.35 chicks per pair; Nelson and Hamer 1995; see Section 4.5.6), but nests with known outcome probably were not selected randomly and may contain bias (e.g., a greater proportion of edge nests with lower success or telemetry nests with unknown outcomes may preferentially exclude successful or unsuccessful nests).

Recent telemetry studies of individual nests also have found a trend of higher success in the central part of the breeding range and lower success in the southern part of the breeding range. Highest reported t levels of breeding success have been found in southern British Columbia (0.46 chicks per pair; Bradley 2002). Insufficient data are available for Alaska;6 nests were located with radio telemetry in 1994 but outcomes were not determined (K. Kuletz, pers. comm.). Lower levels of breeding success were documented in a radio telemetry study in northern California (0.135-0.324 chicks per pair) (R. Golightly, pers. comm.), and very low levels in 2 radio telemetry studies in central California (0.00; Peery et al. in prep.; Burkett, unpubl. data). Estimates of breeding success from telemetry studies involve randomly-located nests but may be biased due to non-random selection of murrelets during capture, capture and radio-tagging effects, radio failures prior to fledging, years sampled, and the small size of areas sampled. All of these biases are greatly exacerbated by the very low sample sizes in these studies. No telemetry-based breeding success data have yet been gathered in Oregon or Washington (although a study is planned for 2004 in Washington), which is important because regional differences are expected due to variation in nesting and feeding habitats.

One major problem with estimating breeding success in telemetry studies is the exclusion of birds that do not lay eggs after capture and radio-tagging but

otherwise appear to be in reproductive condition at capture (e.g., have a brood patch). Many or all of these birds likely laid eggs and failed before or in response to capture. Excluding these birds may lead to artificial inflation of breeding success. Until more information is available, we suggest that higher breeding success values from telemetry or other data are likely to be most representative of true breeding success within areas studied. In general, telemetry data are preferred over adult:juvenile ratios and nests with known outcome due to fewer biases, but telemetry data are not as widely available. In summary, if telemetry-sampled nests are representative of large areas (this assumption has not yet been verified), breeding success is likely higher than assumed in some earlier studies but is still too low to sustain populations with adult survivorship between 0.83 and 0.93 (see Section 3.1.4 – Survivorship; Cam et al. 2003, Peery et al. in prep.).

3.1.4 Survivorship

Little definitive information is available on annual survival or mortality rates in marbled murrelets. Two estimates for adult survivorship have been developed from mark-recapture data in southern British Columbia (Cam et al. 2003): 0.93 (95% Confidence Interval [CI] 0.0.85-0.99; mist net sample alone) and 0.83 (95% CI 0.72-0.90; mist net and dip net sample combined). Large confidence intervals resulted from low recapture rates. Presumably, dip net captures alone were much lower than 0.83 (but were not reported), since the combined sample was 0.83 and the mist net only sample was 0.92.An estimate of 0.82 (95% CI 0.61-0.93) was developed from mark-recapture analysis of dip net sampling in central California (Peery et al. in prep.). True adult survivorship was likely higher than that estimated from dip net captures alone due to various biases, including: recapture avoidance, dispersal, immigration, or mortality. Mist-net captures may have fewer effects from recapture avoidance but may favor successfully breeding adults, especially males (Vanderkist et al. 1999, Cam et al. 2003). Until definitive data are available, adult survival is assumed to be in the range of 0.83-0.93, and may vary between areas due to natural and anthropogenic causes. Previously, population modeling used the estimated value of 0.85 (range 0.81-0.88), derived from other alcids (Beissinger 1995a, Beissinger and Nur 1997).

No data are available on sub-adult survival, but earlier studies used an estimated 88% of adult survival based on estimates in other alcids (Beissinger 1995a, Beissinger and Nur 1997). In southern British Columbia, juvenile survival of 0.86 (95% CI 0.73-1.00) was examined for individuals in the 2-3 months after fledging (Parker et al. in prep.). If this rate is extended over 1 year, juvenile survival could be estimated to be 0.51, but the assumption of

constant survivorship over the first year of life is probably invalid (Parker et al., in prep.). Earlier studies used estimated juvenile survival as 71% of adult survival (Beissinger 1995a, Beissinger and Nur 1997). In summary, no definitive data exist on juvenile and subadult survival rates.

3.1.5 Non-Breeding Adults

Among various species of alcids, it has been found that a small proportion of adults (5-10%), capable of breeding do not breed in a given year (e.g., Harris and Wanless 1995). The proportion of non-breeding adult marbled murrelets has not been well estimated in studies to date due to lack of direct observation and to the difficulties of interpreting available data from brood patches, physiology, replacement eggs, and telemetry (McFarlane Tranquilla et al. 2003a,b,c). Some researchers suspect a large proportion of non-breeding adults in certain populations (e.g., central California), but estimates can range from about 5% to about 70% depending on the year and how data are handled and interpreted (Bradley 2002, Burger 2002, Beissinger and Peery 2003, McFarlane Tranquilla et al. 2003c,d, in press). Theoretically, the proportion of non-breeding adults might be affected by compromised nesting conditions (Peery et al. in prep.), but this has yet to be demonstrated in central California. In most alcid populations, it appears that most adult females lay eggs each year, although increased non-breeding may occur when food availability is low as, for example, in strong El Niño years (see Section 5.1). Earlier studies used 0.90 for the proportion of adults that breed, based on available data for other alcids (Beissinger 1995a, Beissinger and Nur 1997). Cam et al. (2003) used a minimum of 0.80 and a maximum of 0.95 in their population modeling work to bracket likely values. Until more information is available, we suggest that low levels of non-breeding adults ($\leq$0.10) should be considered for marbled murrelet populations during most years, as found in other alcids. However, during years of severe food-web perturbation (e.g., El Niño), a greater proportion of adults may not breed, perhaps 50% (see Section 3.5).

3.1.6 Other Characteristics

Burger (2002) and Lank et al. (2003) also reviewed information on a number of other demographic characteristics:

- Generation time – mean 10 years.

- Age class structure and proportion of breeders: nonbreeders – difficult to determine, but adults likely comprise the vast majority of individuals in the population.

- Productivity and population growth – models can show sustaining or non-sustaining populations depending on levels of fecundity and subadult and adult survival used.

- Critical life cycle stages – although modeled population growth is affected primarily by adult survival, as well as subadult survival, low breeding success can limit populations.

Additional consideration of various aspects of population modeling is presented in Section 3.5 (Demographic Modeling) of this report.

3.2 Distribution and Densities of At-Sea Populations

Marbled murrelets occur primarily within 3 miles (5 km) of shore along the coast of western North America from about 60°N to 34°N (Sealy et al. 1982, Carter and Erickson 1992, Erickson et al. 1995, Nelson 1997, Piatt and Naslund 1995; also Section 5.1 – Marine Habitat Characteristics). Significant features of their global distribution are:

- The northern extreme of the breeding range occurs in Bristol Bay and Cook Inlet (latitude ~60°N), but the northern extreme of occurrence is in the northern Bering Sea and southern Chukchi Sea (latitude ~65-70°N), with 1 record in Russia on the Chukchi Sea coast.

- The western extreme of the breeding range and occurrence is found at Attu Island in the western Aleutian Islands (longitude ~170°E) where murrelets are regularly noted and likely breed in small numbers. Marbled or long-billed murrelets have been noted at the Commander Islands in nearby Russia, but species identity is not clear. Long-billed murrelets are not known to currently occur or breed in the Aleutian Islands, but small numbers may not have been detected.

- Eastern extremes of the breeding range are found in far inland nesting areas in southern British Columbia and northern Washington (longitude ~120° W).

- The southern extreme of the breeding range occurs in central California (latitude ~37° N) but the southern extent of occurrence is in southern California and northern Baja California, Mexico (~32°N latitude).

- Marbled murrelets occur farther than 3.1 miles (5 km) from shore in significant numbers in southern Alaska.

On the local scale, at-sea densities of the murrelet vary considerably. During the breeding season, marbled murrelets are primarily found within daily commuting distance of tree-nesting habitats in coastal old-growth coniferous forests up to 12-62 miles (20-100 km) inland (central California to Kodiak Island) or ground-nesting habitats within similar or shorter distances of the coast (Aleutian Islands and Alaska Peninsula, and overlapping with tree-nesting habitats from Kodiak Island to British Columbia) (Day et al. 1983, Mendenhall 1992). During the nonbreeding season, murrelets in Alaska and British Columbia disperse more widely (but do not appear to undergo long-distance migration), and their distribution depends more on availability of preferred winter foraging habitat and prey, often located in inner protected waters within inlets and fiords (Forsell and Gould 1981, Burger 1995). In California (and to a lesser extent Oregon, Washington, and southern British Columbia), most or at least some murrelets are resident year-round near nesting areas and visit nesting areas during the non-breeding season (Carter and Sealy 1986, Carter and Erickson 1992, Rodway et al. 1992, Speich et al. 1992, Nelson et al. 1992, Naslund 1993). From a large-scale perspective, marine abundance is highest between Kodiak Island and Cook Inlet, AK, and the southwest coast of Vancouver Island, BC. Abundance is lower in: (1) the Aleutian Islands and Alaska Peninsula in the northwestern portion of the range; and (2) from Washington to central California in the southern portion of the breeding range. Very small numbers of murrelets are found in Bristol Bay, northern Bering Sea, south-central and southern California, and northern Baja California, Mexico.

Compared to British Columbia and Alaska, murrelet densities in coastal Washington, Oregon, and California are lower, and there may be more discontinuities in their distribution. However, murrelet at-sea densities also are quite variable in Alaska and British Columbia, with major areas of population concentration and local areas of higher and lower densities (Burger 1995, Piatt and Naslund 1995). Between areas of distributional discontinuity in the 3-state area, moderate densities reflect concentrations of murrelets within geographic sub-populations associated with available breeding habitats in old-growth forests in northern Washington (Zones 1 and 2), central Oregon (Zone 3), northern California (Zone 4), and central California (Zone 6). Discontinuities appear to be related to breaks in available nesting habitat in old-growth forests from logging and natural factors, as well as little use of large river mouths for foraging.

The size of the entire marbled murrelet population in North America is not known with certainty because their secretive nesting habits make them difficult to census on land. As a result, all population estimates are based on at-sea surveys. Ralph et al. (1995) estimated the North American population

at 300,000 birds. Recent updated information for several regions increases the North American estimate to more than 600,000 birds (reviewed in DeGange 1996), and possibly closer to 1 million (Table 3.2-1). The following is a more detailed summary of the distribution and abundance of marbled murrelets north to south by state, province, and Conservation Zone (see map, Figure 2.2-1).

3.2.1 Alaska

Except for natural gaps in the deep waters between most of the Aleutian Islands, between Kodiak Island and the mainland, and between small, patchy numbers in the east Bering Sea (coast of Alaska), the distribution of marbled murrelets in Alaska (from the northern Gulf to British Columbia) is essentially continuous wherever suitable feeding and breeding habitats occur (Mendenhall 1992, Piatt and Naslund 1995). Southeast Alaska is the region having, by far, the highest numbers (Table 3.2-1). From a small-scale perspective, densities at selected areas in southeast Alaska in June 1993 were 59.6 birds per square mile (23 birds per km^2) in Glacier Bay, and 101.1 birds per square mile (39 birds per km^2) in Icy Strait; although for 8 surveys conducted in Icy Strait between June and August in each of 1993, 1994, and 1995, the average was 58.0 birds per square mile (22.4 birds per km^2) (summarized in DeGange 1996).

Piatt and Naslund (1995) summarized marbled murrelet abundance during the breeding season for most of Alaska based on data from surveys conducted in the 1970s to 1990s (Table 3.2-1). Although they considered their data incomplete for waters associated with Cook Inlet, Prince William Sound, and southeast Alaska, these areas were well surveyed during the late 1980s to 1990s by Agler et al. (1998). These various sources allow development of a more reliable estimate for the size of the at-sea population for the state. The population sizes, as summarized for different areas from these studies (Table 3.2-1), indicate that about 860,000 marbled murrelets were present at-sea off the Alaskan coast from the 1970s through the 1990s (but see Section 3.2.5 for qualifications). With approximately 3,170 miles (5,100 km) of coastline (including both sides of Kodiak and Alexander Archipelagos, but 1 side only for other islands), the number of murrelets per mile of coastline was about 272 birds (169 birds per km). These numbers do not take into account the large amount of coastline represented by bays and fjords.

Table 3.2-1. At-sea abundance of marbled murrelet during the breeding season at different locations within their pelagic range[1].

Region[2]	Year(s)	Density birds/km²	Number of Birds[3]	95% Confidence Interval	Source
Alaska					
Offshore	1970s-1990s	--	9,800		Piatt and Naslund (1995)
Bering Shelf	1970s-1990s	--	700		Piatt and Naslund (1995)
Aleutian Islands	1970s-1990s	--	400		Piatt and Naslund (1995)
Alaska Peninsula	1970s-1990s	--	2,900		Piatt and Naslund (1995)
Kodiak Archipelago	1970s-1990s	--	21,900		Piatt and Naslund (1995)
Lower Cook Inlet[4]	1994	3.0	40,200		Agler et al. (1998)
Prince William Sound[5]	1989-94	11.6	104,600		Agler et al. (1998)
Southeast Alaska[6]	1994	19.2	678,600		Agler et al. (1998)
Total	1970s-1990s	--	859,100		
British Columbia Total[7]	1980s-1990s	--	66,500		Burger (2002)
Washington to California					
CZ-1 (WA)	2000	1.61	5,600	2,700- 8,900	Huff et al. 2003
CZ-2 (WA)	2000	0.46	800	500- 1,200	Huff et al. 2003
CZ-3 (no. OR)	2000	4.25	6,700	4,000-10,100	Huff et al. 2003
CZ-4 (so. OR/no. CA)	2000	4.22	4,900	3,800- 9,500	Huff et al. 2003
CZ-5 (no. CA)	2000	0.09	100	300	Huff et al. 2003
CZ-6 (cent. CA)[8]	2000	--	500	338 - 728	Peery (pers. com.)
Total	2000	2.06	18,600		Huff et al. 2003
CZ-1 (WA)	2001	2.55	8,900	5,800-11,900	Huff et al. 2003
CZ-2 (WA)	2001	1.03	1,700	500- 3,800	Huff et al. 2003
CZ-3 (no. OR)	2001	4.77	7,500	5,500- 9,300	Huff et al. 2003
CZ-4 (so. OR/no. CA)	2001	3.33	3,900	3,000- 6,700	Huff et al. 2003
CZ-5 (no. CA)	2001	0.13	100	18 - 300	Huff et al. 2003
CZ-6 (cent. CA)[8]	2001	--	600	441 - 920	Peery pers.com.)
Total	2001	2.52	22,700		Huff et al. 2003
CZ-1 (WA)	2002	2.77	9,700	6,000-13,800	Huff et al. 2003
CZ-2 (WA)	2002	1.56	2,600	800- 3,800	Huff et al. 2003
CZ-3 (no. OR)	2002	3.97	6,300	4,000-10,000	Huff et al. 2003
CZ-4 (so. OR/no. CA)	2002	4.17	4,900	3,500- 6,400	Huff et al. 2003
CZ-5 (no. CA)	2002	0.28	300	30 - 400	Huff et al. 2003
CZ-6 (cent. CA)[8]	2002	--	600	487 - 809	Peery (pers. com.)
Total	2002	2.69	24,400		Huff et al. 2003
North America Total[9]	--	--	947,500		--

[1] *Except for British Columbia, estimates are not adjusted for the number of birds expected on land at nest sites.*
[2] *Conservation zone = "CZ." "Year" represents the approximate period that surveys were conducted.*
[3] *Numbers rounded to nearest 100 birds.*
[4] *Estimate of Agler et al. 1998, (Table 2; 58,227* Brachyramphus murrelets*) adjusted for observed proportion of Kittlitz's Murrelets,* B. brevirostris *= 30.9%; Alger et al., Table 3.*
[5] *Estimate of Agler et al. (1998, Table 2; 113,652* Brachyramphus murrelets*) adjusted for observed proportion of* B. brevirostris *= 8.0%, Agler et al., Table 3.*
[6] *Estimate of Agler et al. (1998, Table 2; 687,061* Brachyramphus murrelets*) adjusted for observed proportion of* B. brevirostris *= 1.2%, Agler et al. Table 3.*
[7] *Midpoint of the population range (55,000-78,000 birds)* estimated *by Burger (2002); the estimate includes all birds representing both the breeding and nonbreeding population components.*
[8] *CZ6 was not surveyed in 2000, 2001, or 2002 under the Effectiveness Monitoring Program. However, this zone was surveyed by Peery et al. in these years. Values are from: Z. Peery, pers. comm., November 20, 2003 and are revised from the data presented in Peery et al. (2002 and 2003). These values were added to CZ1-CZ5 totals to estimate the sum total number of birds during 2000, 2001, and 2002 for the 3-state area.*
[9] *The North American total was calculated using the 3-year average estimated for Washington, Oregon, and California (i.e., mean = 21,900 birds).*

3.2.2 British Columbia

Marbled murrelets are widely distributed throughout most coastal areas in British Columbia (Rodway et al. 1992; Burger 1995, 2002). Densities are highest along the southwest coast of Vancouver Island and the east coast of Morsesby Island of the Queen Charlotte Islands, and lowest along the southern coast of the mainland and east coast of Vancouver Island (adjacent to the Straits of Georgia; Burger 2002). However, large areas have yet to be surveyed, and provincial estimates are very rough. Natural gaps occur in deeper waters between: (1) the Queen Charlotte Islands and the British Columbia mainland and southeast Alaska (i.e., across deeper waters in Hecate and Dixon Straits); and (2) Vancouver Island and the British Columbia mainland and the Olympic Peninsula, Washington (i.e., across deeper waters in Johnstone Strait, Straits of Georgia, and Juan de Fuca Strait). The estimated total number of birds present in the province during the breeding season (May to July) is 55,000 to 78,000 (Burger 2002). Using the mid-point of this estimate (66,500 birds), with about 1,516 miles (2,440 km) (straight line) of coastline (including both sides of Vancouver and Queen Charlotte Islands), the estimated number of murrelets per mile of coastline is about 43 birds (27 birds per km). This is an overestimate of the number at sea at any given moment, because corrections were made for numbers presumably on land at their nest sites.

3.2.3 Washington

As estimated from the most recent data from the Northwest Forest Plan (NWFP) Effectiveness Monitoring Program (EM Program) (Huff et al. 2003), the population of marbled murrelets present at sea along the Washington coast during the breeding season (May to July) for 3 years (2000 to 2002) averaged 9,800 birds (range 6,400 to 12,300; Table 3.2-1). Given that the length of the Washington coastline (not including islands, but including both sides of Puget Sound) is about 575 miles (925 km), there are an estimated 17.1 murrelets per mile of coastline (10.6 birds per km).

Murrelets are widely distributed in Washington, as indicated by a number of surveys. Varoujean and Williams (1995) estimated marbled murrelet densities along the Washington coast from aerial surveys in September 1993. Along the outer coast, densities between the southern portion, roughly from the Columbia River to Destruction Island, were substantially lower (0.8-2.3 birds per square mile [0.3-0.9 birds/km^2]) than densities in the northern portion, from Destruction Island to Cape Flattery (13.2-23.8 birds per square mile [5.1-9.2 birds/km^2]). Densities along the southern border of the Strait of Juan de Fuca ranged from 2.3 to 21.0 birds per square mile (0.9 to 8.1

birds/km^2). Breeding season densities observed by Speich and Wahl (1995) in northern Puget Sound (including the San Juan Islands and southern Georgia Strait) were not greater than 1.1 birds per square mile (0.43 birds/km^2) in any section.

Major gaps in the at-sea distribution of murrelets in Washington occur: (1) in the British Columbia-Washington border region, which includes the deep waters of the Strait of Juan de Fuca between the Olympic Peninsula and Vancouver Island and the area between the San Juan Islands and Howe Sound, British Columbia (i.e., mouth of the Fraser River and the city and suburbs of Vancouver); (2) southern Puget Sound (i.e., off metropolitan areas of Seattle, Tacoma, Olympia, and surrounding urban areas); and (3) the southwestern coast (i.e., north of the Columbia River and off Grays Harbor and Willapa Bay).

3.2.4 Oregon and California

Based on data from the EM Program for the NWFP, the estimated population size present at sea along the Oregon/California coast during the breeding season (May to July) over 3 years (2000 to 2002) averaged 12,133 birds (range 12,100 to 12,200; Table 3.2-1). Given that the length of the Oregon/California coastline within the range of the murrelet is about 705 miles (1,135 km), there were an estimated 17.2 murrelets per mile of coastline (10.7 birds per km).

Marbled murrelets are widely distributed in Oregon and northern California waters (as far south as southern Humboldt County, California) but numbers become small and patchy south of this area to Santa Cruz County (Nelson et al. 1992; Strong et al. 1995, 2003b; Varoujean and Williams 1995; Carter and Erickson 1992; Ralph and Miller 1995; Becker et al. 1997). Survey results from Varoujean and Williams (1995) and Ainley et al. (1995a) suggest that the highest densities along the Oregon coast occur from Cape Lookout to Brookings (14.0-17.6 birds per square mile [5.4-6.8 birds/km^2]), with somewhat lower densities (8.3-13.0 birds per square mile [3.2-5.0 birds/km^2]) in the more northern part of the state, from Cape Lookout to the Columbia River. For California, surveys by Ralph and Miller (1995) showed highest densities (5.2-22.8 birds per square mile [2.0-8.8 birds per km^2]) from the California/Oregon border to Table Bluff (near Loleta, Humboldt County).

Overall, there are 3 major gaps in the distribution of murrelets off Oregon and California. In the north, the gap along the Washington coast beginning at Destruction Island extends south past the Columbia River to Tillamook Head, Oregon. The next major gap in the at-sea distribution of murrelets is

found between Humboldt and San Mateo counties, California. Within this gap, small numbers of murrelets recently have been found to breed in small patches of remaining nesting habitat in Mendocino County (where few if any birds were thought to still breed). Ralph and Miller (1995) recorded moderate to low densities (1.8-3.9 birds per square mile [0.7-1.5 birds per km^2]) from Table Bluff (Humboldt County) to Albion (Mendocino County). No breeding and little possible forest nesting habitat is known in Sonoma, Marin, and San Francisco counties. Surveys by Ralph and Miller (1995) did not record any birds from Albion (Mendocino County) to Half Moon Bay (San Mateo County). In the vicinity of Half Moon Bay, densities of 4.72 birds per square mile (1.82 birds/km^2) were recorded by Ralph and Miller (1995). A small number of murrelets occurs off the coastline adjacent to the old-growth forests in parks within San Mateo and Santa Cruz counties (referred to as the central California or Zone 6 population) during the breeding season.

From southern Santa Cruz County through Monterey County, few murrelets occur during the breeding season, but low numbers occur regularly in this area during the late breeding season (as early as June) and non-breeding season (Peery et al. in prep.; E. Burkett, pers. comm.). This area represents the southern end of available nesting habitats (i.e., old-growth forests within foraging distance of the ocean). Breeding has not been documented in this area but may have occurred in the past.

Farther south, the numbers decrease to very low levels in the breeding season. During the late breeding and non-breeding seasons, small numbers of murrelets occur farther south to Point Conception and in some years as far as just south of the U.S.-Mexico border (Carter and Erickson 1992, Erickson et al. 1995).

3.2.5 North American Population Size Estimate

The abundance estimates provided in Table 3.2-1 have not been adjusted for 1 or more of the effects of 5 potentially biasing factors inherent to pelagic surveys:

1. Flying Birds (Flux) – Counting flying birds usually results in population overestimation, which for fast-flying birds such as murrelets, typically results in an inflation of 25 to 40% (Spear et al. 1992, Spear and Ainley 1997, Clarke et al. 2003). However, the number of flying birds noted during murrelet surveys is generally small (e.g., <2.5%). These birds are usually far from the survey line and do not contribute much to observed

densities of marbled murrelets; thus, this potential bias is thought to be low (Strong et al. 1995; Spear et al. in prep.; Strong, unpubl. data).

2. Double Counting – Flying birds can land ahead (and in the survey strip) of the survey vessel and be recounted, possibly resulting in population overestimation if the bird was counted earlier. Since murrelets spend most of their time on and under the water and birds that flush usually do so while ahead of survey vessels, this bias is probably minimal.

3. Birds that are Underwater – Pelagic counts can miss murrelets that are either foraging underwater or that submerge (escape-dive) to avoid the approaching survey vessel. However, this is not likely to have been a serious problem affecting the accuracy of counts reported in Table 3.2-1 because: (1) escape-diving usually does not occur until the boat is very close and murrelets have already been recorded; and (2) murrelets spend most of their time resting and preening on the water surface between diving bouts.

4. Dispersal – After nest failure or chick fledging, adults and juveniles may disperse at-sea to locations some distance from the foraging areas used during the nesting period. Post-breeding dispersal usually does not occur until after the molt period in late-summer when adults are flightless (i.e., after the breeding season surveys reported in Table 3.2-1), but some birds will disperse earlier. This behavior may affect overall population size estimates because there may be a net influx or outflux within survey areas. Thus, counts in any given survey area could be overestimates or underestimates. Only for counts in Conservation Zone 6 do we suspect a net outflux within the survey area. Given the low numbers of birds breeding in Conservation Zone 6, dispersal may have a significant effect on estimates.

5. Breeding Birds at Nest Sites – Most surveys are conducted during the breeding season when an unknown proportion of the incubating adults may be at nest sites, leading to underestimation. However, breeding is not highly synchronized, high breeding failure rates occur, and incubation primarily occurs in the early part of the survey period. These factors reduce underestimation.

The magnitude of the effect of these 5 potential biases on the abundance estimates for any of the 4 regions (Alaska, British Columbia, Washington, and Oregon/California) is unknown. For analysis purposes in this review, we have assumed that negative and positive influences counter-balanced one another. However, more work is needed to evaluate these biases.

We determined that the current estimate for the North American population of marbled murrelets is 947,500 birds (Table 3.2-1), of which 90.7% (859,100 birds) are found in Alaska, 7.0% (66,500) in British Columbia, 1.0% (9,800) in Washington, and 1.3% (12,100) in Oregon and California. For the latter 12,100 birds, roughly 62% (7,502) occur in Oregon and 38% (4,598) in California. Thus, about 0.8% of the total population is estimated to breed in Oregon, and 0.5% in California.

3.3 Population Trends From Past Studies

Consistent population surveys and the use of standardized protocols have not been conducted until fairly recently for marbled murrelets. This section describes the population trend information that can be obtained from past studies of the marbled murrelet but excludes trends determined from demographic modeling.

3.3.1 Trends in the 3-State Area

Quantitative data on murrelet abundance are lacking at the time scale corresponding with industrial logging of most murrelet habitat in the lower 48 states (1850-1980), but it is strongly suspected that numbers declined greatly during this period (Carter and Erickson 1992, Marshall 1988, Nelson 1997, Ralph 1994, Service 1997). A number of monitoring programs have been initiated since 1989, including efforts in all 6 Conservation Zones; however, few of these data are currently available to assess trends. The only published trend available for the 3-state area is from the Oregon coast, where standardized coastline surveys were conducted from 1992 through 1999. From these data, Strong (2003a) described an abrupt decline in the central Oregon murrelet population (Zone 3), which first became apparent in 1996. From 1997 through the present, near-shore densities of murrelets have not changed appreciably on the Oregon coast (Strong 2003a).

The EM Program for the NWFP united the various at-sea monitoring programs in Zones 1 through 5 into a standardized plan using directly comparable methods (Bentivoglio et al. 2002). The EM Program is of a relatively short time series that does not allow trends to be determined as of yet (Huff et al. 2003). Over the 2000-2002 implementation period of the program, average population estimates have increased in Zones 1 and 2, remained relatively stable in Zones 3 and 4, and remained extremely low in Zone 5 (Table 3.2-1). Estimates in Zone 1, and to a lesser extent in Zone 2, may face a confounding factor if substantial numbers of murrelets from British Columbia immigrate into these zones late in the survey period (Burger 1995, Raphael et al. 2000a, Speich and Wahl 1995). However, most of the

increase in numbers in northern Puget Sound occurs in fall and winter (Speich et al. 1992), after the surveys for the EM Program are complete. Peery et al. (2002, 2003, in prep.) also provide population size estimates for Zone 6 in 1999 (502; CI 327-769), 2000 (500; CI 345-724), 2001 (553; CI 403-757), and 2002 (619; CI 476-805). Mean estimates have increased slightly over the study period.

Because confidence intervals around these estimates are large and the time series short, patterns within zones do not statistically constitute trends. A time series of 5 or more years is estimated to be necessary to assess trends in these populations (Beissinger et al. 1999, Strong 1997).

3.3.2 Trends in Alaska and British Columbia

The more complex marine environment of the marbled murrelet throughout much of British Columbia and Alaska adds to the difficulty in estimating and monitoring populations in these areas. The best long-term data set is from the Prince William Sound area, where there is clear evidence of a significant decline since 1972 (Burger 2002, Klosiewski and Laing 1994, Kuletz et al. 1997, Irons et al. 2000) (Table 3.3-1). A combination of long-term ecosystem changes (due to fishing pressure and marine climate) and mortality of many thousands of murrelets during the *Exxon Valdez* oil spill (Ford et al. 1996, Carter and Kuletz 1995) are considered the major causes of decline. In Clayoquot Sound, BC, a significant decline (~40%) in marbled murrelet population estimates occurred over a 10- to 11-year period from 1982 to 1992-93 and was attributed primarily to loss of nesting habitat (Sealy and Carter 1984, Kelson et al. 1995). Despite some concern that the difference in estimates between 1982 and 1992-93 might reflect undocumented effects of El Niño or annual variability (Burger 2000), various subsequent surveys in Clayoquot and Barkley sounds suggest continuing decline and have confirmed that abundance is lower than in the early 1980s (Burger 2002, Kelson and Mather 1999, Mason et al. 2002).

For the entire marbled murrelet range, since 1972, major declines (22-73%) in populations over a period of a decade or more have been documented in Alaska, British Columbia, and Oregon with no evidence of increase (Table 3.3-1). Only in Oregon has a major decline been verified with annual standardized data (Strong 2003a,b).

3.4 Morphological and Genetic Variation

The range of the marbled murrelet extends from central California north through British Columbia and southeast Alaska to the Aleutian Islands, an

Table 3.3-1. Summary of studies addressing change in marbled murrelet abundance over the past 30 years.

Area	Period	Method	Trend	Source
Alaska	1974-1989	Christmas counts	Possible 50% decline	Piatt and Naslund 1995
	1972-1991	Marine surveys	67-73% decline	Klosiewski and Laing 1995
	1971-1993	Christmas counts	No clear change	Hayward and Iverson 1998
	1989-1998	Marine surveys	No recovery after *Exxon Valdez* spill	Lance et al. 2001
	1972-1993	Marine surveys	Decline	Agler 1999
	1984-1998	Marine surveys	No clear change	Irons et al. 2000
British Columbia	1982-1993	Marine surveys	40% decline, Clayoquot Sound	Kelson et al. 1995
	1982-1996	Marine surveys	22% decline, Clayoquot Sound	Kelson and Mather 1999
	1987-1993	Marine surveys	50% decline, Barkley Sound	Burger 1995
	1996-2000	Marine surveys	Possible decline	Mason et al. 2002
	1982-2002	Review of studies	22-44% decline	Burger 2002
Washington	1996-1999	Marine surveys	No evidence of change	Thompson 1997-1999
	1972-1993	Marine surveys	Possible decline	Speich and Wahl 1995
Oregon	1992-1996	Marine surveys	>50% decline	Strong 2003a
	1997-2003	Marine surveys	No clear change	Strong 2003a,b
California	1995-2001	Occupied detections	Probable decline, Santa Cruz mts.	Suddjian 2001

Source: adapted from Lank et al. 2003.

area that encompasses a wide variety of conditions in both the marine and terrestrial environments. To breed successfully throughout this range, a number of behavioral and morphological adaptations are required. This section summarizes the morphological and genetic variation found throughout the range of the murrelet.

3.4.1 Morphological Variation

Few studies of morphological variation have been published for marbled murrelets (Nelson 1997). Pitocchelli et al. (1995) conducted a detailed mensural comparison of ground- and tree-nesting murrelets in Alaska; they analyzed 6 external measurements and 17 skeletal dimensions among Kittlitz's and marbled murrelets collected during the breeding season. Analysis of variance (not controlling for sampling location) indicated slight but statistically significant differences between ground- and tree-nesting murrelets in wing, gape, bill depth, mandible, and tarsus; however, the 2 nesting types were indistinguishable using principal components analyses.

Hull et al. (2002) compared variation in body mass among murrelets from Desolation Sound and Mussel Inlet, British Columbia; although significant effects of sex, time of day, and season were found, mass did not differ between sampling sites. Murrelets from these areas were significantly lighter than birds from Langara Island (Sealy 1975a), but comparisons were complicated by sampling method and year (Hull et al. 2002). No other morphological data have been published for murrelets. Thus, tree- and ground-nesting murrelets in Alaska do not appear to differ morphologically,

but the extent of geographic variation in morphology throughout the species range is currently unknown.

3.4.2 Genetic Variation

This section summarizes data on genetic variation and population structure from studies conducted on marbled murrelets and other seabirds. Additional information is presented on the genetic and ecological delineation of distinct populations for the marbled murrelet.

3.4.2.1 Studies of Murrelets

Studies of neutral genetic variation[1] in marbled murrelets were first published in 1995 and are ongoing. These studies include analyses of mitochondrial DNA (mtDNA), allozymes, introns, and microsatellites[2].

Genetic Variability and Inbreeding

Estimates of genetic variability in all molecular markers that have been screened for marbled murrelets are similar to other seabirds, with no evidence of either severe recent (Holocene) population bottlenecks[3] or inbreeding (Friesen et al. 1996b, Congdon et al. 2000, Friesen and Piatt 2003).

[1] Neutral variation is genetic variation that is not affected by selection; it includes synonymous substitutions in coding genes, and substitutions in non-coding and non-regulating genes. It is useful for population studies because it can be used to measure the effects of genetic drift, gene flow, and non-random mating without the need to correct for selection (which is extremely difficult to measure).

[2] mtDNA is very useful in population-level studies because its mutation rate is higher than most single copy nuclear genes. It is therefore a sensitive indicator of recent population dynamics. mtDNA is also non-recombining, which simplifies analyses. Furthermore, it is maternally inherited and effectively haploid, which means that the effective population size[12] is one quarter that of nuclear DNA. mtDNA is therefore a sensitive indicator of population bottlenecks and restrictions in gene flow. The mitochondrial control region is especially useful since it is the most rapidly evolving segment of the mitochondrial genome. Mitochondrial genes are easily sequenced using the polymerase chain reaction (PCR). However, mtDNA represents only a single locus. *Allozymes* (metabolic enzymes that can be screened using electrophoresis and special stains) and *introns* (non-coding regions that interrupt the coding sequences of genes) are nuclear markers that are generally unlinked, and so provide independent indices of the genetic relationships among individuals and populations. Allozymes are useful because large numbers of loci can be screened quickly; however, they are relatively slowly evolving and they require fresh or freshly frozen tissue for analysis. Introns evolve more rapidly and show high levels of variation. They can be analyzed using PCR, so they can be screened using small and/or degraded tissue samples; however, analyses are slow and labor-intensive. *Microsatellites* are tandem repeats (generally 10-20 units) of short core sequences (usually 2-4 base pairs). They typically have very high mutation rates and can be screened rapidly using PCR, so they are very useful for comparisons of populations and individuals (e.g., parentage analysis); however, their mutation models are unclear, complicating their interpretation for population-level studies.

[3] A population bottleneck is a temporary but significant reduction in population size.

Population Genetic Structure

Friesen et al. (1996b) published a preliminary analysis of genetic variation among marbled murrelet populations as part of a taxonomic re-assessment of the long-billed murrelet (see Section 2.1). They compared variation in 39 allozyme loci and 1,045 base pairs (bp) of the mitochondrial cytochrome *b* gene among 43 marbled murrelets sampled between the western Aleutian Islands to Oregon. No statistically significant geographic structure (i.e., population structure or population differentiation) was found in cytochrome *b* (Φ_{st}[4] = 0.02, $P > 0.05$). Geographic variation in allozymes was moderate and statistically significant (F_{st}[5] = 0.09, $P < 0.01$), but sample sizes were too small to compare individual populations.

Congdon et al. (2000) published a more comprehensive analysis of genetic variation in marbled murrelets (see also Friesen et al. 1997). They compared sequence variation in 9 nuclear introns among 120 murrelets sampled between the western Aleutian Islands and northern British Columbia, and found moderate and statistically significant population structure (F_{ct}[6] = 0.09, $P < 0.001$). In pairwise comparisons of populations, murrelets from the Aleutian Islands were significantly differentiated from those elsewhere (most $P < 0.05$), whereas little or no differentiation was apparent among populations between the Alaskan Peninsula and northern British Columbia.

An ongoing study of genetic variation in marbled murrelets involves analysis of more rapidly evolving molecular markers (5 microsatellite loci and the mitochondrial control region) from murrelets sampled between the western Aleutian Islands and central California (Table 3.4-1) (Friesen et al. 2003, Friesen and Piatt 2003, Friesen et al. unpubl.). Preliminary results support previous indications that significant population genetic structure exists within marbled murrelets. Estimates of F_{ct} from mtDNA, introns, and microsatellites are all moderate and statistically significant (range = 0.075 to 0.15, Table 3.4-2; all $P < 0.001$) (Friesen et al. unpubl.). Murrelets in the Aleutian Islands have unique control region haplotypes[7], and murrelets in California have unique intron alleles and control region haplotypes (Friesen and Piatt 2003, Friesen et al. unpubl.); although these haplotypes/alleles do

[4] Φ_{st} is an index of the proportion of variation that is distributed among populations; it is generated by an analysis of variance on sequence differences among individuals, and theoretically can vary from 0 (indicating no differentiation among populations) to 1 (indicating that all variation represents differences among populations).

[5] F_{st} is an index of population genetic structure based on genotype frequencies. It represents the reduction in overall heterozygosity that results from individuals breeding within local populations. Like Φ_{st}, it can vary theoretically from 0 to 1.

[6] F_{ct} and Φ_{ct} are the equivalent of F_{st} and Φ_{st}, respectively, from hierarchical analyses of variance in which populations are grouped into regions.

[7] 'Haplotypes' are variants (~ alleles) of non-recombining, haploid genomes such as vertebrate mtDNA.

Table 3.4-1. Number and locations of marbled murrelets sampled for molecular markers.

Region	Sampling Site	Number
Western Aleutian islands	Attu Island	9
Central Aleutian islands	Adak Island	13
Eastern Aleutian islands	Dutch Harbor	15
Shumigan islands	Belkofski Bay	6
	Shumagin islands	4
	Koniuji Strait	2
Mitrofania Bay	Mitrofania Bay	10
Kodiak Island	Shuyak Island	14
Kachemak Bay	Kachemak Bay	16
Prince William Sound	Unakwik Fjord	10
Southeastern Alaska	Lemesurier Island	20
British Columbia	Desolation Sound	30
Northern California	Humboldt County	35
Central California	Santa Cruz	35
Total		**184**

Source: Friesen, unpublished data

not form distinct clades (phylogenetic groups) on the haplotype/allele trees, several occur at high frequency. Friesen et al. (unpubl.) also analyzed intron variation using STRUCTURE, a program that can help delineate genetic populations on the basis of deviations from Hardy-Weinberg and linkage equilibrium, independent of populations defined *a priori* by researchers (Pritchard et al. 2000). Results provide strong support ($P = 1.000$) that murrelets constitute 3 populations, with most birds from California forming a distinct cluster. Most genetic data for murrelets also demonstrate a significant isolation-by-distance effect[8], with pairwise estimates of F_{st} increasing with geographic distance between population pairs (Mantel's tests, $P < 0.05$) (Congdon et al. 2000, Friesen and Piatt 2003, Friesen et al. unpubl.). Thus, results of a number of studies, including several types of molecular markers and varying methods of data analysis, all indicate that statistically significant genetic structure exists in marbled murrelets, with populations from California and the Aleutian Islands differing both from each other and from populations in British Columbia and mainland Alaska. It is important to note, however, that samples from Washington and Oregon are not included in any of these analyses, and that sample sizes from some areas in the Aleutian Islands and California are low. Genetic divergence of Aleutian and Californian populations is consistent both with the lower population sizes and densities in these areas, and with their non-central locations within the world range.

[8]Isolation by distance is an increase in genetic divergence with increasing geographic distance between populations, usually attributed to a decrease in gene flow with distance.

Table 3.4-2. Estimates of F_{st} or its analog for various species of seabirds, based on either mtDNA or nuclear DNA.

Species	Location	Loci	F_{st} or Analog	Reference
MtDNA				
Common murre	north Atlantic	control region	0.00	Moum & Árnason 2001
Ancient murrelet	north Pacific	control region	0.00	Pearce et al. 2002
Grey-headed albatross	circum-Antarctica	control region	0.00	Burg & Croxall 2001
Thick-billed murre	north Atlantic	cytochrome *b*	0.001	Birt-Friesen et al. 1992
Common murre	north Pacific	control region	0.011	Friesen & Piatt 2003
Razorbill	north Atlantic	control region	0.042*	Moum & Árnason 2001
Northern fulmar	northeastern Atlantic	control region	≤0.054	Burg et al. 2002
Marbled murrelet	north Pacific	control region	0.09*	Friesen & Piatt 2003
Red-legged kittiwake	north Pacific	control region	0.17*	Patirana et al. 2002
Short-tailed shearwater	southern Australia	RFLP[a]	0.19	Austin et al. 1994
Lesser black-backed gull	complete range	control region	0.24*	Liebers & Helbig 2002
Pigeon guillemot	north Pacific	control region	0.37*	Friesen & Piatt 2003
Sooty tern	circum-tropical	control region & RFLP[a]	0.38[b]	Avise et al. 2000
Fairy prion	Australia	RFLP[a]	0.56*	Ovenden et al. 1991
Black-browed albatross	circum-Antarctic	control region	0.58*	Burg & Croxall 2001
Black guillemot	north Atlantic and Arctic	control region	0.80*	Kidd & Friesen 1998
Nuclear Loci				
Common murre	north Pacific	Introns	0.00	Friesen & Piatt 2003
Adelie penguin	circum-Antarctica	Microsatellites	0.00	Roeder et al. 2001
Atlantic puffin	northeast Atlantic	Allozymes	0.003	Moen 1991
White-capped albatross	Australia/New Zealand	Microsatellites	<0.016*	Abbott & Double 2003
Pigeon guillemot	north Pacific	Introns	0.03*	Friesen & Piatt 2003
Imperial shag	southern South America	Allozymes	0.06	Rasmussen 1994
Grey-headed albatross	circum-Antarctica	Microsatellites	<0.06	Burg & Croxall 2001
Cory's shearwater	Atlantic/Mediterranean	Allozymes	0.044	Randi et al. 1989
White-headed gulls	Arctic/subarctic	Allozymes	0.069[b]	Snell 1991
Great cormorant	Europe	Microsatellites	0.069[b]	Goostrey et al. 1998
Marbled murrelet	north Pacific	Microsatellites	0.075*	Friesen & Piatt 2003
Marbled murrelet	north Pacific	Introns	0.15*	Congdon et al. 2000
Pigeon guillemot	north Pacific	Microsatellites	0.11*	Friesen & Piatt 2003
Shy albatross	Australia/New Zealand	Microsatellites	<0.12*	Abbott & Double 2003

Table 3.4-2. Estimates of F_{st} or its analog for various species of seabirds, based on either mtDNA or nuclear DNA (continued).

Species	Location	Loci	F_{st} or Analog	Reference
Black-browed albatross	circum-Antarctica	Microsatellites	<0.15*	Burg & Croxall 2001
Cory's shearwater	Atlantic/Mediterranean	DNA fingerprints	0.18*	Rabouam et al. 2000
Great frigatebird	central Pacific	AFLP[c]	0.27*	Dearborn et al. 2003
Rock shag	south Atlantic	Allozymes	0.32*	Siegel-Causey 1997

[a]*Restriction fragment length polymorphisms.*
[b]*Not tested for significance.*
[c]*Amplification fragment length polymorphisms.*
**Statistically significant*

Studies to date have revealed little evidence of genetic differentiation between ground- and tree-nesting populations of marbled murrelets: all estimates of F_{st} and F_{ct} between ground- and tree-breeding populations are very low, and none are statistically significant (Pitocchelli et al. 1995, Congdon et al. 2000, Friesen and Piatt 2003). However, these studies are not conclusive since none removed the effect of range-wide geographic variation before testing for an effect of nesting habit. Furthermore, if variation in nesting habit evolved very recently, genetic differences between tree- and ground-nesting populations would not be detected using neutral molecular markers[1] since morphological and behavioral characters can evolve faster than neutral variation under strong selection. Thus, for purposes of conservation, we should be conservative and assume that tree- and ground-nesting populations are genetically different and ecologically inexchangeable until stronger evidence is available from molecular markers, heritability studies, or transplantation experiments.

Gene Flow

Gene flow between populations can be estimated indirectly using molecular markers; however, most methods make several assumptions that either do not hold in most species, including murrelets, or cannot easily be tested (e.g., Slatkin 1987, Birky et al. 1989, Beerli 1999, Congdon et al. 2000). Thus, no comprehensive estimates of gene flow or immigration based on molecular markers are available for murrelets. However, the probability that an individual is an immigrant into the population from which it was sampled can be estimated from its multilocus genotype using assignment tests[9] (e.g., Rannala and Mountain 1997; Palsbøll 1999; Wilson and Rannala 2003). Using this approach, Friesen and Piatt (2003) analyzed 120 marbled murrelets sampled between the Aleutian Islands and British Columbia. Three individuals (2.5%) had high probabilities of being immigrants into the site from which they were collected (Note that there are no confidence intervals on this number). Furthermore, the existence of private haplotypes/alleles at high frequency in samples from the Aleutian Islands and Californian (see above) suggests that gene flow between these areas and British Columbia/mainland Alaska is restricted. These results are consistent with the generally high natal philopatry of alcids (see Section 2.9).

[9]Assignment tests involve comparison of the mutli-locus genotype of an individual with baseline allele frequencies to determine the probability that the individual (or its immediate ancestors) originated within the population from which it was sampled or from another population.

Population History

Clues about a species' evolutionary and demographic history are contained within DNA sequences, and methods for decoding these clues are currently being developed (e.g., Excoffier et al. 1992, Templeton 1998). Congdon et al. (2000) analyzed mismatch distributions[10] for 9 introns for 120 marbled murrelets sampled between the Aleutian Islands and British Columbia. They inferred that murrelets in the Aleutian Islands diverged from mainland populations during isolation in separate glacial refugia during the late Pleistocene, and that murrelets expanded from these refugia and underwent secondary contact following recession of the glaciers. However, use of mismatch distributions to make inferences about the demographic history of a species makes several assumptions, such as no population subdivision or genetic recombination, which almost certainly do not hold for Congdon et al.'s data. More recently, Friesen and Piatt (2003) applied nested clade analysis[11] (Templeton 1998) to variation in the mitochondrial control regions of 80 marbled murrelets sampled between the Aleutian Islands and central California. They could not find any evidence of historical isolation, but concluded that murrelets probably survived the last glaciation in a single refugium (the location of which is unclear at present), and that populations in the Aleutian Islands and California represent Holocene range extensions, with population differences arising recently *in situ* due to restricted gene flow.

3.4.2.2 Studies of Other Species

In the absence of genetic data for murrelets from Washington and Oregon, and given the preliminary nature of data from California, it is useful to compare data for murrelets with findings for other seabird species. Genetic studies of seabirds have been accumulating over the past 2 decades, with indices of population genetic structure varying from 0 to 0.80 for mtDNA, and from 0 to 0.32 for nuclear loci (Table 3.4-1); values for marbled murrelets are in the low to mid range for mtDNA, and mid- to high range for nuclear loci. The most useful comparison involves the species that is most similar to the marbled murrelet in traits thought to determine population genetic structure. The most important of these traits are the

[10]Mismatch distributions are frequency distributions for the number of sequence differences between pairs of individuals. The shape of the distribution can provide information about the demographic history of a species (e.g., historical bottlenecks).

[11]Nested clade analysis is a recently developed method of genetic analysis that also can be used to make inferences about the demographic history of a species, including fragmentation, range expansion, restricted gene flow, and long-distance dispersal. It is based on the geographic distributions and substitutional relationships of alleles or haplotypes, and makes very few assumptions.

genetically effective population size[12]; geographic distribution of breeding sites; pattern and extent of gene flow[13]; and generation time. The species that is most similar to marbled murrelets in these respects is the pigeon guillemot (*Cepphus columba*), which has similar levels of genetic variability (a proxy for genetically effective population size) (Friesen and Piatt 2003), breeding distribution, and generation time. Friesen and Piatt (2003) compared variation in the mitochondrial control region, 3 introns, and 4 microsatellite loci among 202 pigeon guillemots sampled between southern California and the Aleutian Islands (excluding southeast Alaska and Washington). All 3 types of loci indicated that population genetic structure in this species is strong (global $\Phi_{ct} = 0.34$ for control regions; $\Phi_{ct} = 0.11$ for microsatellites and 0.02 for introns; all $P < 0.001$), with strong isolation-by-distance (Mantel's tests, all $P < 0.01$). Differences were strongest between 3 groups of populations: (1) Aleutian Islands to Prince William Sound, (2) British Columbia, and (3) Oregon and California. This pattern is broadly similar to that for marbled murrelets, and supports evidence that murrelets from California and the Aleutian Islands differ genetically from those elsewhere. In addition, results for pigeon guillemots suggest that genetic differences expressed by the California sample of marbled murrelets may extend north into at least Oregon.

3.4.2.3 Delineation of Distinct Populations

Methods for diagnosing distinct populations for conservation are the subject of current debate (reviewed in Moritz 2002). Originally, distinct populations were defined as subspecies. While this method may work for subspecies that exist in geographically isolated areas, many subspecies (or geographic populations) reflect segments of a species' range that are not geographically isolated from other parts of the range or are well connected by movements. Furthermore, molecular genetic studies have revealed that subspecies designations do not always reflect genetic differences between populations or local adaptations. Molecular markers are now used routinely in management and conservation. However, the relationship between statistically significant genetic differences and biologically meaningful genetic differences is

[12]The genetically effective population size, N_e, is the size of an idealized population that would contain the same amount of variation as observed within the real population. It can be visualized as the number of individuals actually contributing to the gene pool each generation. N_e is typically 1 to 2 orders of magnitude lower than the population census size (N_c) due to several factors, including biases in sex ratio, unequal family sizes, overlapping generations, and population bottlenecks.

[13]Four main models of gene flow have been described: in the *island model* of gene flow, populations occur on habitat islands and exchange migrants at random; in the *one dimensional stepping stone model*, populations occur in a linear distribution of habitat islands, and exchange migrants only (or primarily) with neighboring populations; in the *two dimensional stepping stone model*, populations occur in a two-dimensional distribution of habitat islands, and exchange migrants only (or primarily) with neighboring populations; and in the *isolation by distance* model, the species is distributed continuously, with gene flow declining with distance.

controversial. Most statistical tests of genetic differentiation involve tests for deviations of allele or genotype frequencies from random distributions, and are well-grounded in contemporary statistical theory. But do statistically significant genetic differences necessarily indicate biologically meaningful differences? Can biologically meaningful differences exist in the absence of statistically significant differences? And how do we avoid subdividing species into smaller and smaller management units as molecular and ecological data accumulate?

Various efforts have been made to address this issue:

- *Legal Definition*. In 1996, the Service defined "discrete vertebrate populations" as populations that are "...markedly separated from other populations of the same taxon...", and proposed that "Quantitative measures of genetic or morphological discontinuity may provide evidence of this separation" (61FR4722). Furthermore, the Service argued that a discrete population may merit special conservation status given "evidence that the discrete population segment differs markedly from other populations of the species in its genetic characteristics." However, it did not define "marked separation" or "marked difference." In 1991, the National Marine Fisheries Service argued that Pacific salmon stocks may be considered distinct population segments if they represent "evolutionarily significant units" (ESUs, see below), and that a stock may be considered an ESU if: (1) it is substantially reproductively isolated from other conspecific population units, and (2) it represents an important component of the evolutionary legacy of the species; again, "substantial" and "important" were not defined (but see below).

- *Genetic Viability*. Theoretically, the rate at which a species can adapt and evolve is directly related to its genetic variability. To the extent that local populations differ genetically, loss of a local population will reduce the species' genetic resources and therefore reduce the probability of long-term viability (i.e., increase the probability of extinction). The proportion of neutral variation[1] that is distributed among populations can be estimated using molecular markers[4-6]. However, the effect that loss of local populations would have on a species' viability cannot easily be modeled, and probably varies widely among species.

- *Evolutionary Independence*. Wright (1931) argued that 1 immigrant every second generation (regardless of the pattern of gene flow[13]) is sufficient to counter-act population differentiation through genetic drift; below this level of exchange, populations will tend to diverge. (In actuality, a grey zone exists around this level where differentiation may or may not

occur.) Numerous methods are available to estimate gene flow from molecular data; however, most methods make several assumptions (reviewed in Beerli 1999) that do not apply to marbled murrelets (or many other species) (Congdon et al. 2000) and/or require extensive data sets to generate reliable estimates (e.g., assignments[9]). No robust estimates of gene flow are currently available for marbled murrelets. Furthermore, strong selection can result in ecological or behavioral differences among local populations even in the presence of high gene flow.

- *Evolutionary Significance*. Given the above difficulties, a widespread genetic approach to determining the biological importance of local populations involves the delineation of ESUs. The idea is that an individual's DNA contains information about its evolutionary history, and that populations with unique evolutionary histories should be protected because their variation can never be recovered. Moritz (1994) provided objective criteria for delineating ESUs on the basis of neutral molecular variation; he defined ESUs as populations that are "reciprocally monophyletic for mtDNA alleles and also differ significantly for the frequency of alleles at nuclear loci." No evidence of ESUs according to the definition of Moritz (1994) has been found in marbled murrelets (excluding long-billed murrelets). However, many researchers consider Mortiz's (1994) definition too stringent for delineating genetically distinct populations; it also does not incorporate ecological or behavioral differences among populations (e.g., Crandall et al. 2000).

- *Demographic Independence*. Another approach to the question of the biological importance of genetic differences among populations relates to the demographic independence of populations. Populations that do not exchange migrants will respond separately to natural environmental changes or anthropogenic impacts. Such populations are defined as "stocks" in fisheries, and are generally managed separately. Moritz (1994) advocated defining *management units* (*MUs*) on the basis of "significant differences in allele frequencies, regardless of the phylogeny of alleles," the idea being that populations that exchange so few alleles as to be genetically different will also be demographically independent. Genetic studies of murrelets indicate that statistically significant differences in allele frequencies exist between populations in the Aleutian Islands, mainland Alaska/British Columbia, and California (Section 3.4.2.1). Thus according to Moritz's definition, marbled murrelets include at least 3 genetic MUs: (1) Aleutian Islands, (2) Alaskan Peninsula to British Columbia, and (3) California. Available ecological and behavioral data are consistent with this depiction because these major populations occur

in different marine ecosystems (with associated ecological/behavioral differences; see Chapter 5); significant gaps in breeding distribution occur between them (see above); and limited data show greater movements within than between them.

- *Genetic and Ecological Exchangeability*. Moritz's (1994) definition of an ESU has also been criticized because it does not incorporate adaptive differences among populations or ecological significance (Crandall et al. 2000). Crandall et al. argue that conservation should focus on preserving functional diversity, rather than historical legacy. Given that selection can act very rapidly and can result in local adaptation even in the presence of gene flow, they advocate including information about the ecological exchangeability of populations when delineating distinct populations for conservation. This information includes life history traits, morphology, habitat, quantitative genetic variation, and/or non-neutral molecular variation. Murrelets exhibit geographic variation in population size, feeding habits, and movement patterns that support the 3 management units defined by the genetic data reviewed above. In addition, murrelets have geographic variation in nesting habits, with those from Kodiak Island south nesting in trees, and those from the Alaskan Peninsula west nesting on the ground. Although the genetic basis for this difference is not known, Crandall et al. argue that "A lack of evidence for nonexchangeability should not be used to promote homogenization or other alterations of the gene pool, or to allow contraction or changes in the geographic distribution of populations." Thus, according to Crandall et al. (2000), tree- and ground-nesting populations also should be considered distinct populations until further evidence about exchangeability is available.

The ecological exchangeability of tree- and ground-nesting murrelets, and the genetic affinities of murrelets in Oregon and Washington should also receive additional investigation. Thus, given the available genetics data, murrelets should be considered to include at least 3 "distinct populations": (1) Aleutian Islands; (2) Alaska Peninsula to British Columbia; and (3) California. The possibilities that the Aleutian Islands, southern Alaska mainland (including the Alaska Peninsula) and California include 2 or more distinct populations each should be explored further.

3.5 Demographic Modeling of Marbled Murrelet Populations

This section describes the demographics model used to estimate marbled murrelet populations in the future. The section: (1) provides an overview of other modeling efforts; (2) describes the development of a new demographic

model for murrelets; and (3) presents model population projections over the next 40 years and extinction probabilities over the next 100 years.

3.5.1 Overview of Leslie Matrix Population Models

Given the many difficulties of obtaining and interpreting at-sea or forest census data on marbled murrelets over wide areas, marbled murrelet population trends and population viability analysis (PVA) are best determined with demographic models. These models serve to integrate available information on demography, stochastic variability, environmental variability, rare natural events, and human impacts into a coherent vision of expected future population trends and extinction probabilities. Such models are limited by our knowledge of demographic parameters and their variability over time and often exclude genetic limiting factors and future change in conditions. Until parameters are well estimated over a period of years for probably long-lived marbled murrelets (especially survivorship, breeding success, and immigration/emigration rates, all of which have been shown to most affect population change), we can only expect demographic models, at best, to develop approximate visions of future population projections and probability of extinction. Even such approximate visions provide us with valuable information for research and management purposes. While sensitivity analyses can be used to evaluate effects of differing values for parameters that are not well known, interpretation of such analyses can suffer from the degree of uncertainty of certain scenarios examined or not examined. Due to greater uncertainty about parameter values and other factors over time, population trends and probabilities of extinction far in the future must be regarded as less reliable than those in the near future. However, unless strong data or logic can be provided for why future projections are not plausible, it is desirable to examine less reliable future projections because they allow us to better examine the potential importance of available information on future population status. A reasonable but imperfect view of future population projections is better than no view at all.

Few demographic models of marbled murrelet population dynamics have been published in the peer-reviewed literature (Cam et al. 2003). Most modeling efforts have utilized deterministic Leslie Matrix models (Leslie 1945, Caswell 1989), with the only special feature being that adult year or age classes are combined into a single adult life stage. As such, these models are termed stage-, rather than age-structured models (Caswell 1989). These relatively simple models have only 3 stage classes (in the case of murrelets: age 0-1 or juveniles, age 1-2 or subadults, and ages 2+ or adults) and have relied on emerging information on annual breeding success, adult survival, and sex ratio. Other demographic variables (age of first breeding and

proportion of adults breeding) have been estimated from available information in other alcid species. This general description characterizes most models to date (Beissinger 1995a,b, Beissinger and Nur 1997, Boulanger et al. 1999, Boulanger 2000, Beissinger 2002, Cam et al. 2003). These models (as almost all population models) represent only the female population, because females carry the future reproductive potential for the population. These are easily converted to total population if a constant sex ratio (e.g., 1:1) is used. Such simple stage-based models are appropriate when little information is available about the species, as is the case for marbled murrelets, and populations being modeled because they require fewer assumptions. However, they also run the risk of being over-simplified and not reflective of complicated and cumulative impacts.

Leslie Matrix models have been used in a number of instances to estimate marbled murrelet trends in Oregon, Washington, and California. Among these models, only the Beissinger models were used to forecast the murrelet population into the future, while the others looked at steady state conditions (i.e., abundance trends after the population age structure has stabilized to a stable age distribution). Using breeding success determined from juvenile:adult ratios, Beissinger (1995a) and Beissinger and Nur (1997) forecasted a general population decline of 4 to 7% per annum for California, Oregon, and Washington populations. In addition, Beissinger (1995b) evaluated the effects of different levels of gill-net bycatch mortality on future population size in northern Washington (Zone 1), and Beissinger (2002) evaluated the effect of a 10% reduction in population size on future population projections in northern California (Zone 4). Boulanger et al. (1999) and Boulanger (2000) evaluated high productivity and high survival life history scenarios and found that their model was most sensitive to adult survival (less so to juvenile survival and breeding success) and that long time periods are required to statistically demonstrate population changes due to variability in parameters. In most cases, direct evidence of such changes also must be found through at-sea surveys, which also exhibit much variability and require long time periods to statistically demonstrate changes. Cam et al. (2003) evaluated the effect on future population projections from different estimates of adult survival from mark-recapture studies. All current models for California, Oregon, and Washington populations suggest a population decline of 4 to 7% per annum (e.g., Beissinger and Nur 1997). Cam et al. (2003), however, suggested that a stable population (or slightly declining) may be possible in the Straits of Georgia in southern British Columbia.

These stage-based models have several limitations based on the assumptions used for several parameters:

- Survival Rates – Survival rates for adults were assumed to not change over time although this may not be the case, especially when large mortalities from oil pollution or gill-net fishing affect populations in some Conservation Zones in certain years. Cam et al. (2003) tested for change in murrelet survival rate over time and found no significant effect of time on survival with available mark-recapture data from southern British Columbia, although their ability to "detect such an effect was low" and little oil mortality likely occurs in the studied population.

- Recruitment – In stage-based models, recruitment is assumed to be "knife-edged," where recruitment occurs all at a single age (i.e., age 2-3 in most cases for the murrelet although Boulanger et al. [1999] also examined the effect of knife-edge recruitment at age 4).

- Life Span – An infinite lifespan is assumed (implicit in using a single adult age class) but a slight bias results from a small proportion of the population existing in the model at ages older than the maximum expected lifespan of the marbled murrelet. This bias is greater when the population rate of growth is not close to 1 (i.e., not a stable population). For example, simulation experiments using a model with 22 adult age classes (using a maximum age of 25 – see below) had less than a 1% bias for a decline of 3 to 5% per annum but had a 5% bias for a decline of 12% (i.e., underestimated the decline rate by 5%, when the annual decline was 12% per annum). For most reported annual rates of decline for the marbled murrelet, this bias is not serious.

Results of sensitivity analysis models (Beissinger and Nur 1997, Boulanger et al. 1999) suggested that the rate of population change is most sensitive to adult survival, fecundity, and juvenile survival. However, these sensitivity analyses did not consider the degree of uncertainty or variability in parameter values (e.g., annual fecundity is more variable than adult survival). Greater variability and uncertainty in fecundity increases its importance to long-term population trends.

Two other more complex, unpublished, models of marbled murrelet population dynamics (Akcakaya 1997, Swartzman et al. 1997) evaluated the effect of forest cutting schedules in southern Humboldt County, California, on marbled murrelet populations. These models were developed to better simulate what happens to a murrelet population when nesting habitat is removed than is possible with stage-based models, but some assumptions were required that have not been verified with specific murrelet data. Akcakaya (1997) modeled the effect of forest cutting using a metapopulation model for 3 different portions of Zone 4 (Southern Oregon, Del Norte-

Northern Humboldt Counties, and Southern Humboldt County). Each portion was modeled using a stage-based Leslie Matrix model with parameters based on Beissinger (1995a), except that fecundity values were chosen to make the population stable (i.e., dominant eigenvalue of the survival-fecundity matrix equal to 1.0). However, this model was stochastic in that for each year a different survival value was generated by sampling from a uniform distribution over the range of values given in Beissinger (1995a). A range of annual immigration values between 0 and 5% for juveniles and subadults was used between each portion. Another feature of the model was a carrying capacity for density-dependent survival (a Beverton-Holt model), with carrying capacity set between 75 and 125% of initial population sizes. The effect of logging was examined by reducing the carrying capacity in relation to the amount of habitat removed. A small additional stochastic mortality was added to account for catastrophes (e.g., oil spills). Parameters were generated from a correlated distribution. In other words, rather than independently choosing a parameter value from the range of values each year, the value for 1 year depended on the value for the previous year (i.e., year-to-year parameter values are correlated).

Swartzman et al. (1997) developed the LIMBS model (landscape-level individual murrelet based simulation) to evaluate the effect of forest cutting on marbled murrelet populations in Zone 4. Individual birds were modeled, each breeding individual was associated with a stand area through GIS, and their demographics were determined stochastically. Additional model assumptions beyond those typically found in other models included: (1) calculating probabilities of murrelets returning to their stand of birth (assuming a high degree of philopatry, as in other alcids) or nesting elsewhere (immigration/emigration within Zone 4); and (2) modeling probability of breeding success as a function of stand condition (degree of crown closure and amount of edge versus core area of the stand [computed using GIS], factors which are considered to strongly influence corvid predation [Nelson and Hamer 1995]). Ranges of the parameter values were based on available data on marbled murrelets, as informed by knowledge from other alcids. Individual bird survival and fecundity were affected by comparing the value of Monte Carlo generated random numbers between 0 and 1 with, for example, the probability of survival and generating mortality (i.e., killing the bird) if the number is greater than the survival probability.

Several limitations also exist within these multi-aged models. In particular, various assumptions were made about the distribution of murrelet breeding sites in old-growth forests; breeding success in relation to stand condition; movements of birds due to loss of nesting habitat; immigration/emigration; and mortality due to oil pollution. These assumptions, while reasonable,

have not been verified with actual studies on marbled murrelets, although some are justified through available data on other alcids (Swartzman et al. 1997).

3.5.2 Development of the Marbled Murrelet Conservation Zone Model

One of the main purposes of this 5-Year Status Review of the marbled murrelet is to assess the status and trends of marbled murrelet populations within each of the 6 Conservation Zones (Service 1997). These zones had been originally considered by the U.S. marbled murrelet recovery team to be semi-independent demographic and geographic sub-populations where birds that bred in the zone likely also fed within the zone during the breeding season and to a large degree during the non-breeding season. In addition, zones faced differing threats which should be managed separately, and were functional equivalents of recovery units. To promote viable populations of listed species and lower the risk of extinction, it was desirable: (1) to maintain multiple populations such that catastrophic events could not result in loss of a species throughout its entire listed range; (2) to increase population size within each zone; and (3) prevent large gaps in distribution, which potentially would lower the ability of isolated populations to sustain themselves over time. Boundaries between zones were selected to reflect major marine and terrestrial geographic landmarks that served as approximate divisions between these sub-populations, with low at-sea densities of murrelets or little old-growth forest nesting habitat occurring near boundaries. In some cases, a political boundary occurred in the same area and was selected as the boundary to facilitate management.

In recent years, several population models for the marbled murrelet have been developed and tested, potential flaws have been investigated, and better data for parameter inputs have been gathered. However, a model had not yet been developed and tested for comparable application in all 6 zones. Without use of a model, our assessment of trends would be limited to past trends from available forest and at-sea census data, which were not even available in most zones, and it would be very difficult to compare trends in different zones. A new model was needed to integrate available demographic information for comparable depiction of current expectations of future population trends and probability of extinction in each zone. In addition, much new information on demographic parameters (mainly survival and breeding success) had recently come to light, bringing up major questions about the validity of parameter values used in past models. Finally, no available models have been used to project extinction probabilities into the future.

Sufficient prior modeling work, sensitivity analyses, and demographic information was available to justify preparation of Leslie Matrix models for each Conservation Zone. Using many of the strengths and avoiding some potential weaknesses of past modeling efforts identified in this review, we developed a new stochastic model (termed the "Zone Model") to assess future projected trends in each zone. This model:

- Compares the difference in trends using higher estimates of breeding success from telemetry studies versus lower estimates from juvenile:adult ratios;
- Examines a range of adult survival estimates between low values found in 2 studies and high values found in 1 study and in many other alcid species;
- Examines scenarios with no versus low levels of immigration/emigration movements;
- Uses recent estimated zone population sizes in 2001 from the EM Program for the NWFP;
- Uses improvements in model structure to reduce certain biases;
- Accounts for estimated oil and gill-net mortalities and compares model projections with and without these sources of mortality; and
- Allows for calculation of extinction probabilities.

With the Zone Model, we integrate available demographic information and mortality estimates to produce reasonable future population projections for each zone, as well as for the entire U.S. listed range in California, Oregon, and Washington. This effort is desirable at this time for the purpose of evaluating knowledge of the current status and trends in each zone within the 21st century. We have focused on expected trends in population size over the first 4 decades but examined the probability of extinction over the whole century. In the first 4 decades, many decisions about protecting the species will be required, nesting habitat should stabilize at a certain level (assuming that remaining nesting habitats are not saturated by existing murrelet populations) with little or no benefit from regeneration of second-growth forests, and new data will continue to be gathered on parameters and modeling approaches. We expect that, by the end of 40 years, many population conditions and parameter values will have changed, making longer-term projection of population trends much less reliable. Despite at least some expected loss of nesting habitat in old-growth forests over the

next 40 years, we have assumed in this model that there will be no change in population vital rates over 40 years. Similarly, variation in the California Current is expected over the next century, which may result in extended periods (e.g., decades) of lower or higher prey resources. At present, it is not clear to what degree, if any, oceanic regime shifts may affect prey availability or reproductive rates for marbled murrelets (Service 1997). Since it is unknown if breeding success and survivorship for the marbled murrelet will be better or worse than the ranges of parameter values used in this model over the next 40 or 100 years, we examined extinction probabilities out to 100 years. This, however, may be an optimistic scenario if extensive loss of old-growth forest habitat occurs due to logging or natural events, or if reproductive rates are depressed for decades by marine conditions within the time periods modeled. Although the approach we used for examining extinction probabilities was rough, we could not devise a more reasonable approach with available information. Over the next 20-40 years, new modeling efforts undoubtedly will occur and help refine the Zone Model, using additional information gathered in the future. In addition, annual survey data may show trends after 1 to 2 decades of effort (given high variability) for more direct verification. However, our current knowledge and handling of vital rates may in retrospect be considered sufficient and ranges of parameter values used may not change appreciably in the future. Only future research will verify if parameter ranges are reasonable. We did not incorporate habitat changes into the Zone model (e.g., Swartzman et al. 1997) because of unavailability of GIS data on old-growth forests, incomplete information on forest areas used by murrelets, and time constraints for this review. Similarly, we did not incorporate possible effects of oceanic regime shifts into the model due to insufficient information about possible effects and current inability to predict when in the next century such shifts may occur.

The Zone Model is a female-only, multi-aged, discrete-time stochastic Leslie Matrix model (Swartzman and Kaluzny 1987, Caswell 1989). A multi-aged model was chosen because the marbled murrelet, like other alcids, probably is long-lived, with high adult survival, low annual fecundity, and delayed maturity (Hudson 1985, Gaston and Jones 1998, Burger 2002). We also made several modeling improvements as follows:

- Using multiple year classes instead of 3 stages;
- Using a maximum lifespan of 25 years instead of an infinite lifespan;
- Using ranges for many population parameters to reflect year-to-year variability and uncertainty in demographic parameter values;

- Phasing in the age of first recruitment between the ages of 2-5 years, which occurs in other alcids, rather than all birds recruiting at age 3; and

- Accounting for the possibility of different low levels of immigration and emigration between zones through comparing scenarios of 2%, 5%, and 0.1%.

As in other models, discrete time is used because reproduction in murrelets is highly seasonal, and the model can be updated once a year near the time of reproduction. In the Zone Model, update time is assumed to be just after fledging.

3.5.2.1 Model Equations

The following narrative describes the algorithms used for survival, breeding success, and immigration/emigration.

Survival

Let $N_i(t)$ be the number of female individuals in age class i at time t. Age class i goes from 1 to 25. Population projection from year to year from one age class to the next uses a binomial sample with a sample size equal to the population size in the previous age class in the previous year and a survival probability $s_i(t)$:

$$N_i(t+1) = \text{Binomial}(N_{i-1}(t) * s_i(t))$$

where $s_i(t)$ is the expected survival probability from age i -1 to age i. The survival probability is based on age (it differs for juveniles, subadults, and adults) and the range of values selected for adult survival. Each year, a random adult survival probability is picked from a uniform probability distribution over the possible range of expected survival probability. This is equivalent to an uninformative prior distribution in Bayesian statistics in that we do not have information that any value over the range is more likely than another (Wade 2000). Annual variability derived through this process is assumed to include environmental stochasticity, although this has not been demonstrated.

Survival probability for the oldest (25^{th}) age class is assumed to be zero. Maximum lifespan of 25 years was a rounded-off value of the oldest known alcid (common murre, 26 years, 5 months; Clapp et al. 1982). While some larger alcids may live longer than 25 years, we doubt if smaller alcids live longer than 25 years, and may in fact have slightly shorter maximum life spans. Using this value instead of an infinite maximum lifespan has little

effect on projections because few marbled murrelets survive beyond 25 years in model outputs with infinite lifespan (less than 1%, Figure 3.5-1). A larger percentage of the population exists at ages 15-25 (Figure 3.5-1), and if a much shorter maximum lifespan was found to occur (e.g., 15 years), population projections would be changed significantly. However, the oldest-known banded Cassin's auklets and Xantus's murrelets (i.e., other small alcids within the California Current) are over 15-20 years, and low effort has been expended to recapture older banded individuals (Ainley and Boekelheide 1990, Carter et al. 1992). Thus, 25 years appeared to be a reasonable value for maximum lifespan for marbled murrelets.

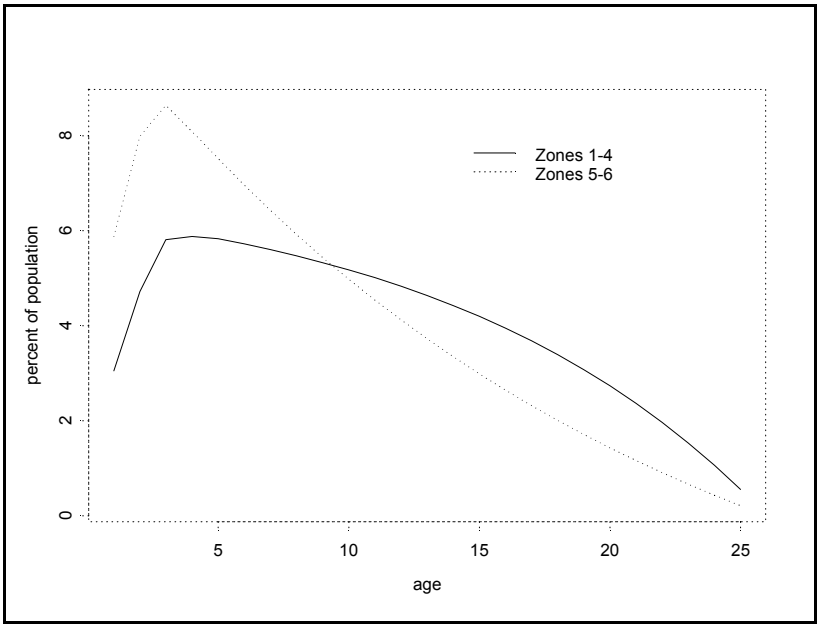


Figure 3.5-1. Initial age distributions used in the Zone Model for Zones 1-4 and Zones 5-6.

Fecundity

We calculated fecundity in 2 steps:

First, the number of breeding female murrelets NB(t+1) is calculated:

$$NB\ (t+1) = \sum N_i(t+1)\ * m_i * breed$$

where m_i is the fraction of females of age class i that are mature, and "breed" is the fraction of mature females that breed.

Second, the number of female fledglings per breeding female is calculated by sampling from a binomial distribution with sample size NB(t+1) and probability of producing a fledging for a breeding female fec(t+1).

$N_1(t+1) = \text{Binomial}(NB(t+1), fec(t+1))$

The parameter fec(t+1), the expected probability of a breeding female successfully fledging, is changed from year to year through sampling from a uniform distribution within the range of mean breeding success values from telemetry studies for each zone.

We compared this approach with mean juvenile:adult ratios, as another estimate of breeding success. Since juvenile:adult ratios are based on total (not just breeding) females we use:

$N_1(t+1) = \text{Binomial}(\sum N_i(t+1) * m_i, ratio(t+1))$

Where ratio(t+1) is the corrected juvenile :adult ratio (Beissinger and Peery 2002).

Immigration/Emigration

The Zone Model includes an equation that allows for immigration and emigration (hereafter collectively referred to as "immigration") of individuals from each zone to adjacent zones. An immigration rate $(mig)_k$ is used that depends on zone k, but not on age or year. In general, for every zone, the population is updated based on immigration rates out and in from adjacent zones:

$$N_{ik}(t+1) = (1 - mig_k)*N_{ik}(t+1) + (mig_{k+1}/2)* N_{ik+1}(t+1) + (mig_{k-1}/2)* N_{ik-1}(t+1)$$

Here $N_i(t+1)$ denotes the i^{th} age class in zone k at time t+1. Because immigration movements of marbled murrelets can be expected to be much higher in juveniles and subadults than in adults (which are expected to have high fidelity to a nesting area), we assumed that immigration occurs solely in age classes 1-3 (i.e., i =1-3 in the equation above). However, given high philopatry in alcids, low immigration rates are expected overall for marbled murrelets. Low immigration between populations of marbled murrelets is also indicated by the available, but limited, genetic information (see Section 3.4).

3.5.2.2 Model Parameters and Initial Conditions

This section provides a description of the parameters applied in the Zone Model, as well as the associated assumptions used in calculating model initial conditions and change over time.

Initial Population Size Conditions

For initial population size, we used available population estimates from 2001 for Zones 1-5 from the EM Program for the NWFP (Huff et al. 2003) and for Zone 6 from the University of California Berkeley (Peery et al. in prep.) as follows: Zone 1 (8,900); Zone 2 (1,700); Zone 3 (7,500); Zone 4 (3,900); Zone 5 (100); Zone 6 (637); and all 6 zones (22,737) (also see Table 3.2-1). We chose to use 2001 values to begin projections because 2001 values were from the middle year of 3 years of available data (and annual means were not significantly different). It also simplified accounting by being the first year of the century for the 40-year and 100-year periods being modeled.

Population estimates were derived from at-sea line-transect surveys conducted between mid May and late July and include marbled murrelets of all ages and breeding status (i.e., breeding adults, non-breeding adults, subadults, and some juveniles). Overall, we consider these estimates to be the best available information and to reflect the approximate total population size for each zone, despite various biases that might lead to slight undercounting (e.g., incubating birds at breeding sites, fledging after July, early dispersal outside of the zone).

Initial Age Distributions

To begin model simulations, we produced age distributions in order to have initial estimates for each age class. Following the method of Beissinger and Nur (1997) and Swartzman et al. (1997), our initial age distribution was the stable age distribution from a deterministic Leslie Matrix model with best estimate parameters for each zone (see below). This represents our best estimate of an unknown quantity. To produce age distributions, breeding success and survival were assumed to be constant at the midpoint of ranges (but this was not how these parameters were determined for future projections). The initial population distribution is the first eigenvector of the survival-fecundity matrix (i.e., matrix with all age i-1 to i survival fractions along the sub-diagonal and fecundities in the first row of the matrix). Initial age distributions for each zone are shown below (Figure 3.5-1). Leslie Matrix model behavior has been shown to not be sensitive to the choice of initial age-distribution.

Sex Ratio

We used a 1:1 sex ratio throughout all year classes (Burger 2002). The same assumption was made in other models (Beissinger and Nur 1997, Swartzman et al. 1997, Akcakaya 1997, Cam et al. 2003).

Age of First Breeding

We phased in breeding between 2-5 years of age to better reflect the manner in which alcids typically recruit into populations and assuming that the age of first breeding in marbled murrelets is neither much earlier nor much later than most other alcids (Swartzman et al. 1997, Gaston and Jones 1998, Burger 2002, Cam et al. 2003). This resulted in a later age of first breeding than assumed in some earlier models (e.g., Beissinger and Nur 1997). The proportion of birds aged 2, 3, 4, and 5 that were mature were 5%, 40%, 60%, and 80%, respectively (Burger 2002). We assume that all birds were considered to breed by age 6, even though some may not until an older age. In Figure 3.5-1, 54-68% of zone populations are over the age of 7 and, including the mature fraction of birds with ages 2-6, more than 80% of zone populations are breeding-aged adults.

Proportion of Adults Breeding

Without murrelet-specific data, Beissinger and Nur (1997) used 90% for the proportion of adults breeding in all years, based on information from other alcids (Hudson 1985). Recent efforts to determine the proportion of adults that breed have resulted in estimates of 31-95% (Burger 2002, Beissinger and Peery 2003). Lower end estimates reflect minimum estimates based on elevated vitellogenin levels or radio-telemetry tracking, whereas upper end estimates reflect the presence of brood patches on captured adults plus radio tracking. Some debate exists about whether: (1) the pattern of timing of elevation of vitellogenin levels is poorly known and low vitellogen levels might reflect birds before or after egg production; (2) the presence of a brood patch might not indicate active breeding in all adults; and (3) the lack of a brood patch might not indicate a lack of active breeding in all adults (although subadults are expected to lack a brood patch). Moderate estimates based on radio-telemetry tracking alone (e.g., 62.5%; Bradley 2002) also are biased because some birds do not visit nest sites after capture and radio tagging due either to effects of capture or if breeding failure occurred before capture and birds stopped flying inland and did not lay a replacement egg. On the other hand, upper end breeding percentages determined from telemetry are similar to percentages determined from a relatively large sample of marbled murrelets collected at Langara Island, British Columbia (15% of the sample were subadults and 100% of adults were breeding based on gonad development and presence of a brood patch; Sealy 1975b) and information from Atlantic puffins (84-95% of adults were breeding; Hudson 1985) and common murres (90-95% of adults were breeding; Harris and Wanless 1995).

We chose to use a 90% value for the percentage of adult marbled murrelets that breed in most years throughout the listed range because this value was consistent with information from other alcids, no natural mechanism has been identified for different physiology or behavior in marbled murrelets, and reliable murrelet-specific data have not yet been developed. However, sufficient evidence has been found in some other alcids in the California Current to indicate that the proportion of adults breeding or breeding success may be much lower during periods of severe food reduction (El Niño and other warm-water periods; Ainley and Boekelheide 1990). To accommodate this evidence, we used an approximate value of 50% of adults breeding during severe El Niños. Marbled murrelets apparently respond to low food availability by changing foraging areas and possibly using different prey species, but at least some birds still lay eggs and certain prey species for murrelets are little affected by severe El Niño conditions. Available information suggests that foraging and possibly breeding success in marbled murrelets are affected by severe El Niños (Burger 2002). However, insufficient information is available to determine any effects during weak El Niños or other warm water years that can affect certain other alcids or seabirds in the California Current (e.g., Ainley and Boekelheide 1990, Wilson 1991). To account for severe El Niño impacts, we reduced the percentage of breeding adults rather than reducing breeding success. Both are possible but insufficient information was available to determine the degree of reduced breeding success (or if any occurs) and there was some evidence that the proportion of adults breeding may be lower in some years than in others (Peery et al. in prep.). We suspect that low prey availability in severe El Niño years may cause deferred breeding mainly in first-time or younger breeders, which likely are more sensitive to lower prey availability. However, most older and experienced adults likely lay eggs and breed with a similar or slightly lower rate of success as in other years because they are better able to compensate for lower prey availability by switching between available prey resources and altering foraging patterns and areas.

For non-El Niño years, we used 90% adults breeding, which reflected all adults attempting to breed when feeding conditions are adequate to good; up to 10% of adults are not expected to breed due to mate or site loss (as shown in common murres in Scotland; Harris and Wanless 1995). For future projections, we assumed that there will be 3 strong El Niño years in every 25-year period (i.e., 0.12 probability of an El Niño). This assumption was based on evidence of severe El Niño effects to seabirds in the California portion of the California Current in 3 breeding seasons (i.e., 1983, 1992, and 1998) in the past 25 years (1979-2003). While rates of severe El Niños may vary between Pacific Decadal Ocillation (PDO) phases, we do not have sufficient information on responses of breeding seabirds to severe El Niños prior to

1972 (Ainley and Boekelheide 1990). We used an average rate over the 1979 to 2003 period to encompass a large period of available data, although most of this period included the latest warm phase of the PDO (i.e., between 1977-1998) when severe El Niños may have been more frequent. Due to the infrequency of severe El Niños, we truncated available data from 1972-78 such that there was similar inclusion of 4-5 years before 1983 and after 1998. However, this truncation approach made little difference to the rate used (i.e., 0.09 versus 0.12). Thus, we sampled from a uniform distribution which, if the value was less than 0.12, resulted in 50% of breeding age adults breeding in all zones. Impacts probably are less in parts of Zone 1 than in other zones, but we could not develop a different value to apply for Zone 1.

Survivorship

We used a range of annual survivorship values (83-92%) reported for adults (ages 3 and higher) from 2 mark-recapture studies in the same area in British Columbia (Burger 2002, Cam et al. 2003). The low end of this range was almost identical to a mark-recapture study in central California (0.82; Peery et al. in prep.). There is uncertainty about where in this range the true value lies because of many potential biases in studies conducted to date and because some variation likely occurs between years and zones. This range covered the majority of adult survival values found in genera *Uria*, *Alca*, and *Fratercula* (circa 90%), included lower estimates for *Cepphus* (circa 83%), excluded lowest values found in genera *Synthliboramphus* and *Aethia* (77-79%), and excluded highest values found in *Uria* (93-99%)(Hudson 1985, Sydeman 1993, Gaston and Jones 1998). Thus, this represents the central portion of the range of means for all alcids. We chose to model the uncertainty by sampling from a uniform distribution over the range selected. This approach reduced possible bias by inappropriately selecting the low end, high end, or center of the range values when insufficient information is available to narrow the range for all zones. This also acted to include year-to-year changes in this important parameter. For juveniles, we used 70.1% of adult values and for 2-year-olds 88.8% of adult values, following the approach of Beissinger and Nur (1997). Annual survival estimates for all age classes are assumed to be similar between zones because insufficient data are available to determine if survival may differ between zones, in the absence of oil and gill-net mortalities. No better approach is currently available.

Nesting Success

To attempt to compare nesting success and at-sea fecundity, Burger (2002) suggested using nesting success and modifying it for: (1) proportion of adult females that breed each year; (2) proportion of fledglings that reach the sea

after leaving the nest; and (3) number of nesting attempts per season. We have not attempted to directly compare estimates of nesting success with at-sea fecundity because we cannot verify such adjustments. Instead, we have examined a high and a low value of nesting success for each zone. High values were determined from telemetry studies and low values from date-adjusted juvenile:adult ratios. Telemetry studies may be biased high and at-sea juvenile:adult ratios are considered to underestimate nesting success. Intermediate values were found for nests with known outcomes, although we did not use these data directly but rather to verify ranges developed from telemetry studies. Current fecundity is probably density-independent because population sizes are depressed, suitable breeding and feeding conditions exist, and murrelets are solitary nesters. It is not clear if fecundity will decrease further with additional loss of habitat. However, fecundity may increase if low-quality nesting habitats are lost or no longer used by murrelets as populations decline.

Telemetry Studies

To obtain nesting success values (i.e., number of fledglings per breeding pair per year) from telemetry studies, we first collated available information on nesting success for 3 recent radio-telemetry studies in southern British Columbia (0.46 fledglings per breeding pair; n=25 nests in 2 years; Bradley 2002), northern California (0.135-0.324 fledglings per egg laid; n=37 nests in 3 years; R. Golightly, pers. comm.), and central California (0.00 fledglings per egg laid; n=12 nests in 3 years; Peery et al. in prep.; Burkett, unpubl. data). British Columbia and northern California telemetry studies were conducted in 1999-2003 (apparently within a newly developing cold phase of the PDO) and may reflect years of high prey availability. However, in central California, no difference was found in breeding success between the severe 1998 El Niño and nearby non-El Niño years (1997, 2001, 2002), at least partly due to apparent high nest predation (Peery et al. in prep.; E. Burkett, pers. comm.). Together, these years bridged the end of the warm phase of the PDO and the new cold phase of the PDO. For this model, we treated breeding success as though there was little difference between breeding success in warm versus cold phases of the PDO and between severe El Niño and non-severe El Niño years. However, we accounted for some potential effects from severe El Niños through adjustment to the proportion of adults breeding (see above). To corroborate these levels of success, we compared these values with nest success for all nests with known outcomes by state and differences in adult:juvenile ratios between zones. We assumed that current nesting success was generally indicative of future nesting success over the next 40 years and that nesting success was higher in the past for all zones. While this assumption is overly simplistic, it is not clear if further loss of

nesting habitat will lead to lower nesting success than already depressed values found in Zones 3-6. However, continued loss of old-growth nesting habitat in Zones 1 and 2 will likely lead eventually to depressed breeding success as found in other zones, but when this may happen in the future is not known. In some zones with large blocks of unentered old-growth forest (i.e., Zones 1, 2, and 4), loss of use of poor-quality habitats over time may lead to higher nesting success of smaller numbers of birds in remaining better-quality habitats. Assuming relatively low loss of old-growth forest over the next 40 years, we expect that breeding success will remain similar over the next 40 years and perhaps even the next 100 years.

For each zone, we applied telemetry nesting success data using various considerations as follows:

- *Zone 1 (Nest Success = 0.380-0.540*): Telemetry studies, with associated estimates of nest success, have not yet been conducted in Washington, and little information from other sources is available for the state. Of 7 nests with known outcome in Washington (Zone 1), 4 (or 0.57) were successful. Although a small sample, this suggested relatively high breeding success. Relatively high juvenile:adult ratios also have been reported in Zone 1, but date-corrected ratios were only slightly higher than in Zones 3 and 4 (Stein and Nysewander 1999; Beissinger and Peery 2003). Telemetry nesting success from the eastern Straits of Georgia in southern British Columbia seemed to be the most suitable surrogate for Zone 1 because of: (1) similar habitat conditions (i.e., similar prey and feeding conditions in inner protected waters); (2) a higher level of suitable nesting habitat in Zone 1 than in other zones; and (3) substantial harvesting of old-growth forests and occurrence of human communities near nesting areas in both Zone 1 and southern British Columbia (Service 1997, Burger 2002). We have no clear evidence that nesting success differs extensively between Zone 1 and southern British Columbia, although the amount of available habitat per unit area may be greater in southern British Columbia. Much greater difference exists between Zone 1 and Zone 4 habitat conditions (i.e., the next nearest locality with telemetry data on nest success) than between Zone 1 and British Columbia habitat conditions. Nesting success reported for British Columbia (0.46) may be biased high due to measuring chick survival to mid-chick period, and excluding some birds that failed prior to capture. However, we chose not to adjust this rate but instead create an arbitrary range of values around this value (i.e., 0.160), similar in size to the largest range used in other zones (i.e., Zone 6 – 0.160; Zone 4 – 0.094); Zone 3 – 0.136). For this review, we also felt that it was important to ensure that future projections for this zone (and Zone 2; see below) accounted for

the possibility of higher breeding success in this zone, instead of merely assuming lower success without sufficient information to prove lower success. Later in this section, we also examined the sensitivity of instead using a lower estimate of breeding success from age-corrected adult:juvenile ratios for Zone 1.

- *Zone 2 (Nest Success = 0.380-0.540)*: No data on nesting success nor adult:juvenile ratios are available in this zone. We applied the same level of nesting success as in Zone 1 (0.380-0.540) because the majority of birds occur near the north end of Zone 2 (which may reflect similar prey to inner or more northern waters) and use moderate-quality forest habitats on the west slopes of the Olympic Mountains, conditions more similar to Zone 1 than to Zone 3.

- *Zone 3 (Nest Success = 0.324-0.460)*: No telemetry data on nesting success are available for Zone 3, but nests with known outcomes in Oregon (mostly Zone 3) have had relatively high values (0.429, n=21 nests) compared with Zone 4 telemetry data (0.230-0.324; see below) or Straits of Georgia telemetry data (0.46). Adult:juvenile ratios are relatively high and similar to Zones 1 and 4 (Beissinger and Peery 2003). Prey and feeding conditions should be most similar to Zone 4, but forests are highly fragmented in Zone 3 compared to the Redwood National and State Park study area in Zone 4, Olympic Mountains in Zones 1 and 2, or the northern Cascade Range in Zone 1. There is some possibility that Oregon nests may not be representative of breeding success in Zone 3 (e.g., exclusion of 2 successful nests from southern Oregon in Zone 4 would reduce Oregon nest success from 0.429 to 0.368; K. Nelson, pers. comm.). We considered that a range from the top of the Zone 4 range (0.324) to the Straits of Georgia value (0.46) was most appropriate for this zone, particularly since the success of nests with known outcomes (0.368 and 0.430) fell within this range.

- *Zone 4 (Nest Success = 0.230-0.324)*: Nesting success from telemetry data for Redwood National and State Parks was considered to be generally representative of Zone 4 since most birds in this zone nest in the area studied or similar nearby habitats. However, the range of nesting success values calculated by Hebert and Golightly (R. Golightly, pers. comm.) reflected a minimum value based on 28 eggs with known outcomes and a maximum value based on assuming that all 7 eggs without known outcome did in fact fledge. We did not consider values in the lower end of the reported range to be representative for estimating nesting success per breeding pair because: (1) for 2 of 28 eggs studied, nest failure was reported twice (i.e., for the first and replacement egg instead of once per

breeding pair); and (2) 7 out of 17 hatched nests studied had unknown outcomes but at least half should have hatched (i.e., some nests had older chicks near fledging at the time of radio failure, and 5 of 10 hatched nests with known outcome fledged [P. Hebert, pers. comm.]). To roughly account for these issues, we chose to use the upper half of the reported range (0.23-0.324) for Zone 4, although this study also excluded some birds with brood patches that may have failed before capture which might bias nesting success upwards. We could not corroborate these values with other data. Few nests have been studied without telemetry in northern California (Zone 4) but 1 with known outcome failed due to chick death (Nelson and Hamer 1995). Nesting success for California presented in Section 4.5 included telemetry data for northern California and telemetry and non-telemetry data from central California (K. Nelson, pers. comm.). Adult:juvenile ratios are relatively high for Zone 4 and similar to Zones 1 and 3 (Beissinger and Peery 2003).

- *Zone 5 (Nest Success = 0.00-0.16):* No data on nesting success nor adult:juvenile ratios are available for this zone. We applied Zone 6 values for nesting success (0.00-0.16) because: (1) forest habitats are very limited and likely are of poor quality in this zone; and (2) poor reproductive success in Zone 6 may reflect the southern part of the breeding range.

- *Zone 6 (Nest Success = 0.00-0.16):* Nesting success from telemetry data for Redwood State Parks was considered to be generally representative of Zone 6 since most birds in this zone nest in the area studied. However, all nests included in the telemetry study failed (0.00), whereas some other nests with known outcomes were successful and small numbers of hatching-year birds have been observed at sea (Singer et al. 1995, Beissinger and Peery 2003). When telemetry data were combined with non-telemetry data, a higher nesting success was evident (0.16; Peery et al. in prep.). Similarly, when telemetry and non-telemetry data in central and northern California were combined, a similar value (0.178) of nest success emerged. Juvenile to adult ratios are relatively low in Zone 6 (Beissinger and Peery 2003). Thus, we considered that a range of 0.00-0.16 seemed most representative for the Zone 6 population.

Juvenile-to-Adult Ratios

To obtain minimum nesting success values (i.e., number of fledglings per breeding pair per year) from date-corrected juvenile:adult ratio studies, we used collated information provided in Beissinger and Peery (2003), as follows: Zone 1 – 0.089 (.012); Zone 3 - 0.080 (.013); Zone 4 - 0.084 (.016); and Zone 6 - 0.038 (.007). No data were available for Zones 2 and 5. We

used Zone 1 values for Zone 2 and Zone 6 values for Zone 5, using similar logic as provided for telemetry studies. We consider that direct use of date-corrected juvenile to adult ratios for nesting success is incorrect because ratios do not account for: (1) fledgling deaths during the flight from the nest to the ocean; (2) juvenile or adult deaths for months after first arrival of juveniles on the ocean; and (3) different distribution and dispersal patterns between juveniles and adults (Carter and Stein 1995, Burger 2002). However, due to little information to evaluate biases or further adjustments of ratios, we have used these ratios as, possibly biased, lower-end estimates of nesting success for comparison of model outcomes with, also possibly biased, higher-end estimates of nesting success values derived from telemetry studies.

Immigration

We included low levels of annual immigration between adjacent zones into the Zone Model because low levels of immigration have been reported between island colonies in some other alcids, usually within a few hundred km of each other (Harris 1983, Hudson 1985, Harris and Wanless 1991, Halley and Harris 1993, Divoky and Horton 1995, Swartzman et al. 1997). To account for low immigration between zones, we used 2% annual immigration from each zone to an adjacent zone but confined immigration to the first 3 year classes (juveniles and subadults), which appear more likely to disperse widely during the winter prior to establishing a nesting site in a natal or non-natal area. The value of 2% was chosen to reflect low immigration, allow for significant net immigration from adjacent high to lower abundance areas, and was generally consistent with genetic information on immigration between Aleutian Islands and southern Alaska/British Columbia populations. Half of the immigration was assumed to occur from each of the adjacent zones for Zones 1-4. In Zone 1, 1% was assumed to immigrate into southern British Columbia and the same amount (i.e., 1% of Zone 1 population size) was assumed to immigrate from southern British Columbia. In Zone 6, 2% immigration was assumed between Zone 6 and 5. In Zones 5 and 6, a lower immigration rate could be expected based on the small population size and low availability of forest nesting habitat. To explore the effect of different possible levels of low immigration, we compared 2%, 5%, and 0.1% (i.e., essentially none) levels of immigration.

Parameter Independence and Variability

We assumed that the parameter values are independent of each other. Thus, when we sampled values for survival probability and fledging success in any year, we did so independently. It is possible that some parameters, such as

breeding probability and breeding success, may be correlated. This could be included in the model, as done by Akcakaya et al. (1997) by choosing correlated parameter values. This would likely increase the year-to-year variability in model forecasts. However, we felt that there is insufficient evidence to justify such correlated parameter values for survival and fecundity.

Another point of clarification is the use of mean probability values to represent model parameter ranges and the use of binomial distributions to generate the actual number surviving or fledging or breeding from populations with these mean parameter probabilities. Because the parameters are considered to be mean or expected values and not actual fractions surviving or breeding, it is appropriate to generate the actual outcomes from sampling from a population with these expected probabilities. An analogy is the difference between the standard deviation, a measure of variability around the mean of a population (the outcome of a single experiment), and the standard error, a measure of variability in the means (outcomes of many experiments).

Oil Spill and Gill-Net Losses

In the Zone Model, we included estimated mortality of murrelets from oil spills and gill-net fishing bycatch (Tables 3.5-1 and 3.5-2; see Section 5.4). In stage-based models, such mortality was considered to be accounted for by survival estimates. However, with uncertainty about what true survival rates are and variation in the degree of anthropogenic mortality in different zones, we did not assume that such mortality was accounted for by survival estimates and applied oil spill and gill-net losses to affected sub-populations. We assumed that similar mark-recapture survival estimates in Zone 6 (0.82 for dip-net captures; Peery et al. in prep.) and the Straits of Georgia (0.83 for combined mist-net and dip-net captures; Cam et al. 2003), which gave similar survival estimates, did not include losses due to gill-net and oil spill mortality. Mark-recapture survival estimates for southern British Columbia would be much lower if only dip-net captures were used (as indicated by the higher value for mist net captures, 0.92), leading us to suspect that significant biases may exist in mark-recapture estimates from dip-net captures alone, possibly due to capture avoidance. These biases may have been reduced in Zone 6 dip-net capture studies, leading to higher but possible still depressed estimates. We suspect that model sampling from a range of adult survival values (0.83-0.92) does not account for additive mortality from oil spills and gill-nets. To model these sources of mortality, we assumed that future gill-net and oil spill losses would reflect recent lower levels of losses recorded between 1992 and 2003 (see Section 5.4.3). For any model year, we

Table 3.5-1. Estimates of oil mortality of marbled murrelets by year and zone.

Year	Zone 1	Zone 2	Zone 3	Zone 4	Zone 5	Zone 6
1977	1-2	1	1	1	0	1-3
1978	1-2	1	1	1	0	1-3
1979	1-2	1	11-201	1	0	1-3
1980	1-2	1	11-201	1	0	1-3
1981	1-2	1	1	1	0	1-3
1982	1-2	1	1	1	0	1-3
1983	1-2	1	21-81	1	0	1-3
1984	7-18	11-22	1	1	0	11-63
1985	7-18	1	1	1	0	1-3
1986	1-2	1	1	1	0	51-103
1987	1-2	1	1	1	0	1-3
1988	1-2	21-261	1	1	0	1-3
1989	1-2	1	1	1	0	11-53
1990	1-2	1	1	1	0	11-53
1991	7-18	176-351	26-51	1	0	1-3
1992	1-2	1	1	1	0	11-53
1993	1-2	1	1	1	0	1-3
1994	1-2	1	1	1	0	1-3
1995	1-2	1	1	1	0	1-3
1996	1-2	1	1	1	0	7-15
1997	1-2	1	1	152	0	123-126
1998	1-2	1	1	1	0	7-15
1999	1-2	1	262	136	0	1-3
2000	1-2	1	1	1	0	1-3
2001	1-2	1	1	1	0	7-15
2002	1-2	1	1	1	0	7-15
2003	1-2	1	1	1	0	1-3

randomly sampled from the years 1992-2003, assumed that zone loss in the model year was similar to zone loss in a sampled year, and generated a number lost for each year and zone k through oiling (oiled[year,k]) and gill-nets (gilled[year,k]). However, because populations in most zones are considered to be declining, we assumed losses from equivalent-sized oil spills or gill-net fishing effort in the future to not be as large as those that occurred in 1992-2003. As such, we normalized the loss for each zone by multiplying it by the ratio of the current year's population in each zone to the initial population (2001) in each zone.

To translate this loss into losses at age, we sampled from age classes (1 to 25) with a random sample of size oiled (year,k)[gilled(year,k)] from a probability distribution of the age distribution in that year (i.e., the number in each year class divided by the sum of all year class numbers). Thus, the age distribution was perceived as a probability distribution for the probability that lost birds come from each age class. All losses for each age class were then subtracted from the estimated population. Oil spill and gill-net losses are assumed to be proportional to the abundance in the different age classes.

Table 3.5-2. Estimates of gill-net mortality of marbled murrelets by year and zone.

Year	Zone 1	Zone 2	Zone 3	Zone 4	Zone 5	Zone 6
1977	76-100	8-12	0	0	0	0
1978	76-100	8-12	0	0	0	0
1979	76-100	8-12	0	0	0	0-20
1980	76-100	8-12	0	0	0	150-300
1981	76-100	8-12	0	0	0	0-20
1982	76-100	8-12	0	0	0	0-20
1983	76-100	8-12	0	0	0	0-20
1984	76-100	8-12	0	0	0	0-20
1985	76-100	8-12	0	0	0	0-20
1986	76-100	8-12	0	0	0	0-20
1987	76-100	8-12	0	0	0	0-20
1988	76-100	8-12	0	0	0	0
1989	76-100	8-12	0	0	0	0
1990	76-100	8-12	0	0	0	0
1991	76-100	8-12	0	0	0	0
1992	19-25	2-6	0	0	0	0
1993	19-25	2-6	0	0	0	0
1994	19-25	2-6	0	0	0	0
1995	19-25	2-6	0	0	0	0
1996	19-25	2-6	0	0	0	0
1997	19-25	2-6	0	0	0	0
1998	19-25	2-6	0	0	0	0
1999	19-25	2-6	0	0	0	0
2000	19-25	2-6	0	0	0	0
2001	19-25	2-6	0	0	0	0
2002	19-25	2-6	0	0	0	0
2003	19-25	2-6	0	0	0	0

The assumption of oil loss being reduced in proportion to population size is conservative in that it assumes that remaining birds are not aggregated and thus at higher risk of oiling. However, fragmentation of nesting habitats in Zones 4 and 6 may result in greater aggregation of remaining birds during the breeding season and make them more susceptible to oil spill mortality.

3.5.3 Zone Population Projections

Using the Zone Model, we projected:

- Population size for each zone for 40 years (plots show the median value for each year and the 95% confidence limit) (2.5% and 97.5% quantiles from 50 Monte Carlo simulation runs of the model; Figure 3.5-2); and

- Extinction probabilities for each zone for 100 years, with extinction defined as less than 30 individuals (15 females) in the zone population (Figure 3.5-3). We used this definition of extinction to ensure that zone sub-population sizes would be so low that extinction was almost certain. In fact, extinction may be almost certain at higher zone sub-population sizes. Thus, calculated time to extinction may be earlier than projected

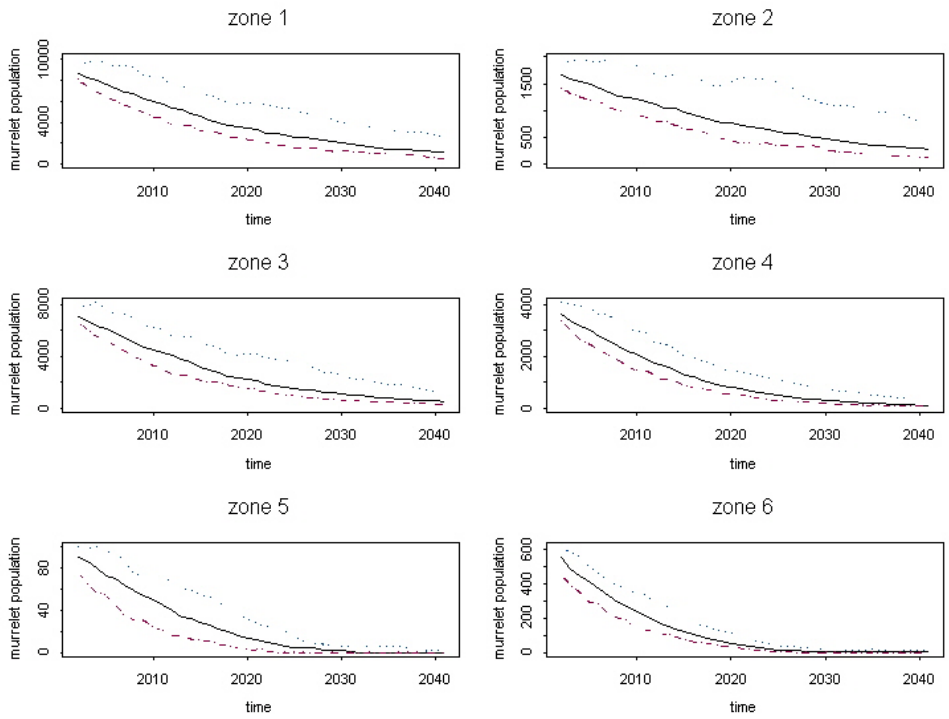


Figure 3.5-2. Population size forecasts for Zones 1-6 over 40 years (2001-2040), assuming a 2% annual immigration rate between adjacent zones, high-end fecundity, and including gill-net and oil spill mortality.

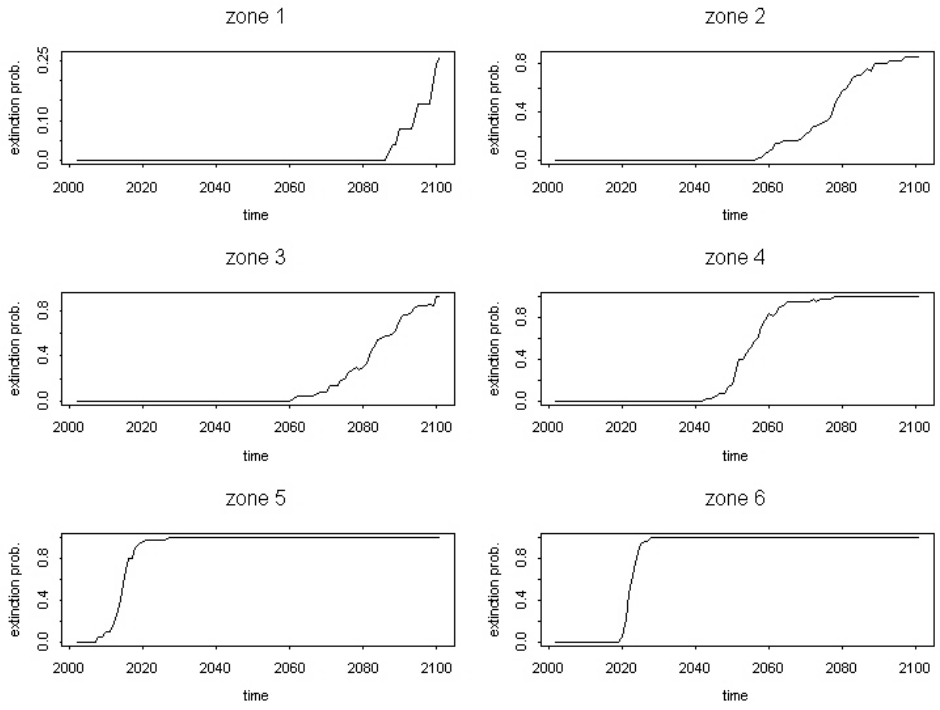


Figure 3.5-3. Forecast of probability of population extinction for Zones 1-6 over 100 years, assuming a 2% annual immigration rate, high-end fecundity, and including gill-net and oil spill mortality.

with the zone model. However, we also used zone population estimates from Huff et al. (2003), which may be at least 15% lower than actual population sizes (see Section 3.2).

- The median total population for California, Oregon, and Washington populations (Zones 1-6) by adding the median, 2.5%, and 97.5% quantile populations for each zone over 100 years (Figure 3.5-4).

- To determine annual rates of population change, we averaged annual change over each decade of the simulation, specifically 2001-2010, 2011-2020, 2021-2030, and 2031-2040 (Table 3.5-3).

The percentage declines are greatest in the first decade and are much smaller in the final decade because by this time zone populations are quite small and in some of the zones has already reached 0 in many of the Monte Carlo runs (Figure 3.5-2).

Forecast results suggest that zone sub-populations will initially decline between 3.0 and 6.2% per annum (i.e., average for first decade from 2001-2010), similar to estimated decline from earlier models (e.g., Beissinger and Nur 1997). Greatest decline is forecast in Zone 6 and least decline in Zone 2

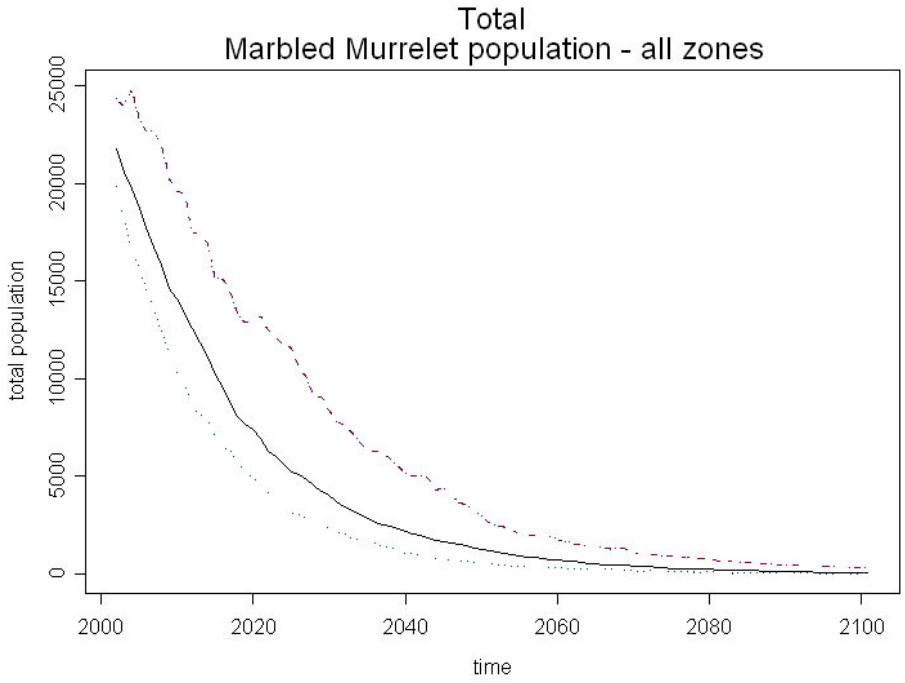


Figure 3.5-4. Population size forecast for the listed range of the marbled murrelet in California, Oregon, and Washington, assuming a 2% annual immigration rate, high-end fecundity, and including gill-net and oil spill mortality.

Table 3.5-3. Annual percentage rate of population decline in each zone, assuming 2% annual immigration rate. Numbers in parentheses denote population increases.

	Years (with oil/gill-net loss)			
Zone	**1-10**	**11-20**	**21-30**	**31-40**
1	3.4%	3.1%	2.6%	2.2%
2	3.0%	2.8%	2.4%	2.1%
3	3.9%	3.5%	2.9%	2.3%
4	4.8%	4.0%	3.1%	2.4%
5	5.0%	4.3%	3.3%	2.5%
6	6.2%	4.6%	3.3%	2.5%

(Table 3.5-3). Extinction probabilities of 100% within 40 years (i.e., by 2040) are projected in Zones 5 and 6 (Figure 3.5-4). Other sources also have previously identified these zones as having very high potential for extinction in the near future (Carter and Erickson 1992, Service 1997). Extinction is less certain in Zones 1-4 within the 21st century, but extinction probability is still high in Zone 4 (over 80% by 2060) and Zones 2 and 3 (over 80% by 2100). Only Zone 1 has a greater probability of remaining extant than becoming extinct over the 21st century (i.e., extinction probability of 25% by 2100).

After 100 years, mean population size for the listed portion of the range in Washington, Oregon, and California was projected to be 45 murrelets, after 100 years, with a probability of extinction of 16% (calculated by assuming extinction is independent in each zone and taking the product of the extinction probabilities for each zone). As noted earlier, this calculation may be optimistic because it assumes that the range of population parameters used does not change over 100 years, nesting habitats are not greatly different from today, and mortality from oil spills and gill-nets are similar to recent years.

While projected trends and extinction probabilities of zone sub-populations may be alarming to some readers, these predictions are not much different than what could be surmised from existing information and previous modeling efforts (e.g., Beissinger and Nur 1997, Service 1997). The Zone Model has merely served to collate and standardize various pieces of available information for each zone for comparison between zones. Since this is the first version of the Zone Model, we expect that additional data on parameter values and other considerations will lead to improvements in the reliability of population projections over time. However, we believe that it is doubtful that future projections will differ substantially from those presented here, unless major changes to existing information or substantial new information is uncovered. Sensitivity analyses below investigate differing responses of the Zone Model to changes in immigration rate, fecundity, and oil and gill-net mortalities. We did not investigate model sensitivity to possible changes to

other parameters because of insufficient information to justify different scenarios and shortage of time for this review.

3.5.4 Sensitivity Analyses

This section presents the results of analyses of model sensitivity to immigration rates, fecundity, and oil spill and gill-net mortality.

3.5.4.1 Sensitivity of Model Estimates to Immigration Rate

Since the magnitude of immigration between zones is highly uncertain and may lead to erroneous projections for Zones 2 and 5 (i.e., these zones are adjacent to zones with much larger sub-populations), we further compared the results from the default value of 2% migration (1% from each adjacent zone) for ages 1-3 individuals, with runs having 5% migration (Figures 3.5-5 and 3.5-6) and 0.1% (Figures 3.5-7 and 3.5-8; Table 3.5-4).

Rates of population decline and times to extinction were similar, with different levels of assumed immigration for Zones 4 and 6. Lower immigration rates resulted in higher rates of decline and earlier times to extinction in Zones 2 and 5, while higher migration rates resulted in reduced decline and a delay of extinction in these zones. The opposite effect of immigration was seen in Zones 1 and 3, although not as marked (Figure 3.5-3). This sensitivity study to changes in immigration rate suggests that the rate of population decline and probability of extinction for relatively small zone populations adjacent to larger zone populations are sensitive to immigration rates. However, trends in zones with larger sub-populations were not sensitive to differing low levels of immigration from adjacent zones.

3.5.4.2 Sensitivity to Fecundity

Given that population trends have been found to be sensitive to fecundity values (Beissinger 1995a, Beissinger and Nur 1997, Boulanger 2000) and since uncertainty exists about true fecundity from available data, we compared sub-population trends in zones using possibly high fecundity from telemetry studies with using possibly low fecundity from date-corrected adult to juvenile ratios for each zone.

Forecasts with low fecundity had the strongest effects on populations in Zones 1-3, predicting higher annual rates of decline (Table 3.5-4, Figure 3.5-9) and higher and earlier extinction probabilities (Figure 3.5-10) than with high fecundity. Zones 5 and 6 were affected in a similar manner because low estimates of fecundity from telemetry studies and low estimates of fecundity from adult to juvenile ratios both occurred in Zone 6 (with estimates

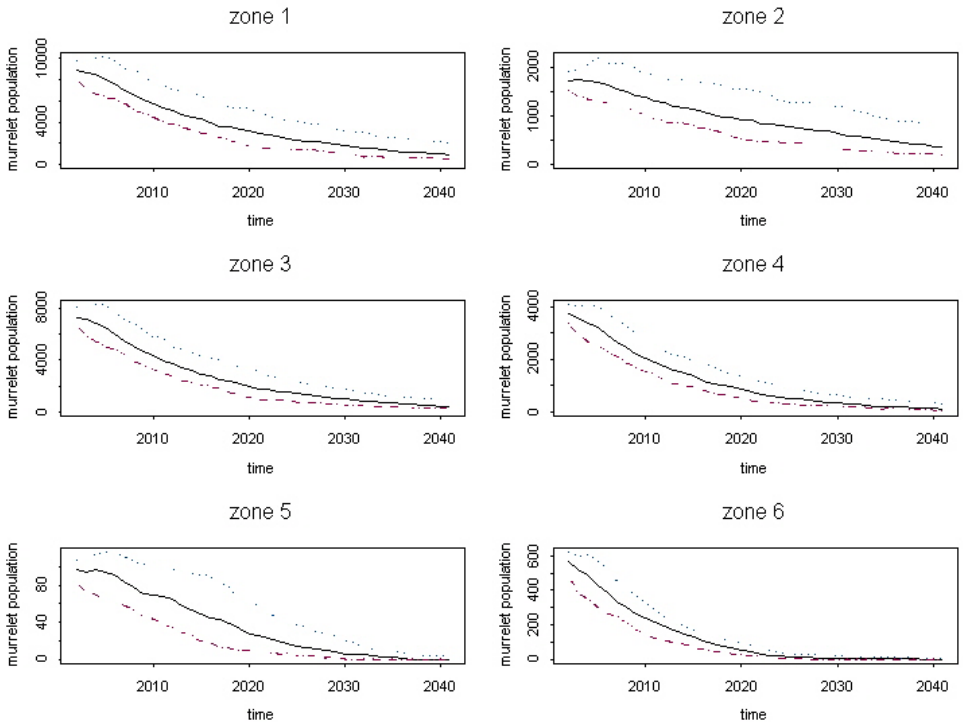


Figure 3.5-5. Population size forecasts for Zones 1-6 over 40 years, assuming a 5% annual immigration rate, high-end fecundity, and including gill-net and oil spill mortality.

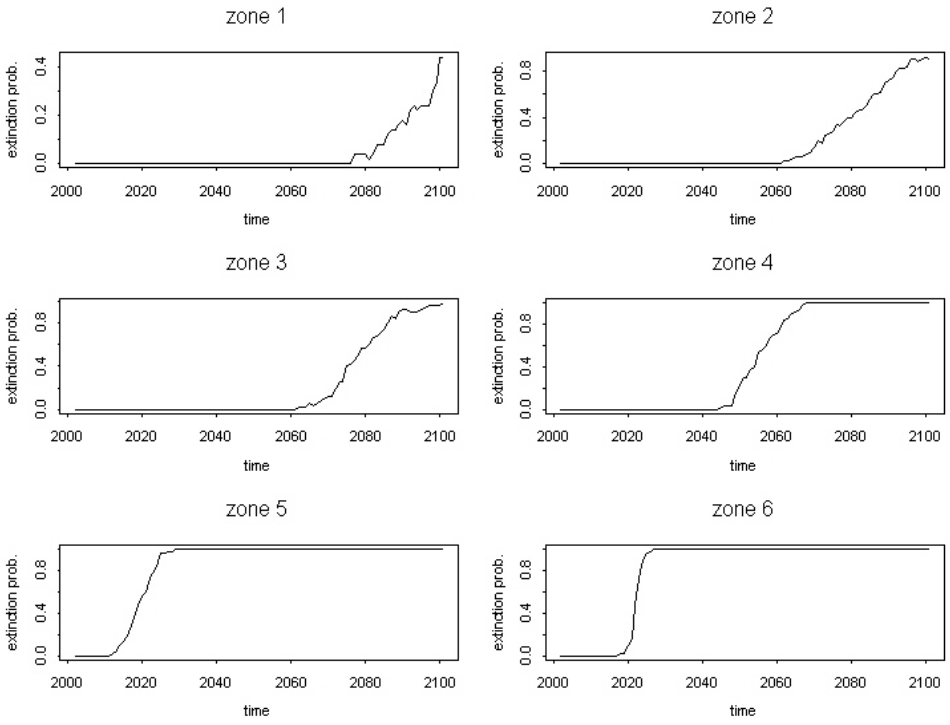


Figure 3.5-6. Probability of population extinction for Zones 1-6 over 100 years, assuming a 5% annual immigration rate, high-end fecundity, and including oil spill and gill-net mortality.

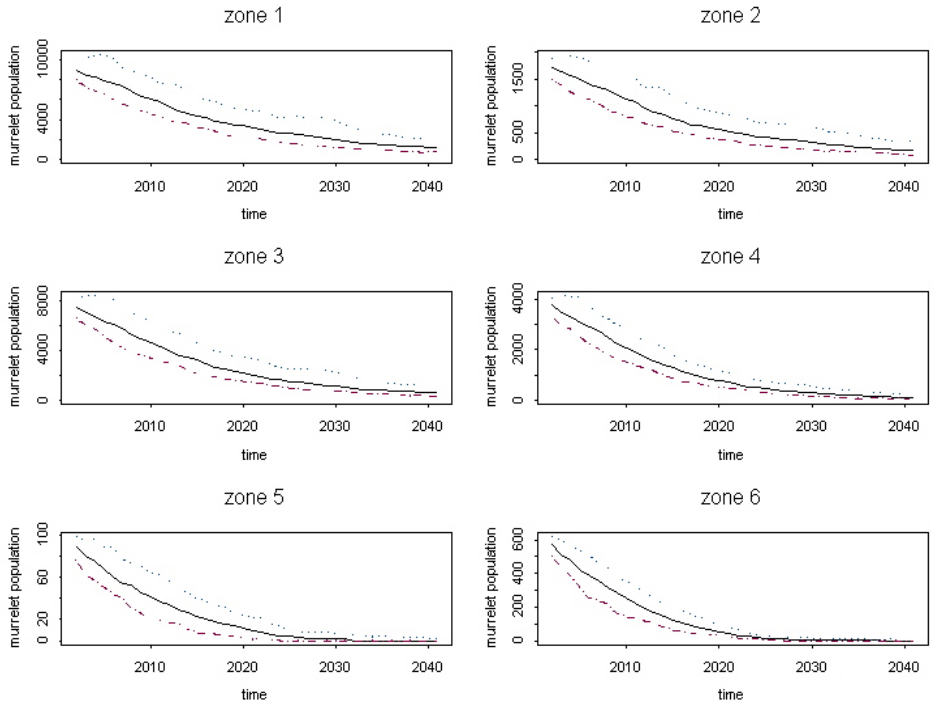


Figure 3.5-7. Population size forecasts for Zones 1-6, assuming a 0.1% annual immigration rate, high-end fecundity, and including gill-net and oil spill mortality.

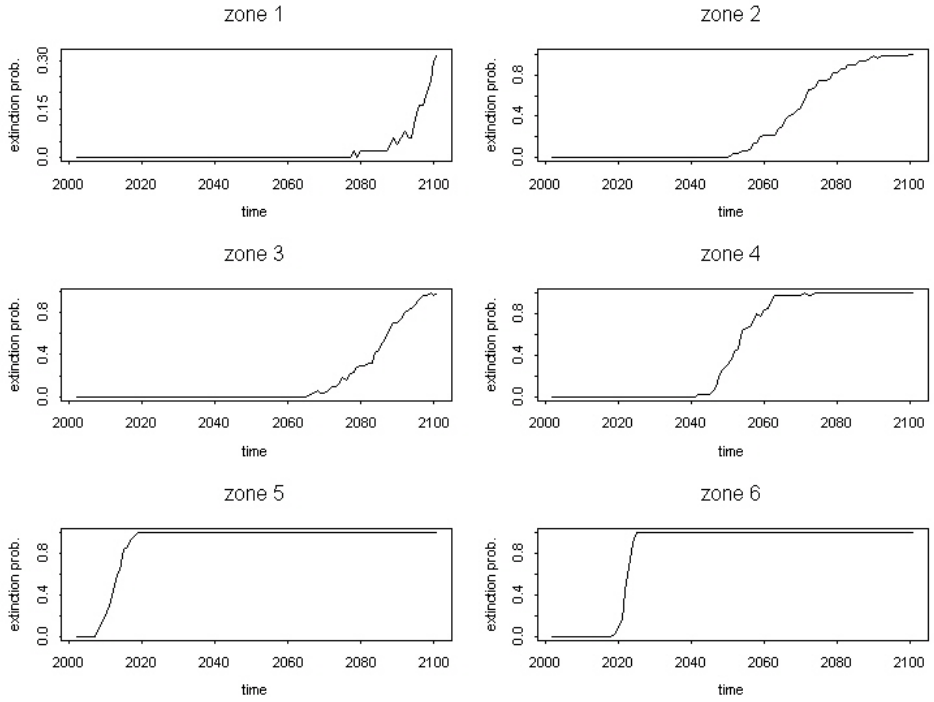


Figure 3.5-8. Forecast of probability of population extinction for Zones 1-6 over 100 years, assuming a 0.1% annual immigration rate, high-end fecundity, and including gill-net and oil spill mortality.

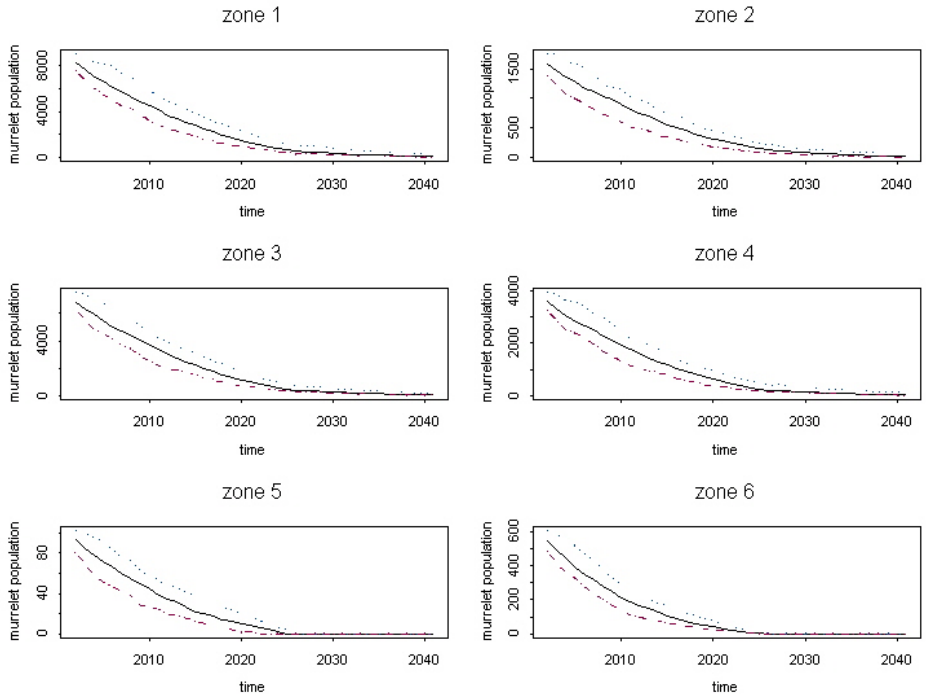


Figure 3.5-9. Forecast of population size for Zones 1-6 over 40 years, assuming low fecundity rates, including oil spill and gill-net mortality and a 2% annual immigration rate.

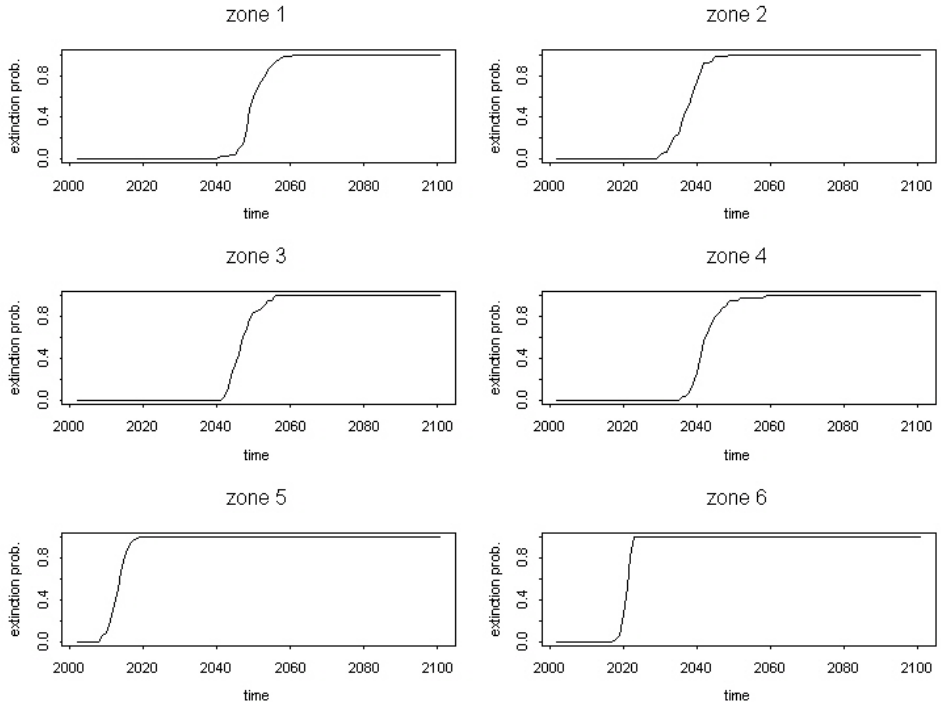


Figure 3.5-10. Forecast of murrelet extinction probability for Zones 1-6 over 100 years, assuming low fecundity, including oil spill and gill-net mortality and a 2% annual immigration rate.

Table 3.5-4. Sensitivity of the rate of population decline (average for first 10 years of forecast) in each zone to immigration rate, level of oil spill and gill-net mortality, and fecundity estimates. The percentage entries marked in **bold** denote the default run with which other runs are to be compared. These include oil spill and gill-net mortality based on 1992-2003 values and fecundity based on telemetry data. High oil spill mortality is set at 1.5 times the 1992-2003 values. The low fecundity values are based on juvenile to adult ratios.

	Annual Immigration Rate			Oil/Gill-Net Mortality		Fecundity
Zone	2%	5%	0.1%	No mort	High mort	Low
1	**3.4%**	3.9%	3.5%	3.4%	3.3%	4.9%
2	**3.0%**	2.4%	3.8%	2.4%	3.0%	4.8%
3	**3.9%**	4.5%	4.1%	3.8%	4.1%	5.0%
4	**4.8%**	4.9%	4.9%	4.7%	4.9%	5.1%
5	**5.0%**	3.0%	5.8%	4.9%	4.9%	5.9%
6	**6.2%**	6.2%	6.1%	5.4%	6.5%	6.5%

extended to Zone 5). However, Zones 3 and 4 had a similar time to extinction as Zones 1 and 2, using low fecundity, whereas Zones 1 and 2 had a later time to extinction and lower extinction probability using high fecundity. Using low fecundity, populations in all zones would be extinct with almost 100% probability within 60 years (i.e., by 2060). Zones 5 and 6 would be extinct with almost 100% probability within 20 years (i.e., by 2020).

These forecasts draw immediate attention to the need for better fecundity estimates in all zones, with focus on improving knowledge of potential biases and their magnitude. At present, we believe that high fecundity values are closer to true fecundity than low fecundity values because potential biases are less for the telemetry techniques used to determine high fecundity, values. However, much debate exists in the research community about the merits and drawbacks of different techniques of calculating fecundity, and much additional work is needed to refine fecundity estimates.

3.5.4.3 Sensitivity to Gill-Net and Oil Spill Mortalities

Future population projections in each zone incorporated rough estimated mortality of murrelets due to oil spills and gill-net bycatch. Estimates were based on recent estimated mortality in 1992-2002, normalized by zone population size. While we expect that these sources of mortality will continue to affect marbled murrelets, it is unclear if zone estimates in the 1992-2002 period will be indicative of future levels of mortality because of the sporadic nature of oil spills and changes in fisheries. As such, we simulated future population response to the unlikely scenario without future gill-net and oil spill mortality loss (Figure 3.5-11 and 3.5-12) and a possible scenario of higher mortality due to gill-net bycatch and oil spills (i.e., 1.5 times 1992-2002 estimates, Figure 3.5-13 and 3.5-14). We note that the latter scenario still may severely underestimate oil and gill-net mortality in Zones 1

and 2 because: (1) no large spills were noted in 1992-2002 but high risk of oil spills occurs in these areas and past mortality in 1977-1991 was much higher; and (2) gill-net fishing effort may increase for a period of time before regulations can be applied to reduce murrelet mortality or before fishing effort again reduces due to low fish catch. Sensitivity forecasts using different scenarios of oil spill and gill-net mortalities on murrelet populations suggest that reducing oil spill and gill-net mortalities can significantly reduce annual rates of decline in Zones 2 and 6, but mortality reductions have less effect on decline in other zones (Table 3.5-4). Extinction probabilities are less affected by oil spill and gill-net mortalities because we implemented these factors as reducing abundance in proportion to the population size. Thus, if the population initially declines more rapidly due to increased oil spill and gill-net mortalities, future population loss will be reduced. Thus, this assumption may lead to very conservative projections of impacts to population size because, as populations decline, they may become more susceptible to such mortalities or may be compromised in other ways by their small size or localized distribution.

3.5.5 Summary

In summary, results of the Zone Model show that:

- All zone populations are in decline with mean annual rates of decline per decade (over 40 years) between 2.1% and 6.2%. Highest rates of decline are forecast for Zone 6 (2.5-6.2%) and lowest rates of decline for Zone 2 (2.1-3.0%). These rates of decline from the Zone Model are generally consistent with earlier models that forecast declines of 4-7%.
- Probabilities of extinction (defined as less than 30 individuals per zone) were calculated as follows: (1) 100% within 40 years for Zones 5 and 6; (2) 0% within 40 years and 100% within 100 years for Zones 2-4; and (3) 0% within 40 years and 25% within 100 years for Zone 1. Viable zone populations may not exist well before reaching extinction.
- Probability of extinction for the 3-state listed population within 100 years is 16%, but viable populations may not exist well before reaching extinction. Mean 3-state population size at 100 years is estimated at 45 individuals, with all birds remaining in Zone 1.
- Projections are sensitive to differing immigration rates (0.1-5%) in Zones 1, 2, 3, and 5. Higher rates of decline and earlier time to extinction can occur in Zones 2 and 5 with slightly lower immigration rates. Lower rates of decline and later times to extinction can occur in Zones 1 and 3 with slightly higher immigration rates.

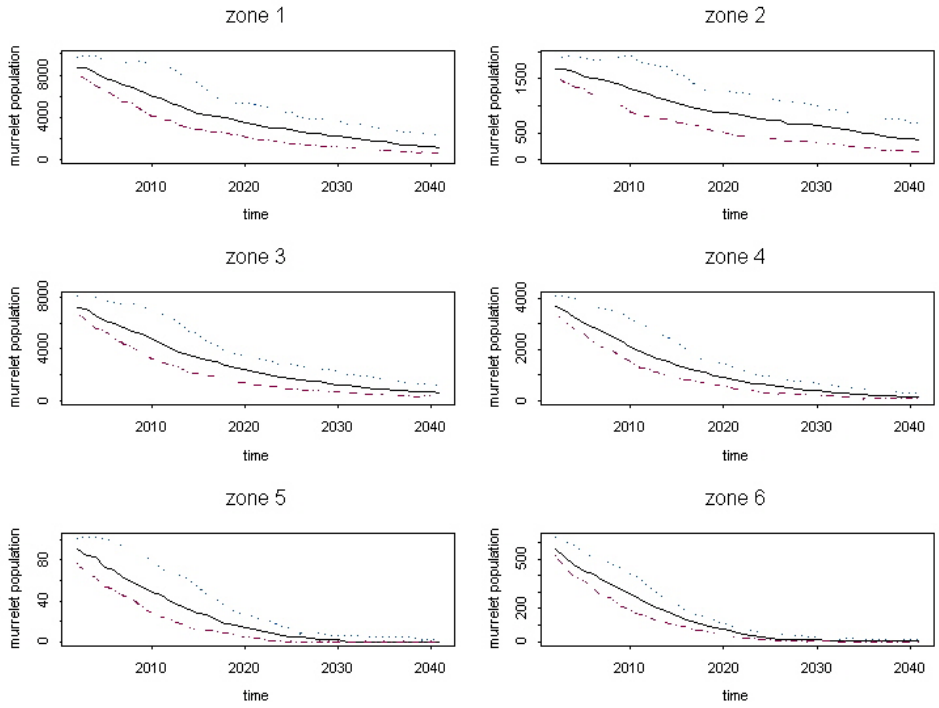


Figure 3.5-11. Forecast of murrelet population for Zones 1-6 over 40 years, assuming high-end fecundity, no gill-net and oil spill mortalities, and a 2% annual migration rate.

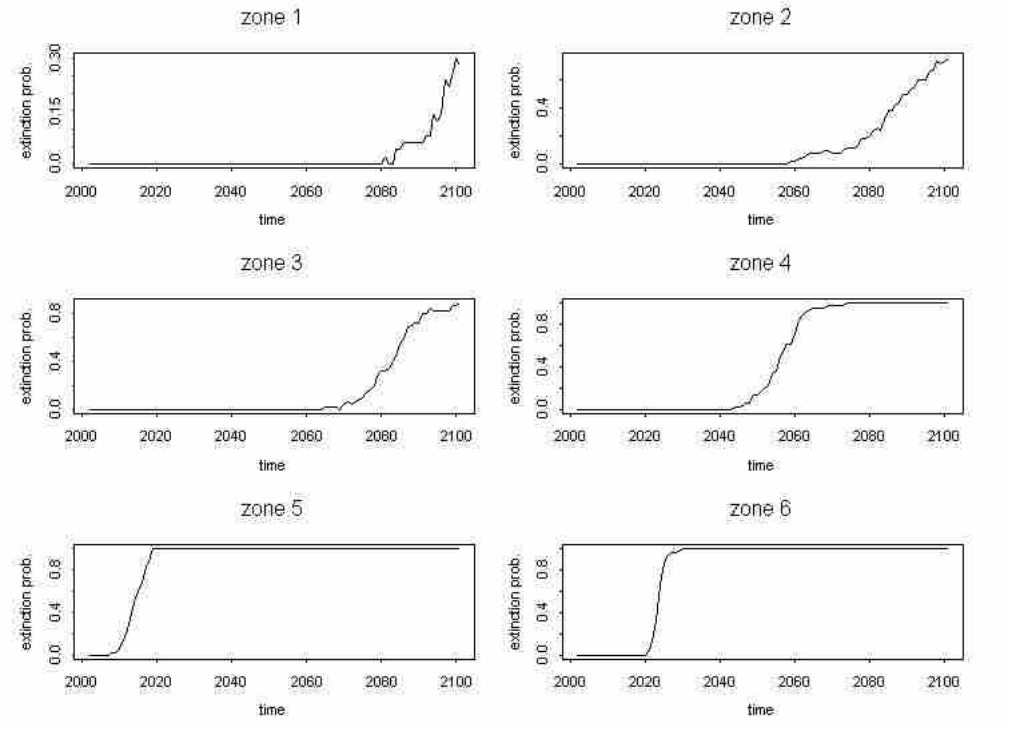


Figure 3.5-12. Forecast of murrelet extinction probability for Zones 1-6 over 100 years, assuming high-end fecundity, no gill-net and oil spill mortalities, and a 2% annual immigration rate.

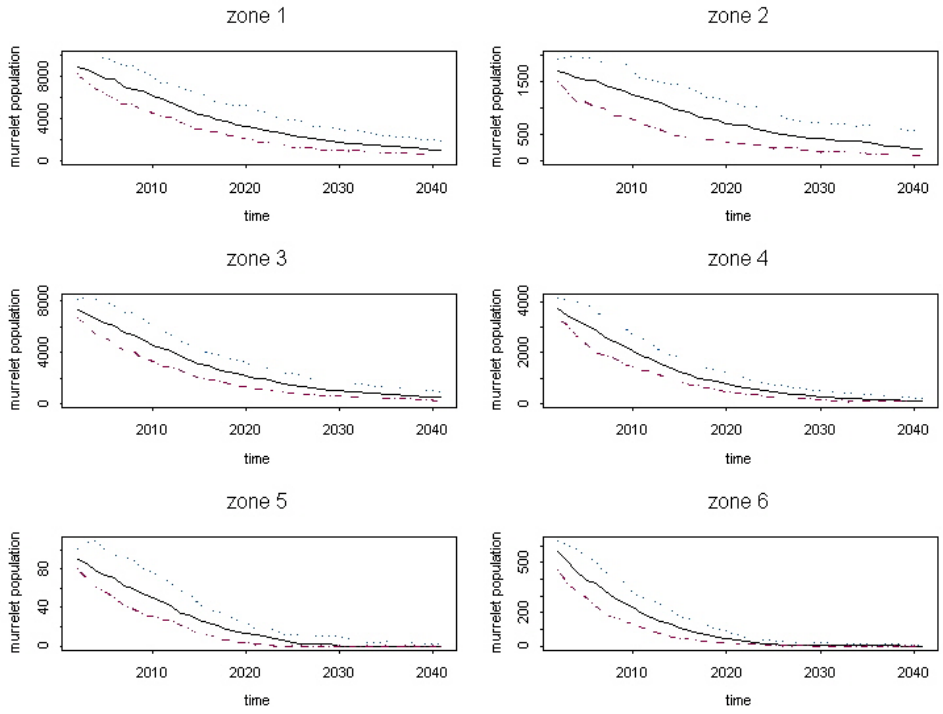


Figure 3.5-13. Forecast of murrelet population for Zones 1-6 over 40 years, assuming high-end fecundity, higher oil spill and gill-net mortality rates, and a 2% annual immigration rate.

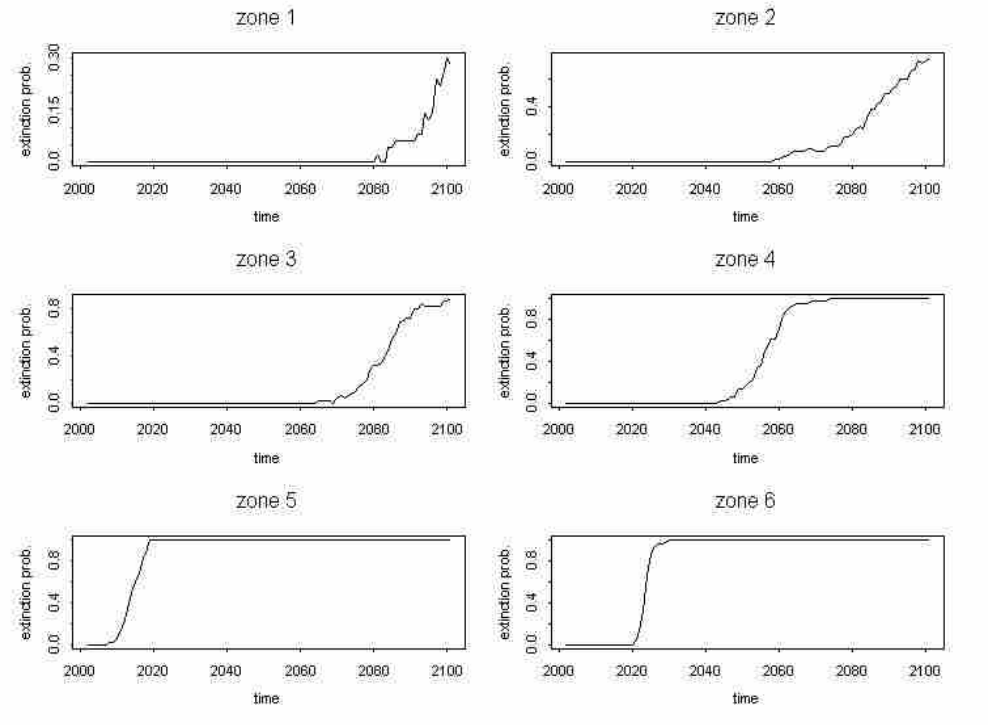


Figure 3.5-14. Forecast of murrelet extinction probability for Zones 1-6 over 100 years, assuming high-end fecundity, higher oil spill and gill-net mortality rates and a 2% annual immigration rate.

- Projections are sensitive to differing estimates of fecundity (telemetry: 0-54%; adult:juvenile ratios: 4-9%) in Zones 1-4. Higher rates of decline and higher and earlier extinction probabilities were found with lower fecundity values in Zones 1-3. Zones 3 and 4 had similar extinction probabilities and time to extinction as Zones 1 and 2, with lower fecundity values. Fecundity values in Zones 5 and 6 were similar using either scenario.
- Reducing oil spill and gill-net mortalities can reduce rates of decline in Zones 2 and 6 but have less effect on other zones. Extinction probabilities for different mortality levels were not comparable due to use of proportionate mortality within the model structure.

3.6 Population and Demographic Threats

As a species, the marbled murrelet is subject to a number of threats unrelated to changes in the marine and terrestrial habitat but possibly having an interactive or cumulative effect. This section describes the threats related to genetics, disease, and disturbance from research activities.

3.6.1 Genetic Threats

Most potential genetic threats to endangered species involve loss of genetic variation. This loss can occur at any of 3 levels: within individuals, among individuals within populations, and among populations. In addition, genetically structured species may not recolonize areas quickly after extirpation, peripheral populations may be especially vulnerable to extinction for genetic reasons, and hybridization may threaten genetically distinct populations and/or species. Research into these 6 problems has expanded greatly over the past 5 years; some but not all of them are concerns for marbled murrelets.

3.6.1.1 Loss of Variation Within Individuals

Genetic variation within individuals is commonly expressed as heterozygosity (H_O). "Inbreeding" is commonly defined as mating between close genetic relatives. It may result either from population decline or from non-random mating, (e.g., if immigration between populations is restricted). Because relatives have a high probability of sharing alleles (inherited from their common ancestor), inbreeding results in a loss of heterozygosity (or conversely, an increase in homozygosity). This loss of heterozygosity can result in reduced fitness ("inbreeding depression") due to expression of deleterious recessive alleles and/or loss of heterozygote advantage[14] (Hansson

[14] Heterozygote advantage refers to fitness advantages that heterozygous individuals sometimes have over homozygotes; e.g., in disease resistance.

and Westerberg 2002), with the severity of inbreeding depression depending in part on the history of inbreeding in the population (Charlesworth and Charlesworth 1987). Some researchers have theorized that inbreeding depression may lead to an extinction vortex, in which a cycle of population decline, inbreeding, and inbreeding depression accelerates extinction (reviewed in Amos and Balmford 2001).

Empirical evidence exists both against and for the occurrence of inbreeding and inbreeding depression in declining populations. In some recent studies, inbreeding coefficients[15] did not increase with population decline at the rate predicted by theory (Amos and Balmford 2001); these findings were attributed to behavioral (mate choice) and/or physiological mechanisms that reduce inbreeding (Jenions and Petrie 2000, Tregenza and Wedell 2000; e.g., guppies, Kelley et al. 1999; seals, Amos et al. 2001a; sparrows, Freeman-Gallant et al. 2003). Some researchers have argued that deleterious alleles ("genetic load") get "purged" from declining populations by inbreeding, and several experiments have reported an actual increase in fitness following several generations of inbreeding, suggestive of genetic purging (e.g., Saccheri et al. 1996). Furthermore, inbreeding depression is generally not detected until population size gets very small, and even a small amount of immigration can reduce the effects of inbreeding depression (Amos and Balmford 2001). However, evidence that inbreeding depression can be a threat to declining populations is strong. Empirical evidence for inbreeding depression in both captive and wild populations is extensive (reviewed in Hedrick and Kalinowski 2000). Purging appears to reduce inbreeding depression only in some traits and in some populations (Hedrick 1994, Ballou 1997, Byers and Waller 1999, Amos and Balmford 2001, Reed et al. 2003)[16]. Several recent studies indicate that inbreeding depression can be compounded by environmental stress (flowering plants, Hauser and Loeschke 1996; *Drosophila*, Dahlgaard and Hoffmann 2000, Reed et al. 2002; mice, Jimenez et al. 1994; sparrows, Keller et al. 1994). And apparently even low levels of inbreeding can result in fitness losses in natural populations (seals, Coltman et al. 1998, 1999, Amos et al. 2001b; deer, Coulson et al. 1998, Slate et al. 2000; sheep, Coltman et al. 1999; albatrosses and whales, Amos et al. 2001b).

Molecular evidence to date provides no indication of inbreeding in marbled murrelets (Section 3.4.2). Thus, inbreeding depression does not appear to pose an immediate threat to this species.

[15] Inbreeding coefficients are indices of the relatedness of an individual's parents, usually calculated by comparing an individual's heterozygosity with heterozygosity predicted under random mating within the population.
[16] This is probably because inbreeding also reduces heterozygote advantage and increases the expression of sublethal alleles.

3.6.1.2 Loss of Variation Among Individuals within Populations

According to Fisher's Fundamental Theorem of Natural Selection, the rate at which a population can respond to selection is directly related to its genetic variability. Because population decline reduces a population's genetic variation through genetic drift and inbreeding, population decline in theory will reduce a population's ability to adapt and evolve. Direct empirical support for a correlation between genetic variation and population fitness is difficult to derive, but theoretical support is strong and many researchers and managers assume that population decline results in loss of adaptability. Some researchers are challenging this assumption, however. In theory, population-level variability is lost at a rate that increases exponentially with time, on the order of $1/(2N_e)^t$ (where N_e is the genetically effective population size[12] and t is time in generations). Thus, calculations suggest that few species of mammals that are declining should have lost more than 10% of their genetic variation through drift (Amos and Balmford 2001). Several species that have undergone recent severe population bottlenecks do not appear to have lost neutral variation[1] (e.g., Antarctic fur seals, Wynen et al. 2000). Furthermore, variation that is important to fitness (e.g., variation associated with disease resistance) will be maintained by selection, so that heterozygosity may actually increase during population decline (Bancroft et al. 1995, Pemberton et al. 1996). Finally, several species have recovered dramatically following severe population bottlenecks (e.g., the Mauritius kestrel, Groombridge et al. 2000).

Nonetheless, some recent empirical research does support a link between genetic variation and population viability. Several studies clearly show a positive correlation between population size and genetic variation (reviewed in Frankham 1998), and loss of variation following population decline has been documented directly in a few natural populations (e.g., the Mauritius kestrel, Groombridge et al. 2000). Importantly, Saccheri et al. (1996) found a higher extinction probability for inbred versus outbred populations in a metapopulation of Glanville fritillary butterflies in Finland. Reed and Frankham (2003) found a significant correlation between population-level measures of genetic diversity and population fitness in a meta-analysis of 34 studies. Furthermore, variation that is maintained by selection during population decline will only be variation that is advantageous during the decline, not variation that may be important during future environmental challenges. Variation associated with immune defense is especially significant in this respect, as increasing numbers of studies indicate that genetic variation buffers populations against disease epidemics (Amos and Balmford 2001, Altizer et al. 2003).

Although reliable estimates of genetically effective population size are not yet available for marbled murrelets, genetic variation in neutral molecular markers is similar to other species of seabirds, including several species with large and/or increasing population sizes such as common murres (Section 3.4.2); thus, loss of population-level variation is not an immediate concern for marbled murrelets.

3.6.1.3 Loss of Variation Among Populations

Theoretically, if local populations differ genetically, loss of a population will reduce the species' genetic resources and therefore its potential to adapt and evolve (see Section 3.6.1.2). The amount of variation that is lost will depend on the extent of genetic divergence among populations, and can be estimated from Wright's F_{st} and its analogs[4-6] for neutral variation[1]. To the extent that differences are adaptive, local adaptations will also be lost. Extensive evidence exists for the loss both of neutral variation (Daugherty et al. 1990) and of local adaptations (Greig 1979) following loss of a population in genetically structured species.

Estimates of F_{st} and its analogs for marbled murrelets (Section 3.4.2.1) indicate that approximately 90% of their neutral genetic variation is contained within populations, and approximately 10% is distributed among populations in California, British Columbia/mainland Alaska, and the Aleutian Islands (Section 3.4.2). Thus, loss of any of these populations will reduce the species' resources and compromise its long-term viability. Furthermore, if differences in nesting habits are genetically based, loss of either type of behavior will represent a loss of adaptive variation.

3.6.1.4 Low Recolonization Potential

In theory, the extent of genetic divergence between populations is inversely related to the amount of immigration between them. Low immigration will lead to strong genetic structuring due to the action of genetic drift and/or selection within local populations, whereas high immigration will counter-act local differentiation, resulting in weak population genetic structure. Thus, species with weak structure should recover from local disturbances as local recruitment is supplemented by immigration, whereas species with strong structure may be slow to recover from local disturbance. Results of studies of seabirds affected by the *Exxon Valdez* oil spill in 1989 support this general prediction: Comparison of mtDNA, microsatellites and introns among common murres from throughout the North Pacific indicated that population genetic structure is essentially zero (Table 3.4-1), and murres are now listed as having recovered from the spill on the basis of population

surveys (Exxon Valdez Oil Spill Trustee Council 2002). A similar analysis of pigeon guillemots revealed relatively strong population structure (Section 3.4.2), and guillemots are currently listed as "failing to recover." Note that, because genetic divergence of populations requires time, this relationship assumes that populations are at equilibrium between mutation, migration, and genetic drift; if a species has undergone recent changes in population size or distribution (as appears to be true of murrelets), population genetic structure will reflect historical conditions rather than contemporary gene flow.

Population genetic structure in marbled murrelets is intermediate between murres and guillemots (Friesen and Piatt 2003; Table 3.4-2), and this species is listed as "recovering" from the *Exxon Valdez* oil spill (Exxon Valdez Trustee Council 2002). Thus, murrelets may recover from local disturbances, but recovery will probably be slow, even under conditions that promote population growth (i.e., adequate breeding success, survival, and immigration). If genetic differences among populations have arisen recently (Section 3.4.2.1), contemporary immigration may be very low, and the species may be very slow to recover from local disturbances. Studies in southern Alaska also occur within the central part of the breeding range with much higher populations and likely greater levels of breeding success and immigration than occur in the southern portion of the range. Thus, recolonization potential in the southern portion of the range is likely to be lower than in the central portion of the range.

3.6.1.5 Vulnerability of Peripheral Populations

Given their generally small size, relative isolation, or less suitable habitats, peripheral populations are expected to be especially vulnerable to extinction. Furthermore, peripheral populations often house unique genetic variation that may be adaptive, and may be the source of future speciation events (reviewed in Lesica and Allendorf 1995). Empirical studies of peripheral populations are extensive. Although a few studies either found higher diversity in peripheral populations or failed to find the expected relationship (reviewed in Lesica and Allendorf 1995, Vucetich and Waite 2003), most found reduced fitness (e.g., Jump and Woodward 2003), as well as decreased genetic diversity and increased genetic divergence (e.g., Hundertmark et al. 2003, Jump et al. 2003, Schwartz et al. 2003).

Marbled murrelet populations in the Aleutian Islands and California are genetically different and may be considered to be peripheral in a genetic sense because they correspond with the northern and southern portions of the current breeding range. However, within the context of geography,

ecology, and demographics, we have considered only certain California sub-populations (Zones 5 and 6) and the western part of the Aleutian Islands population as truly occurring at the peripheries of the breeding range. Given that nesting or feeding habitats are reduced and/or discontinuous in these areas, population sizes are low, and populations differ genetically from those in central parts of the range, murrelets in the western Aleutian Islands and central and north-central California are especially vulnerable to extinction.

3.6.1.6 Interspecific Hybridization and Introgression

Interspecific hybridization can have serious implications for conservation (Pacheco and Friesen 2002). Several behavioral and morphological observations suggest that marbled and Kittlitz's murrelets may hybridize: they often form large aggregations in areas of sympatry; they sometimes occur in mixed pairs that remain together when disturbed; birds with plumage intermediate between the 2 species have been observed in areas of sympatry; and hybridization has been documented in other alcids (Pacheco and Friesen 2002). In 2002, Pacheco and Friesen compared variation in the mitochondrial cytochrome *b* gene and 5 nuclear introns among 131 marbled murrelets sampled between Attu Island and British Columbia, and 17 Kittlitz's murrelets from Kachemak Bay. They found no evidence either of F1, F2, or back-cross hybrids, or of historical introgression[17] between the species. However, their sampling included only a small number of Kittlitz's murrelets, and no birds with unusual plumage. Thus, the importance of interspecific hybridization is unknown, and further studies should be done.

3.6.2 Disease

The recent emergence of diseases in free-ranging birds in coastal marine systems is an indicator of declining ecological integrity. Diseases in seabirds are expected to increase significantly in the near future as ecological stressors in the marine environment, primarily coastal pollution, increase (Epstein et al. 1998a,b; Friend et al. 2001; S. Newman, pers. comm.). Combined with other environmental stressors such as ocean climate changes, overfishing, and habitat loss, diseases will be especially significant with respect to declining or threatened and endangered species. These species may not recover from single-event losses or the cumulative effects of a variety of diseases and other threats (Friend et al. 2001).

The recent emergence of bacterial, fungal, parasitic, and viral diseases and biotoxins (from harmful algal blooms) has affected numerous populations of

[17]Introgression is the transfer of genetic material between species following hybridization.

seabirds, including a variety of penguins, loons, albatross, shearwaters, tropicbirds, sulids, pelicans, cormorants, eiders, sea ducks, gulls, and terns (Epstein et al. 1998a,b; Friend and Franson 1999; Kuiken 1999; Friend et al. 2001; NWHC 2003; S. Newman, pers. comm.; Table 3.6-1). No studies have been conducted on diseases in marbled murrelets, and no diseases have been documented to have caused murrelet mortality. Four murrelets were suspected to have died from domoic acid toxicosis (a biotoxin produced by an algal bloom) in central California in 1998 (Burkett et al. 1999; Newman and Burkett in prep.; E. Burkett, pers. comm.; Table 3.6-1), but insufficient evidence was obtained to fully support this diagnosis, and El Niño and/or other factors may be involved. In British Columbia, MacBean (1989) documented 2 juvenile murrelets killed by paralytic shellfish poisoning in 1989. Several parasites have also been found on or in dead murrelets in California but were not suspected to be the ultimate cause of mortality (pers. comm., E. Burkett, pers. comm.; Table 3.6-1). While extensive blood health screens have been conducted on murrelets to evaluate biochemical parameters and cell counts, no red blood cell parasites were observed (Newman 1998).

The National Wildlife Health Center of the U.S. Geological Survey (USGS) has been tracking the effects of West Nile virus (WNV) since its discovery in the United States in 1999 (NWHC 2003). WNV has caused extensive mortality in more than 160 avian species, particularly corvids and raptors, but also in the seabirds listed in Table 3.6-1. Birds (especially corvids) are the natural host and reservoir of WNV, allowing mosquitoes to spread the virus to other birds inhabiting the same area (NWHC 2003). As of 2003, West Nile Virus has been confirmed in southern California (in humans, birds, and mosquitoes) and western Nevada (in humans), and samples of dead birds and mosquitoes have been submitted from Washington, Oregon, and California for testing (CDC 2003, NWHC 2003, USGS 2003). It is believed that it is only a matter of time before the virus is present throughout California and the Pacific Northwest (NWHC 2003).

Many species of forest birds have been documented to have WNV, including, but not limited to, Steller's jays, common ravens, black-capped chickadees (*Poecile atricapillus*), American robins (*Turdus migratorius*), yellow-rumped warblers (*Dendroica coronata*), song sparrows (*Melospiza melodia*), and purple finches (*Carpodacus purpureus*) (CDC 2003, NWHC 2003). Because marbled murrelets nest in forests where mosquitoes are present, and in some cases abundant, they are thought to be as susceptible to WNV as other forest bird species, and potentially more susceptible than other seabird species (S. Newman, pers. comm.). Habitat loss, habitat fragmentation, predation, energetic costs, and other stressors in or related to their forest environment

Table 3.6-1. Documented emergent diseases in a variety of seabird species.[1]

Species	Viruses		Bacteria [3]	Fungi [4]	Parasites [5]	Biotoxins [6]
	West Nile	Other [2]				
Marbled murrelets					X	X
Other alcids [7]		X	X		X	X
Penguins		X	X	X	X	
Loons	X		X		X	X
Albatross					X	
Shearwaters [8]		X			X	X
Tropicbirds		X				
Sulidae [9]						X
Pelicans	X	X	X		X	X
Cormorants	X	X			X	X
Eiders			X	X	X	
Sea ducks[10]		X	X		X	
Gulls	X	X	X	X	X	X
Terns	X	X	X	X	X	X

[1] *Cells include Xs for diseases found in the literature reviewed. These species could be found to be affected by other disease categories with further research.*
[2] *Includes Newcastle (*Rubulavirus *sp.), Avian Pox (diphtheria, contagious epithelioma*, *poxvirus),* E. Equine Encephalomyelitis *(arbovirus), and Influenza.*
[3] *Includes Avian Cholera (*Pasteuella multocida*), Avian Tuberculosis (*Mycobacterium avium*), Salmonellosis (*Salmonella *sp.), Chlamydiosis (*Chlamydia *sp.), and Erysipelas (*Erysipelothrix rhusiopathiae*).*
[4] *Includes Candidiasis (*Candida albicans*) and Aspergillosis (*A. Fumigatus, A. Flavus *and others).*
[5] *Includes Intestinal and Renal Coccidiosis (*Eimeria *sp.), Sarocystis (*S. rileyi, S. *sp.), Eustronylidiosis (*Eustrongylides *sp.), Hemosporidiosis (*Plasmodium *sp.,* Haemoproteus *sp.,* Leucocytozoon *sp.), worms (*Amidostomum *sp.,* Epomidiostomum *sp.,* Cheilospirura spinosa*), and leeches (*Theromyzon *sp.).*
[6] *Includes Avian Botulism (*Clostridium botulinum*) and Harmful Algal Blooms (HAG) such as* Alexandrum tamarensee*,* Gymnodinium breve*,* G. Sanguinium*, and Domoic Acid (usually caused by* Pseudontizchia *sp.).*
[7] *Includes guillemots, murres, and puffins.*
[8] *Includes shearwaters and fulmars.*
[9] *Includes boobies and gannets.*
[10] *Includes scoters and the harlequin duck (Histrionicus histrionicus).*

could add to their susceptibility to WNV. Conversely, it is possible that murrelet nest location (solitary, high in trees), breeding behavior (relatively short nest attendance except during incubation), and foraging behavior (solitary or in pairs) may reduce their susceptibility to WNV and/or lower transmission risk within populations.

On a potentially positive note, because corvids seem to be the most susceptible to WNV, declines in these species may benefit murrelet populations. Predation by corvids has been documented as a significant cause of nest failure. It is possible that decreased corvid populations may allow murrelets greater nesting success. Any increase in nesting success by murrelets could be temporary if corvid populations rebound quickly, which might be expected by a species with high nesting success and large clutch size.

In conclusion, murrelets will likely be affected by 1 or more diseases or biotoxins in the near future because of the cumulative effects of stressors in both their marine and forest environments. The potential for murrelets to

contract WNV seems high given their nesting habits, the abundance of mosquitoes in forests, the presence of the disease in other forest-nesting bird species, and the proximity of many murrelet nesting areas to open fields, clearcuts, or areas of human activity where corvids are abundant. The most important question is what the potential impact of these diseases will be on murrelet population viability. Intensive monitoring and research (e.g., testing dead and live birds) should be conducted to determine the potential impacts of disease on murrelet survival and recovery.

3.7 Summary and Conclusions

This section summarizes the information on marbled murrelet population and demographics discussed in Chapter 3.

3.7.1 Demographic Characteristics

Current research confirms the following data regarding murrelet demographic characteristics:

- Breeding begins at 2-5 years.
- Replacement eggs for a failed first clutch occur, but the extent of this behavior is not known.
- Adult survivorship is estimated to be between 0.83 and 0.93.
- Limited data on breeding success indicates a rate ranging from 0-0.46.
- Low levels (~0.10) of non-breeding adults probably occur in marbled murrelet subpopulations.

Previously, laying of replacement eggs had not been known, and the age when breeding begins had been considered to be 3 years. Previous estimates of adult survival had relied on estimates from other alcids and were slightly lower than current estimates.

3.7.2 Distribution and Densities of At-Sea Populations

To estimate population size of marbled murrelets, researchers must rely on at-sea counts as terrestrial data are difficult to collect on a wide scale and extremely limited. The entire North American population of the marbled murrelet is currently estimated to be around 950,000 birds (Huff et al. 2003). Earlier estimates ranged from 300,000 (Ralph et al. 1995) to 600,000 birds (reviewed in DeGange 1996). An increase in the total population of

murrelets from earlier estimates reflects the development and consistent use of a standardized survey protocol and increased research into marbled murrelet demographics and population since the mid-1990s. Washington, Oregon, and California currently support a combined 2.3% of the total population, or about 21,900 birds.

Five areas off the Washington, Oregon, and California coast exhibit a discontinuity in murrelet at-sea distribution, indicating the existence of groupings, or sub-populations over the 3-state range. These 5 areas are:

- The British Columbia-Washington border region, which includes the area between Howe Sound, BC and the San Juan Islands and the deeper waters of the Strait of Juan de Fuca;
- Southern Puget Sound, WA;
- Destruction Island, WA to Tillamook Head, OR;
- Humboldt County to Half Moon Bay, CA; and
- Southern end of breeding range in central California (southern Santa Cruz and Monterey Counties).

Despite these gaps in distribution, moderate at-sea densities of marbled murrelets occur along the coast of the 3-state region, although at lower densities than in Alaska and British Columbia.

3.7.3 Population Trends from Past Studies

Murrelet numbers declined drastically in relation to historic (1850-1980) logging in Washington, Oregon, and California. Current trend data are limited because comprehensive, standardized studies for all of Washington, Oregon, and California have only begun in the past few years. Survey data from the EM Program for the NWFP have been collected only for 3 years, and the analyses resulting from these data have large confidence intervals; thus, patterns apparent in the data do not constitute statistically determined trends. However, trend data for small areas indicate declines in murrelet numbers; no available trend data indicate that murrelet populations are increasing. Reliable trend data from the Oregon coast between 1992 and 1999 show an abrupt decline in 1996, but estimates for this area have remained stable since then.

The best long-term data are from Prince William Sound, which has shown a significant decline in murrelets since 1972. Long-term ecosystem changes

and the *Exxon Valdez* oil spill are considered to be the major causes of decline. Since 1972, major declines (22-73%) in populations over a period of a decade or more have been documented in Alaska, British Columbia, and Oregon with no evidence of increase.

3.7.4 Morphologic and Genetic Variation

Morphological variation among murrelets has been studied only to a limited extent. The few studies completed did not discern significant differences between murrelets from areas in British Columbia and no morphological differences were observed between ground-nesting and tree-nesting murrelets in Alaska. Morphology is but 1 indicator of differentiation between populations and behavioral differences are a factor that needs to be considered during an analysis of ecological distinction between populations. No studies have been conducted comparing the morphology of birds from the extreme ends of the marbled murrelet's range. However, results from genetic studies suggest that populations from California and the Aleutian Islands differ from each other and from populations in Alaska Peninsula/British Columbia. Studies have not yet been conducted on murrelets from Washington and Oregon.

3.7.5 Demographic Modeling

All previous models suggest population declines of 4 to 7% per year for marbled murrelets in Washington, Oregon, and California. Previous modeling efforts concentrated on the development of deterministic Leslie Matrix models. These stage-based models have some limitations based on their relatively simplistic design. The Zone Model developed for this Evaluation Report uses:

- The most recent data on breeding success, survival, and zone population size;
- Improvements in the model structure to reduce certain biases; and
- Scenarios that account for oil and gill-net mortality and immigration.

Future projections in certain zones are sensitive to breeding success, immigration, and oil/gill-net mortality. Specifically, the Zone Model indicates that:

- All Conservation Zone populations are in decline, with mean annual rates of decline per decade (over 40 years) between 2.1 and 6.2%, with highest rates of decline in Zone 6 (2.5-6.2%).

- The probability of extinction for populations in Zones 5 and 6 is 100% within 40 years.

- The probability of extinction for populations in Zones 2-4 is 0% within 40 years and 100% within 100 years.

- The probability of extinction for Zone 1 is 0% within 40 years and 25% in 100 years.

- The probability of extinction in the 3-state listed population is 16% in 100 years. Mean population size at 100 years was estimated to be 45 birds, all remaining in Zone 1.

- Loss of population viability occurs before extinction occurs, but we could not estimate when or at what population level that populations become non-viable.

3.7.6 Population and Demographic Threats

For marbled murrelets in Washington, Oregon, and California, the primary population and demographic threats include:

- **Loss of Genetic Variation Among Populations**. Given that there are at least 3 genetically distinct populations of marbled murrelets, loss of any of these populations would reduce the species' genetic resources and compromise its long-term viability.

- **Low Recolonization Potential**. Low immigration rates and concomitant strong genetic structure in murrelets indicate that the species would probably be slow to recover from local disturbances.

- **Declining Populations**. Modeled trends indicate that marbled murrelet populations in Washington, Oregon, and California are in a state of decline. The probability for extinction is especially prominent for marbled murrelets in Zones 5 and 6.

- **Disease**. Recent emergence of bacterial, fungal, parasitic, and viral diseases and biotoxins in seabirds poses an increasing threat to marbled murrelets. In addition, the recent expansion of West Nile Virus to the western United States poses an additional threat to nesting murrelets from mosquitoes in forest habitats.

Loss of genetic variation among individuals and inter specific hybridization do not appear to be current threats to murrelets.

4.0 TERRESTRIAL HABITAT

4.0 TERRESTRIAL HABITAT

Since the marbled murrelet was listed in 1992, studies of terrestrial habitat suitability from British Columbia to California have consistently confirmed that, in general, murrelets select old-growth forests for nesting, typically within about 37 miles (60 km) of the coast (Lank et al. 2003). The principal working definition of "suitable habitat" for nearly all studies of murrelets has relied on use versus non-use of an area. More refined analyses of habitat suitability have examined murrelet use at different scales, ranging from the nest site and tree to the large landscape scale. In addition, some recent studies have begun to examine nest success and nest density as measures of habitat suitability.

The first section of this chapter describes the estimated amount and distribution of marbled murrelet habitat throughout the listed range based on information available at the time of the 5-Year Status Review. Section 4.2 discusses the methods and sources of error associated with detecting murrelets in terrestrial habitat; Section 4.3 summarizes marbled murrelet habitat characteristics at various spatial scales based on various methods and models; and Section 4.4 presents data on murrelet densities at inland sites. Finally, threats to marbled murrelets in terrestrial habitat are presented in Section 4.5.

4.1 Habitat Amount and Distribution

Though our knowledge of murrelet habitat use has increased, the ability to generate accurate estimates of the amount of suitable habitat across the listed range of the species remains hampered either by inadequate vegetation cover databases, or as in the case of private lands in Oregon, the absence of such databases. The lack of accurate habitat maps, combined with the continually evolving understanding of the species' nesting ecology, limits the degree to which we can analyze habitat availability within the listed range of the species. Nevertheless, rough estimates of the amount of suitable marbled murrelet habitat in the listed range are available. We refined these estimates to generate the amount of existing occupied habitat in the 3-state area by quantifying the amount of known occupied habitat, as determined by survey data, and applying occupancy indices to unsurveyed suitable habitat.

4.1.1 Data Sources and Limitations

Data on estimated amounts of suitable marbled murrelet habitat were summarized from 16 sources, including submittals from Federal and State and Service files (e.g., consultation records) for the 5-Year Status Review. Although these data represent the best available information, they consist of a wide variety of habitat descriptions. These descriptions range from estimates based on vegetation classes only with no size, age, or structural classification, to more precise estimates using surveys of the entire ownership and mapped locations of all platform trees. In some cases, estimates of northern spotted owl (*Strix occidentalis*) habitat were used to approximate available murrelet habitat. While there is overlap between the 2 species in habitat use, the spotted owl is known to use younger forests that are not likely to have suitable nesting characteristics for murrelets. Thus, estimates of suitable habitat are dependent on the definitions used by the many administrative units that provided the data and likely overstate the amount.

Further refinement of suitable marbled murrelet habitat comes from an attempt to describe relative habitat quality. This description has typically involved applying usage per area estimates, or comparisons of numbers of detections of murrelets. In particular, detections of behaviors indicating occupancy of an area have been documented at different rates for different habitat types, suggesting that high quality habitats may be defined based on their relative utilization. Most research supports the hypothesis that stands with more old-growth trees have higher rates of occupancy and provide more nesting opportunities and better cover from predators and adverse weather (Ralph et al. 1995). Indeed, managers and researchers have made numerous decisions on the value or relative quality of habitat based on various indirect measures including, but not limited to, survey results, stand size and configuration, number of platform trees (structure), timber volume and stem density, and a combined measure of canopy closure and second growth sub-canopy layers (e.g., Pacific Lumber Company [PALCO] Habitat Conservation Plan (HCP) in California, Elliott State Forest HCP in Oregon, and the Washington Department of Natural Resources [WDNR] HCP in Washington).

Data on the indirect measures of relative habitat quality described above were not available on a large landscape scale; however, we did attempt to gather information on 1 direct measure of habitat quality—the amount of suitable older, un-managed stands that are high quality versus stands with a remnant or residual older tree component (e.g., stands of old-growth from which commercial timber has been selectively removed at some point in the past) that provide relatively lower quality habitat. Unfortunately, this

information was not consistently available and does not allow for a reasonable discussion of relative habitat quality.

Habitat estimates from private lands were difficult to obtain and are not comprehensive for the region. In some areas, such as California, much of the privately owned habitat is quantified; in other areas, such as Oregon, information is completely lacking. In Washington, the submission by the Washington Department of Fish and Wildlife (WDFW) included only some occupied acres on private lands. It did not include suitable habitat that has been surveyed but not found to be occupied, nor suitable habitat that has not been surveyed. In addition, this estimate likely does not include all occupied habitat in the state because it is apparent that at least 1 major land owner did not submit survey information to WDFW.

One recommendation of the Expert Panel is to complete the initial habitat modeling and mapping that is being conducted under the Effectiveness Monitoring Plan for the NWFP (Madsen et al. 1999) as quickly as possible. That effort would provide a much needed accurate baseline of murrelet nesting habitat at a landscape level in the 3-state area and will be important for future monitoring.

4.1.2 Estimated Suitable Habitat

Based on the available data, the Service estimates that as of 2003, slightly more than 2.2 million acres (890,312 ha) of suitable marbled murrelet nesting habitat occur within the listed range (Table 4.1-1, Figure 4.1-1). Table 4.1-1 shows current estimates of suitable habitat by state, Marbled Murrelet Conservation Zone (Zone) (in some cases estimates had to be assigned to 2 adjacent zones), and ownership category. The estimate of suitable murrelet habitat in Table 4.1-1 is a general approximation that cannot be directly compared to other estimates that have been generated over the last 11 years due to the evolving definition of suitable habitat and methods used to quantify habitat. In addition, this is likely an overestimate given the lack of details on platform presence on many ownerships.

Washington State contains approximately 48% of the suitable habitat in the 3-state area, with Oregon and California containing approximately 35% and 17%, respectively (Table 4.1-1). Furthermore, Table 4.1-1, along with the data presented in Section 3.2, illustrates that there is a positive correlation between the amount of suitable nesting habitat available and estimates of murrelet population size from at-sea surveys at the Conservation Zone scale. Conservation Zone 1 contains the largest amount of habitat and the largest

population, while Zones 5 and 6 contain the least amount of habitat and the smallest populations.

The relationship between estimates of murrelet population size and habitat amounts by Conservation Zone were examined using the Pearson correlation statistic (SPSS 10.0 1999). For habitats recorded in 2 Conservation Zones, we divided the habitat equally between each zone to derive estimates of total habitat. Estimates of remaining habitat by Conservation Zone were correlated with the number of birds estimated at sea (Huff et al. 2003) during the breeding season ($n=6$, $r=0.913$, $P=.006$) (Figure 4.1-2). Similar relationships at the watershed scale have been reported from several studies using radar counts to estimate population size (see Section 4.4.1).

The estimates of suitable murrelet habitat in Table 4.1-1 for Washington and California are fairly complete for most land ownerships; however, we were unable to estimate suitable murrelet habitat for privately owned lands in Oregon and could not account some private lands in Washington. To assess the magnitude of missing habitat information, the Service completed a geographic information system (GIS) analysis of total acreage in private commercial forest land ownership in Oregon within 50 miles (80 km) from the coast. The data layer used was Forest Ownership Western Oregon–Western Oregon Industrial Land Ownership. Assembled from various sources, it is available on the internet (GEO 2003; www.sscgis.state.or.us). Based on these data, it appears that there are 2,709,516 acres (1,096,507 ha) of commercial forest lands within the range of the murrelet in Oregon (50 miles [80 km] inland from the coast). The vast majority of that ownership (87%) occurs within 35 miles (56 km) of the coast, which coincides with most known murrelet sites in Oregon. Thus, a relatively large amount of land within the murrelet's range in Oregon is in private ownership. The Service is aware of some occupied habitat on private lands in Oregon, but for the most part, the amount of suitable habitat on these lands is unknown. Due to the lack of State regulation of harvesting in murrelet habitat on private lands in Oregon and the failure to require pre-project surveys, it is likely that most suitable habitat has been lost from these lands over the 11 years since listing.

The data presented in Table 4.1-1 take into account several recent range contractions (Oregon and California) that resulted from additional surveys conducted since 1995 where no birds were detected. These surveys and their results are summarized below.

- Inner North Coast Ranges of California – Surveys for murrelets in the inner North Coast Ranges of California were conducted in 1995, 1996,

Table 4.1-1. Estimates of suitable and likely to be occupied marbled murrelet habitat in 2003, as summarized from local land managers[1]

State	Conservation Zone	Ownership	Unit	Habitat Acreage				Occupancy Index[3]	Data Source
				Suitable	Remnant	Total	Likely to be Occupied[2]		
CA	6	State/private	State & County Parks & private land	7,250	unknown	7,250	7,250	1.00	PALCO HCP BO
CA	5	Federal-NPS	Point Reyes NRA	85	unknown	85	0	0.00	NPS submission (H. Sakai, pers. comm.)
CA	5	Federal-NPS	Golden Gate NRA	269	unknown	269	0	0.00	NPS submission (H. Sakai, pers. comm.)
CA	5	Federal-NPS	Muir Woods NM	343	unknown	343	0	0.00	NPS submission (H. Sakai, pers. comm.)
CA	5	State/private	State Parks & MRCo.	430	unknown	430	430	1.00	PALCO HCP BO
CA	4	Federal-BLM	Arcata NF	25,640	unknown	25,640	4,045	0.16	BLM CA State Office submission (Acting State Director, pers. comm.)
CA	4	Federal-USFS	Six Rivers NF	191,548	26,005	217,553	7,001	0.03	R5FS subm-update Q. Youngblood-ac in RDA not inc
CA	4	Federal-USFS	Klamath NF	85,976	0	85,976	0	0.00	R5 FS submission- acres in RDA not included
CA	4	Federal- NPS	Redwood NP	40,920	360	41,280	41,280	1.00	NPS submission - <1 ac habitat estimated degraded
CA	4	private	Pacific Lumber	1,087	2,397	3,484	3,484	1.00	PALCO HCP BO- Mamu Conservation Areas/FEIS
CA	4	private	Simpson & Stimpson	699	unknown	699	699	1.00	PALCO HCP BO + FEIS
CA	4	State- CDFG	Owl Creek	753	unknown	753	753	1.00	CDFG submission
CA	4	State- State Park	Grizzly Creek	1,221	unknown	1,221	1,221	1.00	PALCO HCP BO, CDFG submission +FEIS
CA	4	Tribal-Hoopa	Hoopa Reservation	6,062	unknown	6,062	213	0.04	M. Higley , pers. comm. 10-21-03-ac in RDA not incl.
CA	4	Tribal-Yurok	Yurok Reservation	2,541	unknown	2,541	250	0.10	Weitchpec Boundary Timber Sale "A" BA, March 2003; PALCO HCP BO
			State total	**364,824**	**28,762**	**393,586**	**66,626**		
OR	3&4	Federal-BLM	Roseburg	82,638	8,290	90,928	Unknown	n/a	BLM Oregon State Office submission (E. Borg, pers. comm.)
OR	3&4	Federal-BLM	Coos Bay	96,359	4,000	100,359	60,696	0.60	BLM Oregon State Office submission (E. Borg, pers. comm.); email from J. Guetterman, August 2002
OR	4	Federal-BLM	Medford	5,751	unknown	5,751	w/ Siskiyou est	n/a	BLM Oregon State Office submission (E. Borg, pers. comm.)
OR	4	Federal-USFS	Rogue River /Siskiyou NFs	95,165	unknown	95,165	20,000	0.21	R6 FS (L. Goodman, pers. comm.); L. Webb, pers. comm.
OR	3	Federal-BLM	Salem	31,720	79,930	111,650	23,000	0.21	BLM Oregon State Office submission (E. Borg, pers. comm.); email from S. Hopkins, August 2002
OR	3	Federal-BLM	Eugene	41,440	unknown	41,440	34,582	0.83	BLM Oregon State Office submission (E. Borg, pers. comm.); email from D. Huber, August 2002
OR	3	Federal-USFS	Siuslaw NF	248,546	21,797	270,343	270,343	1.00	R6 FS submission (L. Goodman, pers. comm.)
OR	3	State-ODF	Oregon Department of Forestry	79,072	unknown	79,072	Unknown	n/a	ODF submission (R. Mannix, pers. comm.)
			State total	**680,691**	**114,017**	**794,708**	**408,621**		
WA	1&2	Federal-USFS	Olympic NF	247,672	12,059	259,731	No est. avail	n/a	R6 FS submission (L. Goodman, pers. comm.); L. Webb, pers. comm..
WA	1&2	Federal-NPS	Olympic NP	402,785	unknown	402,785	322,228	0.80	NPS submission (H. Sakai, pers. comm.)
WA	1&2	private	multiple lands	11,068	unknown	11,068	11,068	1.00	WDFW submittal (J. Buchanan, pers. comm.) - 473 ac removed harvest/blowdown
WA	1&2	Tribal- Makah	Makah Reservation	3,600	unknown	3,600	No est. avail	n/a	BO Makah For. Mgmt Plan 1999-2008 (1-3-99-F-0534)
WA	1&2	State WDNR	Department of Natural Resources	22,132	unknown	22,132	No est. avail	n/a	BO WDNR HCP - total acres minus take:148,572-74,286-52,154
WA	1	Federal-USFS	Mt. Baker Snoqualmie NF	227,314	unknown	227,314	No est. avail	n/a	R6 FS submission (L. Goodman, pers. comm.)
WA	1	Federal-USFS	Gifford Pinchot NF	16,071	unknown	16,071	No est. avail	n/a	R6 FS submission (L. Goodman, pers. comm.)
WA	1	Federal-NPS	San Juan NHP	39	unknown	39	0	0.00	NPS submission (H. Sakai, pers. comm.)
WA	1	Federal-NPS	North Cascades NP	60,265	unknown	60,265	No est. avail	n/a	NPS submission (H. Sakai, pers. comm.)
WA	1	Federal-NPS	Mt. Rainier NP	25,278	unknown	25,278	8,780	0.35	NPS subm - 150 ac degraded by hazard tree removal (H. Sakai, pers. comm.)
WA	1	Tribal- Lummi	Lummi Reservation	520	unknown	520	520	1.00	M. Ostwald, pers. comm., Service
WA	2	Tribal- Quinault	Quinault Reservation	5,951	unknown	5,951	2,925	0.49	BO for North Boundary Area Unit Mgmt Plan and 10-yr
			State total	**1,022,695**	**12,059**	**1,034,754**	**345,521[4]**		
			Grand totals	**2,068,210**	**154,838**	**2,223,048**	**820,768[4]**		

[1] *Data compiled by the Service, Region 6, Portland and Lacey, Washington Field Office. Documents are available in administrative record.*
[2] *Likely to be occupied acreages were calculated by the Service based on occupancy rate data from surveys of potential murrelet habitat in each Conservation Zone.*
[3] *Occupancy index was calculated by dividing the acres of likely to be occupied habitat by the amount of suitable habitat.*
[4] *Totals represent a sum of available data but likely are not an accurate estimate of occupied habitat due to extensive missing data.*

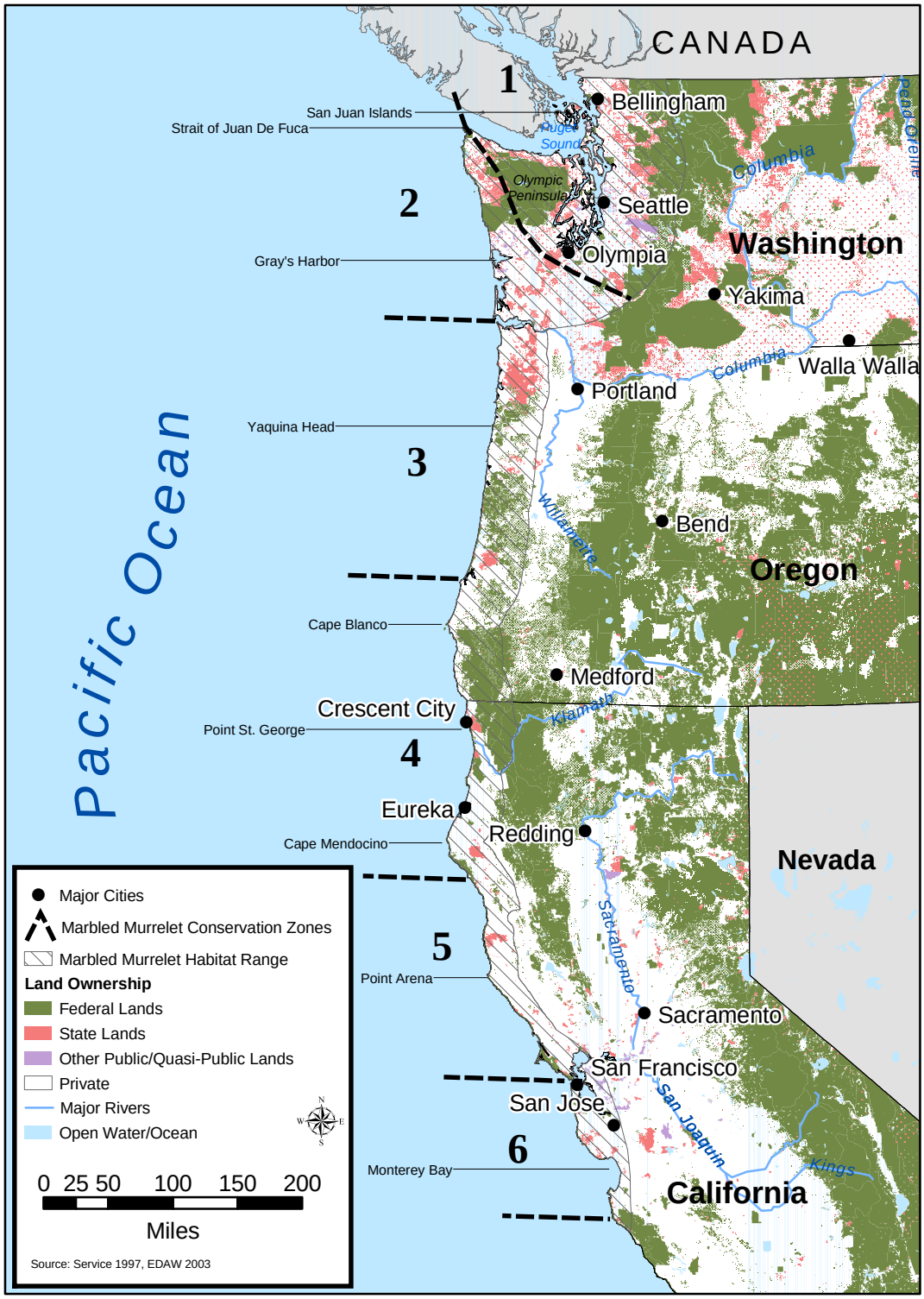
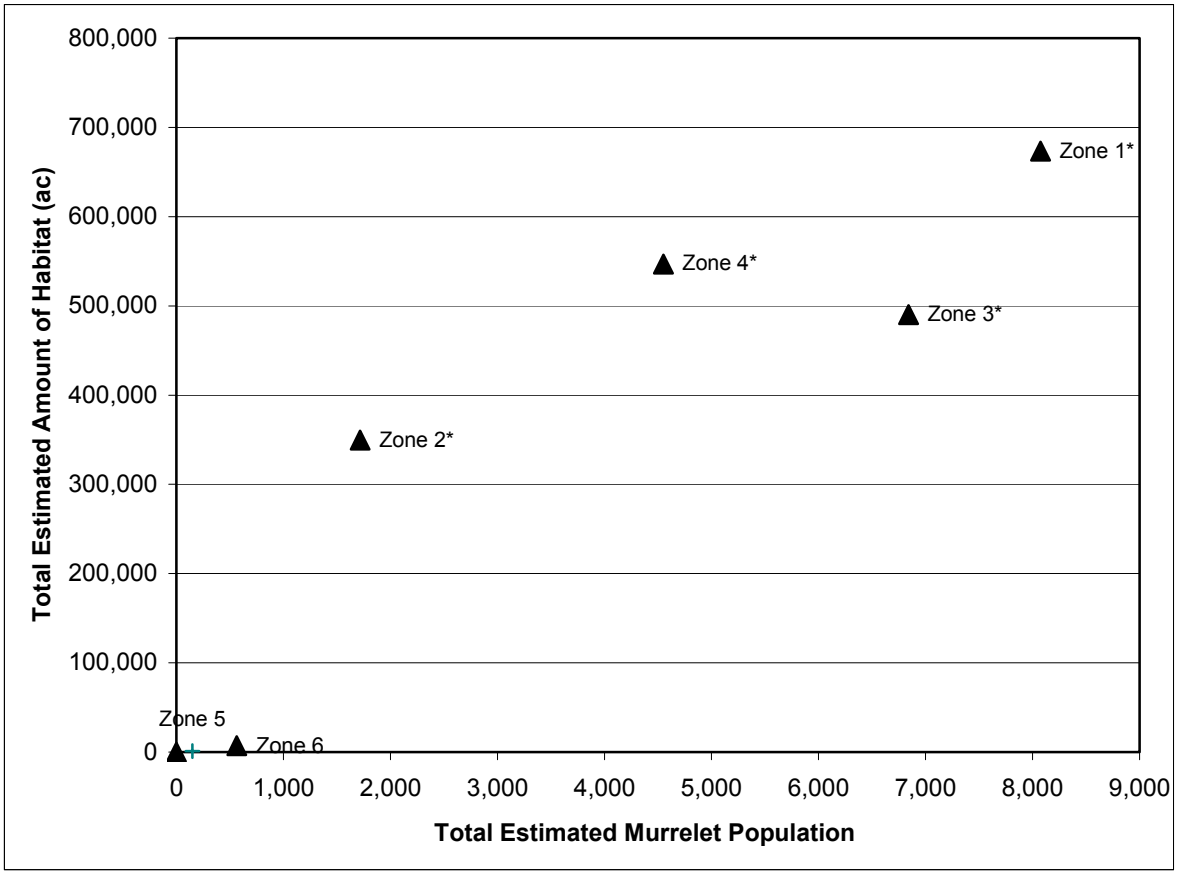


Figure 4.1-1. Marbled Murrelet Range, Conservation Zone Boundaries, and Land Ownership.



Source: acreage estimates from Service, population estimates (Huff et al. 2003).
**For acreages reported as belonging to both Zones 1 and 2 or between Zones 3 and 4, acreage estimates were split evenly between the 2 zones for calculations of total available habitat.*

Figure 4.1-2 Comparisons of estimates of suitable marbled murrelet habitat summarized from local land managers to estimates of murrelet population size by Conservation Zone.

1998, and 1999 and covered lands on the Six Rivers and Klamath National Forests and the Hoopa Indian Reservation lands, north of the boundary between the Klamath Mountain and Northern California Coast Range sections, and south of Frying Pan Ridge near Happy Camp, California (Hunter et al. 1998; Schmidt et al. 2000). Murrelets were not detected during these surveys, and the Service agreed that additional surveys were no longer necessary. Prior to the completion of these surveys, the Six Rivers National Forest reported 576,742 acres (233,400 ha) of suitable murrelet habitat and the Klamath National Forest reported 138,628 acres (56,101 ha). Survey results revised the estimate of suitable murrelet habitat to 217,553 acres (88,041 ha) and 85,976 acres (34,793 ha) for these 2 national forests, respectively (see Table 4.1-1).

- Southwest Oregon – Analysis of murrelet survey results from 1988 through 2001 on the Siskiyou and Rogue River National Forests and the Medford District of the Bureau of Land Management (BLM) demonstrated that murrelets are not likely to occur east of the western hemlock/tanoak vegetation zone in southwest Oregon (Alegria et al. 2002). Prior to the completion of surveys in southern Oregon, the Siskiyou and Rogue River National Forests and the Medford BLM reported 271,046 acres (109,689 ha) of suitable murrelet habitat. Survey results revised this estimate to 100,916 acres (40,839 ha) (USFS 2003).

Largely due to funding limitations, there remains an area in northern California between the Oregon border and the Klamath River where the inland distribution has not been redefined through survey results. Thus, the amount of suitable habitat and inland distribution in this border area can only be inferred from the distributions known in areas immediately to the north and south.

Contrary to the range reduction that has occurred in Oregon and California, in Washington, surveys in the western Cascades have resulted in murrelet detections at distances farther inland (up to 70 miles [113 km]) than previously known. Suitable habitat in the western Cascades occurs primarily on United States Forest Service (USFS) and State lands. The acres listed in Table 4.1-1 for the Gifford Pinchot National Forest (GPNF) do not include habitat beyond 50 miles (80 km) inland.

4.1.3 Ownership Pattern and Distribution of Habitat

Federal lands account for the majority of suitable murrelet habitat in the 3-state area (Table 4.1-1). Approximately 2 million (93%) of the 2.2 million total acres (809,375 of the total 890,312 ha) are located on Federal lands (Table 4.1-1). Other public lands (e.g., State and County) and private ownership account for approximately 8% of the total area, while Tribal lands account for less than 1% of the total area of suitable habitat. The ownership pattern within each Conservation Zone (Figure 4.1-1) plays a significant role in the overall habitat distribution due to the different land management practices on Federal, State, and private lands.

Designated critical habitat (FR 61:102 May 24, 1996, pp 26255-26320) for the marbled murrelet includes 3,887,000 acres (1,573,011 ha) in 32 Critical Habitat Units (CHUs) (Figure 4.1-3). It is important to note that these CHUs include some non-suitable habitat due to boundary delineations. In California, some Late Successional Reserve CHUs are located in areas

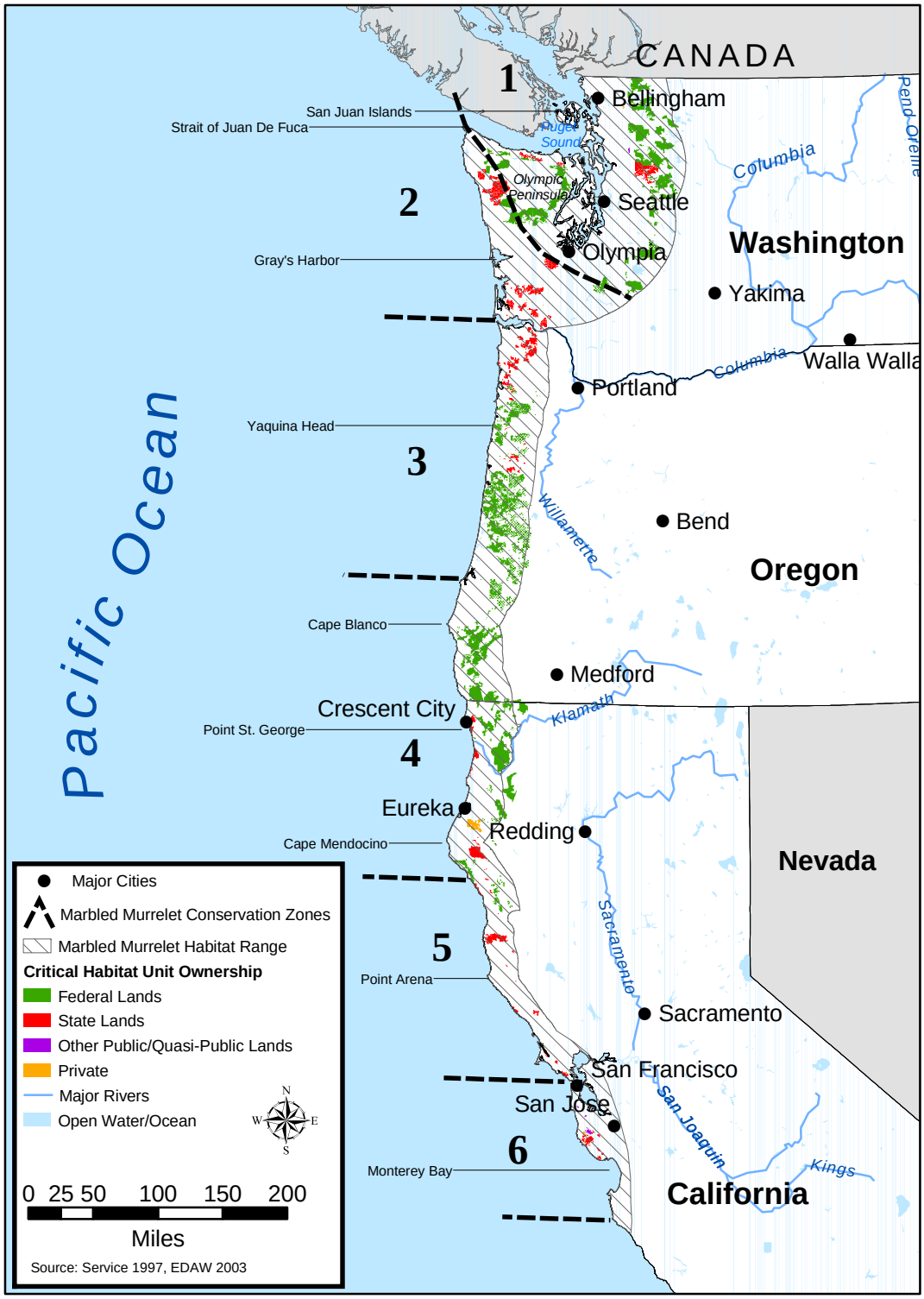


Figure 4.1-3. Critical Habitat Units for the Marbled Murrelet.

currently thought to be outside the inland range of the species. USFS lands in Late Successional Reserves (LSRs), which were established by the NWFP (USFS and BLM 1994) account for 78% of this area; State lands comprise about 21% of the CHU area, while private, County, and City lands account for 1.2, 0.2, and 0.003%, respectively (Service 2002). The sections below describe the ownership patterns and suitable habitat distribution in each of the 6 Conservation Zones.

- Puget Sound (Conservation Zone 1) – The majority of suitable murrelet habitat in Zone 1 occurs in northwest Washington and is found on USFS and National Park Service lands, and to a lesser extent on State lands. Habitat along the eastern and southern shores of Puget Sound has been replaced by urban development, and the remaining suitable habitat is a considerable distance from the marine environment (Service 1997). The largest population of murrelets in the 3-state area is found off the coast of Zone 1 (Huff et al. 2003).

- Western Washington Coast Range (Conservation Zone 2) - Suitable murrelet habitat north of Grays Harbor in Zone 2 occurs largely on State, USFS, National Park Service, and Tribal lands, and to a lesser extent on private lands. Alternatively, the majority of habitat in the southern portion of Zone 2 occurs primarily on State lands, with a small amount on privately owned lands. Some of the privately owned lands have recently been purchased and put into the Federal refuge system. The fourth largest population of murrelets is found off the coast of Zone 2 (Huff et al. 2003).

- Oregon Coast Range (Conservation Zone 3) - The majority of high quality suitable murrelet habitat in Zone 3 occurs along the central Oregon coast on USFS and BLM lands. Alternatively, northwest Oregon contains less suitable habitat that is generally lower in quality and found in small scattered patches. What is left of suitable habitat is largely found on State lands and has been subject to a long history of timber harvest and wildfire. In western Oregon, private forest industry lands consist of more younger age classes than Federal and State lands; 90% of the stands on private lands are 60 years of age or younger (Adams et al. 2001). On non-Federal lands in western Oregon, only about 5% of the stands have an average stand diameter of 21 inches or greater. In the Oregon Coast Range, 64% of the land is privately owned, while 12% is State owned and 24% is managed by Federal agencies (Wimberly et al. 2000). Approximately 8% of murrelet habitat on Federal lands in this area is not in Late Successional Reserves (letter from Doug Heiken, Oregon Natural Resources Council). The second largest population of murrelets in the 3-

state area is found off the coast of Zone 3, primarily on the central Oregon Coast where most of the suitable habitat remains (Huff et al. 2003).

- Siskiyou Coast Range (Conservation Zone 4) - Most suitable murrelet habitat in Zone 4 occurs in proximity to the coast. Habitat in southwest Oregon is, in general, high quality, occurring largely on USFS lands and to a lesser extent on BLM lands. Northern California contains several large parks and reserves, and to a lesser extent some privately owned lands, that are known to contain murrelets. This suitable habitat is surrounded by large areas of unsuitable habitat. Though large amounts of habitat occur on BLM and USFS lands farther inland, they contain few murrelets, likely because of the distance from marine waters. The third largest population of murrelets in the 3-state area is found off the coast of Zone 4 (Huff et al. 2003).

- Mendocino (Conservation Zone 5) - Suitable murrelet habitat in Zone 5 is very sparse and largely limited to State, County, and National Park lands. Most of the habitat that occurred in this zone historically has been harvested, and that which remains is lower quality, found in scattered small patches in parks and on private lands. Very few murrelets occur off the coast of Zone 5, likely due to the small amount and low quality of habitat present inland (Huff et al. 2003).

- Santa Cruz Mountains (Conservation Zone 6) - Suitable murrelet habitat in Zone 6 is restricted to small pockets of State and County park lands and private lands in San Mateo and Santa Cruz Counties. Like Zone 5, most of the habitat that occurred in this zone historically has been harvested, and that which remains is lower quality, found in smaller patches, and highly impacted by human recreation use. A small population of murrelets occurs off the coast of Zone 6 (Peery et al. in prep.).

4.1.4 Estimated Amount of Likely Occupied Habitat

Some proportion of identified suitable habitat for marbled murrelets may in fact not be occupied due the absence of nesting structure or due to its spatial distribution. Because remote sensing methods likely overestimate the amount of habitat actually used for nesting, the Service defined "occupied habitat" as that portion of suitable habitat known to be occupied by nesting murrelets, per the criteria in the Pacific Seabird Group survey protocol (Evans Mack et al. 2003), or expected to be occupied based on survey history in the area and the application of an occupancy index to unsurveyed areas.

For example, although approximately 100,000 (40,469 ha) acres of late-seral forests occur on the Siskiyou and Rogue River National Forests and the Medford BLM, survey results in the area closest to the coast suggest that 26% of the suitable habitat is actually occupied by murrelets (Table 4.1-1). This 26% occupancy index was then applied to unsurveyed areas to generate an estimate of likely occupied habitat (L. Webb, pers. comm.) for this region. Where published data were lacking, the Service solicited professional judgments from local biologists and considers these simple estimates to be the best available information (Table 4.1-1). Estimates of likely occupied habitat for each state are summarized below.

- California: The Service estimates that there are approximately 66,626 acres (26,963 ha) of likely occupied habitat in California, or only 16% of the reported suitable habitat. Much of what is considered suitable habitat on USFS lands is likely not occupied by murrelets, and is the result of our incomplete understanding of the inland distribution of the species. The quality of much of this habitat is variable, with good quality habitat in unmanaged redwood forest and lower quality habitat in the managed Douglas-fir forest or the residual redwood forest.

- Oregon: The Service estimates that there are approximately 408,621 acres (165,364 ha) of likely occupied habitat in Oregon, or 51% of the reported suitable habitat. As there are many areas with suitable habitat for which no occupancy index could be estimated, this estimate is clearly a minimum value. Much of this habitat varies in quality, with good quality habitat generally found in the unmanaged western hemlock/Douglas-fir (*Pseudotsuga menziesii*) stands and lower quality habitat in the managed stands of the same type. Where there are comparable estimates, the amounts of habitat considered likely to be occupied are considerably lower than the estimated amount of suitable habitat. This could be the result of our inability to accurately classify murrelet habitat at landscape scales, our incomplete understanding of the inland distribution of the species, or murrelets' preference for using a smaller subset of available habitat (possibly higher quality habitat).

- Washington: Using the data available for this 5-Year Status Review, there are a minimum of 345,521 acres (139,828 ha) of likely occupied habitat in Washington. As there are many areas with suitable habitat for which no occupancy index could be estimated, this estimate is clearly a minimum value.

As the estimates of likely occupied habitat are derived from numerous sources and inconsistent survey coverage, the area currently occupied by murrelets in the 3 states cannot be ascertained with any reasonable

confidence. Again, the completion of the habitat mapping project under the Effectiveness Monitoring Plan for the NWFP (Madsen et al. 1999) and additional research on the actual occupancy of suitable habitat are necessary to more fully understand the distribution of nesting murrelets, particularly at a landscape level. Additional research is needed on each of the following topics using landscape level indicators of murrelet habitat, as identified by Madsen et al. (1999):

- Acres of suitable marbled murrelet nesting habitat
- Acres of potential or recruitment nesting habitat
- Habitat patch size and spatial distribution
- Interpatch distances or proximity of habitat patches
- Distribution of habitat in relation to the marine environment.

4.2 Inland Detectability

A primary difficulty in determining occupied habitat at inland sites is the ability to detect marbled murrelets at potential breeding sites in the forest ecosystem. Detecting murrelets at inland sites is extremely difficult because of poor visibility conditions for ground observers during the dawn activity period, poor viewing conditions in closed canopy forests, and the species' small size, rapid flight speed, and cryptic plumage (Hamer et al. 1995). This section describes the various survey methods used to detect potential breeding sites, the revisions to survey protocols since the species was listed, and the error rates associated with the different types of surveys.

4.2.1 Survey Methods and Ability to Detect Breeding Sites

A standardized inland survey protocol for the marbled murrelet was first developed in 1990 (Paton et al. 1990); the Pacific Seabird Group (PSG) took the lead role in drafting and revising the protocol beginning in 1992. PSG is a professional scientific organization that has taken a lead role in coordinating and promoting research on murrelets. With this survey protocol, murrelets could be detected at potential breeding sites by both auditory and visual observations. Over the last decade, this protocol has been updated several times, with the latest version published in 2003 (Paton et al. 1990; Ralph and Nelson 1992; Ralph et al. 1993, 1994; Evans Mack et al. 2000, 2003). In British Columbia, a similar effort to create a standardized inland survey protocol was completed in 2001 (RIC 2001).

The 2003 inland survey protocol recommends 2 basic survey types: intensive auditory and visual surveys and radar surveys (Evans Mack et al. 2003). Intensive auditory and visual surveys use a single ground-based observer positioned at a single survey station located in potential breeding habitat during a 2-hour survey period near dawn. These surveys are designed to determine presence or probable absence at a specific site, determine if a site is occupied, and document activity levels. Intensive surveys have also been used to locate nest sites; examine habitat relationships by comparing habitat at sites determined to be occupied and unoccupied, or between high and low detection sites; and examine seasonal and environmental factors that affect murrelet activity patterns. A summary of the protocol methods for auditory and visual surveys, and revisions that have occurred between 1992 and 2003, is provided in Table 4.2-1.

Radar surveys use a stationary marine radar system, modified for use in the terrestrial environment, to detect and track murrelets in flight. The major uses of radar surveys include:

- Detecting the presence of murrelets in an areas;
- Locating "hotspots" of activity over an area;
- Providing an index of abundance for a drainage or stand;
- Determining daily activity patterns of murrelets; and
- Population monitoring (Hamer et al. 1995, Cooper and Blaha 2002).

There are currently no survey protocol guidelines for radar; however, survey recommendations have been published by Cooper and Hamer (2003).

Radar studies indicate that audio-visual observers detect an average of 10 to 23% of all marbled murrelets within 656 feet (200 m) during intensive murrelet surveys, although the percent detected varies widely among sites and among days within a station (Cooper and Blaha 2002). The greater efficiency of radar surveys over standard surveys is due to radar's ability to detect murrelets regardless of light levels and over a greater portion of the landscape. Radar has the advantages of detecting silent birds that are likely to be nesting, can examine a landscape out to a 0.75-mile (1.2-km) radius, which is 40 times the area of a typical ground observer, can detect birds through darkness and fog, and does not rely on the bird vocalization for detection (Hamer et al. 1995). Radar data indicate that 25% of murrelet movements at inland sites occurred before the standardized Inland Forest Survey Protocol survey start time (Cooper and Blaha 2002).

Table 4.2-1. Summary of protocol definitions, recommendations, and modifications, 1990-2003.

Year	Protocol Definitions, Recommendations, and Modifications
1990[1]	▪ Defined "detection" as the sighting or hearing of 1 or more murrelets acting in a similar manner. Recommended 2 survey types: o General *Objective* – Determine the distribution of murrelets over a broad area and identify areas with concentrations of birds during the breeding season. 10 minutes at each survey station along a transect line. *Methods* – 8-10 stations were visited during a 120-minute census period beginning 45 minutes before official sunrise. Stations on road transects were spaced 0.5 to 1 km apart while stations on trails were spaced 250 m apart. *Probable absence in survey year* – 4 surveys/transect over the breeding season. o Intensive: *Objectives* – (1) determine activity levels at a specific site; (2) compare activity levels between sites and monitor changes in activity levels at specific sites between seasons or years; (3) detect murrelets at sites with low densities or determine the probable absence of birds; and (4) determine if a site appeared to be used for nesting. *Methods* - Select 1 or more survey station with a clear view of the sky and place a single surveyor at that station for 120-minute period. *Probable absence* – 3 survey visits during the peak activity period. ▪ Limitations: o Did not specify the number of years of surveys required to determine probable absence. o Did not identify which specific flight behaviors would be indicative of nesting. o Definition of an occupied site and which flight behaviors were likely determinants of probable nesting were not discussed in detail. o Range of potential habitat to survey was not defined, except to mention surveying old-growth and mature stands within certain distances of marine waters.
1992[2]	▪ Classified sites into 3 categories: probable absence, presence, and occupied: o *Occupied*: Defined as sites with evidence of nesting (egg shell fragments, downy chick, etc.) or observation of at least 1 of the behaviors indicative of nesting: flights below, through, into, or out of the forest canopy within potential habitat, and birds observed landing, departing, or calling from tree. o *Presence*: All other flight behaviors such as circling and flights above the top of the forest canopy. Presence sites were where murrelets had been detected, but no subcanopy or "occupied" detections had been observed. o *Probable absence*: Sites where no murrelets were detected after the required number of surveys had been completed. ▪ Defined suitable habitat as mature and old-growth forest. ▪ Recommended 4 surveys/yr and 2 consecutive years to determined presence or absence. ▪ Defined 30 acres as the maximum survey station size. ▪ Limitation: Did not define the number of surveys needed to determine occupancy at sites where presence was detected.
1993[3]	▪ Modified habitat definition to include potential habitat, which was defined as younger conifer forests with deformations or structures suitable for nesting. ▪ Defined peak inland activity period as the last 3 weeks in July and recommended at least 1 survey during this time. ▪ Recommended surveys stations <30 acres in steep or brushy terrain. ▪ Limitation: Did not define the number of surveys needed to determine occupancy at sites where presence was detected.
1994[4]	▪ Similar to 1993 protocol. ▪ Recommended number of survey visits to sites did not change. Suggested increased survey effort in a given year or an increase in the number of years of surveys to detect occupancy for sites with a status of presence.

Table 4.2-1. Summary of protocol definitions, recommendations, and modifications, 1990-2003.

Year	Protocol Definitions, Recommendations, and Modifications
	▪ Limitation: Did not define the number of surveys needed to determine occupancy at sites where presence was detected.
2000[5]	▪ Recommended 4 visits in each of 2 consecutive years to establish presence and 10 visits over 2 years to detect occupancy. ▪ Redefined potential habitat to include forested areas with a residual tree component, small patches of residual trees, or 1 or more potential nest platforms. ▪ Identified parameters to use when selecting stands to conduct murrelet surveys; recommended a walk-through of the stand to look for the presence of potential nest platforms. ▪ Emphasized that the failure to identify potential habitat to survey could result in loss of occupied sites. ▪ Recommended the addition of supplementary survey stations with improved visibility that are >50 m from the edge of the stand, or are adjacent to a stream where hearing is limited but visibility is good. ▪ Recommended tandem surveys, where 1 observer is placed adjacent to a stream with good visibility but limited hearing and another observer is placed at a survey station with quiet conditions. ▪ Recommended a 30-minute survey extension in rainy, cloudy, or foggy conditions. ▪ Suggested repeating the survey if environmental conditions such as rain, wind, or other factors limited the observer's ability to detect murrelets for more than 12 cumulative minutes of the 2-hr survey period. ▪ Limitation: Did not provide guidance on how to distribute the 10 visits over the 2-year period.
2003[6]	▪ Similar to 2002 protocol. ▪ Recommended a 2-stage sampling approach if a 95% confidence of survey outcome was desired: o A minimum of 5 survey visits in each of 2 years to detect presence. o 9 survey visits in each of 2 years to determine occupancy of a site once presence had been detected.

[1] *Paton et al. 1990*
[2] *Ralph and Nelson 1992*
[3] *Ralph et al. 1993*
[4] *Ralph et al. 1994*
[5] *Evans Mack et al. 2000*
[6] *Evans Mack et al. 2003*

Radar surveys indicate that approximately 14% of the murrelets detected on intensive surveys are birds passing over the stand of interest on their way to another area (Cooper and Blaha 2002). Cooper and Blaha (2002) detected murrelets 100% of the days with radar in 1999; thus, the mean number of days required to determine "presence" with radar was 1.0. The number of days required to determine "presence" with the audio-visual technique was significantly higher, ranging from 1 to 5 plus days, with a mean >2.3 days. Radar has been found to detect 2 to 10 times the number of murrelets compared to standard surveys, and also provides much better estimates of the number of birds using an area compared to the ground survey protocol that provides no estimate of bird density. Audio-visual surveys do not allow estimates of absolute bird numbers (Paton 1995, Ralph et al. 1994). Several detections could arise from a bird or flock repeatedly circling the observer, and the numerical relationship between detection and number of birds is not known.

Because radar-based counts have low among-day variability, radar sampling also may be well suited for long-term population monitoring. Cooper et al. (2001) estimated that given the variation levels in abundance recorded using radar, it would take 4 to 5 years of surveys (with 1 to 2 visits per site per year)

to detect a 10% drop in population with >90% power. The only other option that has been explored for long-term monitoring has been at-sea surveys. This technique can have high variation among counts (Becker et al. 1997); more importantly, it is difficult to associate marine abundance with use of an inland area on a small scale because of this species' great foraging range (Whitworth et al. 2000) and variation in prey habitat availability (Nelson 1997).

The major limitations of the radar technique are: (1) it cannot determine occupancy (and sometimes presence) because birds flying near or within the canopy are shielded from the radar and missed; (2) it cannot be used at all sites because of topographic and physiographic constraints and would be difficult to use at inland watersheds that are roadless and have little access; (3) species identification errors are possible; (4) repeated entries by some birds cause overestimates; and (5) X-band radar cannot be used during rain (but can be used during drizzle and fog). However, there are methods that have the potential to minimize many of these limitations. Perhaps the greatest limitation is that radar cannot be used at all locations. Radar cannot see behind hills, and the echoes of hillsides, trees, and other objects can obscure large portions of the radar screen, resulting in missed murrelet detections. Cooper and Blaha (2002) found that using a lift-assisted radar greatly improved the ability to conduct surveys at various sites on the Olympic Peninsula. It was possible to use radar at 56% of randomly chosen sites; without the lift-assistance, observations could only be conducted at 15% of all sites. Raphael et al. (2002a) suggest that the analysis of relationships between habitat conditions and the number of murrelets can be extended beyond the subset of drainages that have suitable topography for radar surveys. If investigators can identify a subset of drainages that are suitable for radar-based counts within a geographic area, that subset can be used to quantify relationships between habitat conditions and murrelet abundance. Models derived from these subsets could then be applied to drainages that are otherwise unsuitable for radar work. However, caution should be used in this approach, since drainages with substantial differences in topography or access may not necessarily be comparable because of differences in habitat quality or other factors. Species identification errors can be limited by conducting audio-visual surveys in conjunction with radar surveys, which allows for a visual verification of subject detections (those at the lower end of murrelet flight speeds), and by limiting surveys in times of high winds. To limit the possibility of counting repeat entries, Burger (2001) restricted his analysis to pre-sunrise counts. Sunrise is a convenient, seasonally adjusted, and biologically meaningful event that appears to separate the first and second pulse of incoming birds.

4.2.2 Survey Effectiveness

There are a variety of factors that can affect the ability of observers to detect murrelets at inland sites. These factors may include differences between survey sites, daily variation in detection rates, effects of season and weather, proportion of sky visible at survey stations, year, distance of the site from marine foraging areas, oceanographic factors, and habitat quality (O'Donnell 1995, Nelson 1989, Baldwin 2002). Occupied sites with low detection rates overall, higher canopy closures, locations far from marine foraging areas, and with lower habitat quality will likely have lower probabilities of detecting occupied behaviors.

There is strong evidence that detection probabilities for both presence and occupancy behaviors vary among sites, but the magnitude of these differences is unknown (Nations and Manly 2002). In a nested analysis of variance, Hamer (1997) found that variability in total detection (occupied and unoccupied behaviors combined) rates was consistently highest between survey sites, followed in order by variation between survey days, physiographic provinces, and years. These differences in variation held true for visual-only, auditory-only, and both visual and auditory detections combined. Variability in detection rates was highest for auditory detections and lowest for detections of birds both seen and heard. Sites with lower detection rates (≤10 detections/survey morning) had significantly lower detection probabilities compared to sites with high detection rates (Hamer 1997). Study results on the various factors affecting detection rate are summarized below.

- Season – In a study on the effect of season on detection probability, Baldwin (2001a) classified each survey visit as a binary variable with values defined as "occupied behaviors observed" and "occupied behaviors not observed." The proportion of visits with one or more occupancy detections were low through the breeding season but higher during a 1-week period in the middle of July, although the magnitude of this difference was considered small (Baldwin 2001a). Inland detection rates are known to peak in the last half of July throughout the geographic range, coinciding with the main fledgling period (Hamer and Cummins 1991, Rodway et al. 1991, Naslund 1993, O'Donnell et al. 1995, Brown et al. 1999). In addition, post-sunrise detections and length of detection periods have also been shown to peak in July (Brown et al. 1999). Detection rates were lowest in April during the pre-alternate molt period, and also low in September and October, corresponding to the adult pre-basic molt (O'Donnell et al. 1995, Brown et al. 1999, Carter and Stein 1995). Occupied behaviors were more likely to occur before sunrise than

after, and group size became larger after sunrise (Naslund and O'Donnell 1995). Jodice and Collopy (2000) found that temporal variability in murrelet detection rates was high, with variation in occupied detections higher than overall daily detections. Intra-annual variation in counts of daily detections within stations was not strongly or consistently related to date or weather (Jodice and Collopy 2000).

In an analysis of survey data collected in Oregon using resampling techniques and statistically generated detection data, Jodice et al. (2001) assessed the feasibility of using audio-visual survey data as a monitoring tool to assess the spatial and temporal trends in murrelet detections. They found that the large temporal variability in detection data had a great effect on the reliability of the mean and standard deviation estimates calculated from 12 survey strategies they tested. They concluded that the effectiveness at estimating multi-year trends in detection data was also poor, and that audio-visual surveys might only be reliably used to estimate annual declines in murrelet detections that were 50% or more per year (Jodice et al. 2001).

- Weather Conditions – Weather conditions also affect the detectability of murrelets at inland sites. Naslund and O'Donnell (1995) reported that murrelet detections tended to begin later, last longer, and reach higher peak levels on cloudy or foggy mornings. However, Jodice and Collopy (2000) urged caution in interpretation of these data by pointing out that "most of the data used in these analyses were not collected from studies specifically designed to address this question." Rodway et al. (1991) reported higher detections and longer activity periods on cloudy mornings versus clear mornings. In Oregon, however, Jodice and Collopy (2000) used multivariate analyses to evaluate the influence of weather on the daily detection patterns of murrelets. They concluded that weather and date variables explained little of the variability in daily detection activity. In a study of murrelet detection patterns in British Columbia, Rodway et al. (1993) found higher murrelet detection rates and longer periods of detection on cloudy days in comparison to clear days. However, this relationship was significant for only 1 of 2 study sites.

  In British Columbia, Burger (2001) used pre-and post-sunrise counts of marbled murrelets detected by radar to evaluate the influence of weather on murrelet detections. Pre-sunrise counts averaged "1.4 times higher on cloudy or drizzly/foggy days than on clear days," but no significant differences in detection rates were apparent in comparisons between cloudy days and drizzly/foggy days. Burger's (2001) analyses of pre-

sunrise counts revealed that the higher counts occurred on cloudy mornings (versus clear) and that there was no difference in detection rates comparing cloudy with drizzly/foggy mornings. For the post-sunrise period, Burger reported "high post-sunrise counts, indicating repeated nest visits were equally likely on clear, cloudy, or drizzly/foggy mornings." Burger (2001) could not test the influence of heavy drizzle or rain on murrelet detection rates, as the radar equipment would not function properly under these circumstances.

Another radar study examined the daily, monthly, and annual patterns of marbled murrelet detections in 12 large river valleys on the Olympic Peninsula (Cooper et al. 2001). Counts of murrelets flying landward were significantly greater on days with high cloud cover (>50%). No differences in counts were found, however, between days when the ceiling height was ≤656 feet (200 m) above the ground level and those when it was >656 feet (200 m).

Horton and Harrison (2001) examined the correlation between daily murrelet detections and a number of summarized weather variables. Like Jodice and Collopy (2000), they found that these measures were not well-correlated. However, they also employed another analytical approach in which they compared the weather conditions existing at the time of each murrelet detection with the availability of those conditions across the study. This method detected significantly ($P<0.01$) greater proportions of murrelet detections during the following weather conditions: low ceiling (below 2 canopies), high cloud cover (100%), no rain, and any level of fog. These findings are consistent with long-standing observations of field biologists – that murrelets are often detected more frequently during cloudy, foggy weather; and that rain can compromise the observers' abilities. They note that while these patterns were measurable, they were subtle. Only small differences between observed and expected numbers of detections were noted (200-300 out of >6,800).

- Canopy Closure - Detection rates also vary with canopy closure. O'Donnel (1995) showed that survey stations with a larger percent of open sky could be expected to have a greater percent of murrelet observations. In a comparison of survey stations located at streambeds versus adjacent forest stations, numbers of occupied behaviors observed were 6 times greater at streambed stations (Rodway and Regehr 2000). Although the larger size of the openings at streambed locations accounted for much of the difference, detection rates were still lower at forest stations even after accounting for differences in opening size. The

authors concluded that streambeds were being used as flight corridors, thus inflating the number of detections recorded compared to adjacent forest stations.

- Year - Annual variation in detection rates and detectability has also been demonstrated (Hamer 1997, Baldwin 2001b, Smith and Harke 2001, Jodice and Collopy 2000, Horton 2003). An analysis by Baldwin (2001b) using survey data collected in California, Oregon, and Washington for pairs of survey years from 1991 to 1998 showed that some sites could be occupied in 1 year but not the other. The estimated proportion of sites observed to change status from each pair of years studied varied from 18 to 65% per year, with a weighted average of 39%. Therefore, in some years, the probability of detecting occupied behaviors may be zero at some sites, and a 1-year survey would risk misclassifying these sites as unoccupied. In a study by Jodice and Collopy (2000), there was also substantial inter-annual variability in means of daily detections within sites between years. Horton (2003) examined inland detection levels from a set of 4,856 murrelet surveys conducted at 642 sites on the western Olympic Peninsula between 1994 and 2001. Multiple regression analyses suggested that annual murrelet detections (total and subcanopy detections per survey) were associated with year and that the data suggested a nearly 10% annual decline in detections over the study area.

- Inland Distance - Since detection rates are known to decline with increasing inland distance, survey stations located far inland are likely to have lower detections. Suggestions for improving the reliability of surveys in low abundance areas were summarized by Hunter and LeValley (1996).

- Marine Factors - Marine factors could also have an influence on inland detection levels. Horton (2003) examined inland detection levels from a set of 4,856 murrelet surveys conducted at 642 sites on the western Olympic Peninsula between 1994 and 2001. Multiple regression analyses suggested that annual murrelet detections (total and subcanopy detections per survey) were associated with winter sea-surface temperature (and year). He suggested that estimates of murrelet inland detection levels for monitoring purposes should consider the influences of oceanographic phenomena (Horton 2003).

4.2.3 Error in Classifying Occupied Sites

Stauffer et al. (2001) described a method to incorporate the uncertainty of detection in presence/absence surveys so that options for power (the

probability of successfully obtaining at least 1 detection with repeated visits to a sampling unit) and sample size can be factored into the survey design. He showed that probabilities of detection based on the binomial model could be misleadingly high for small numbers of visits, possibly leading to greater likelihood of false negatives (undetected occupancy). He demonstrated that fewer numbers of visits are necessary to obtain a 95% power for a sampling unit as the conditional probability of detection increases. He noted that influences on the probability of detection may include species density, sampling effort, habitat structure, visibility, observer bias, ambient noise, season, and effects of weather.

The primary factor influencing the ability to correctly classify a truly occupied site is the probability of detecting presence or occupied behaviors (see Table 4.2-1 for definitions of presence and occupied behaviors) on a single survey visit (Max et al. 1995). This probability incorporates all the various factors that affect the ability of observers to detect murrelets. Observing the particular behaviors required to classify a site as occupied is difficult. It is easy to miss observing the required behavior in a single visit to a site, even by highly trained biologists (Max et al. 1995), resulting in a false negative (undetected presence). When the required behavior is observed, biologists are confident of observing it correctly. So misclassifications are believed to be almost entirely in 1 direction. Although classifying a site as occupied when in fact it is not occupied is possible, this error (false positive) is judged to be unlikely enough to be ignored. Therefore, of critical importance in the survey method is controlling the chances of misclassifying individual sites as unoccupied when they are actually occupied. Because not all visits to an occupied site result in a determination of occupancy, one needs to account for the probability of not detecting occupancy on any single visit. Researchers call this probability "q." The error rate in misclassification is thus controlled through the selection of the maximum number of times (s) a site will be visited. By increasing the number of visits to a site, the misclassification error can be theoretically driven to low levels (Figure 4.2-1). In summary, the value of q, the chance of misclassifying an occupied site on a single visit, is critically important for developing an effective protocol that minimizes the misclassification error (Max et al. 1995).

Since timber harvest operations have largely driven the need to conduct surveys for marbled murrelets, many of the truly occupied sites misclassified as unoccupied from the use of earlier protocol versions have likely been harvested. By analyzing changes in the number of visits required by the inland survey protocol since the species was listed in 1992, we can assess the likely error in classifying truly occupied sites as unoccupied and estimate the percentage of sites potentially lost due to survey error over the last 11 years.

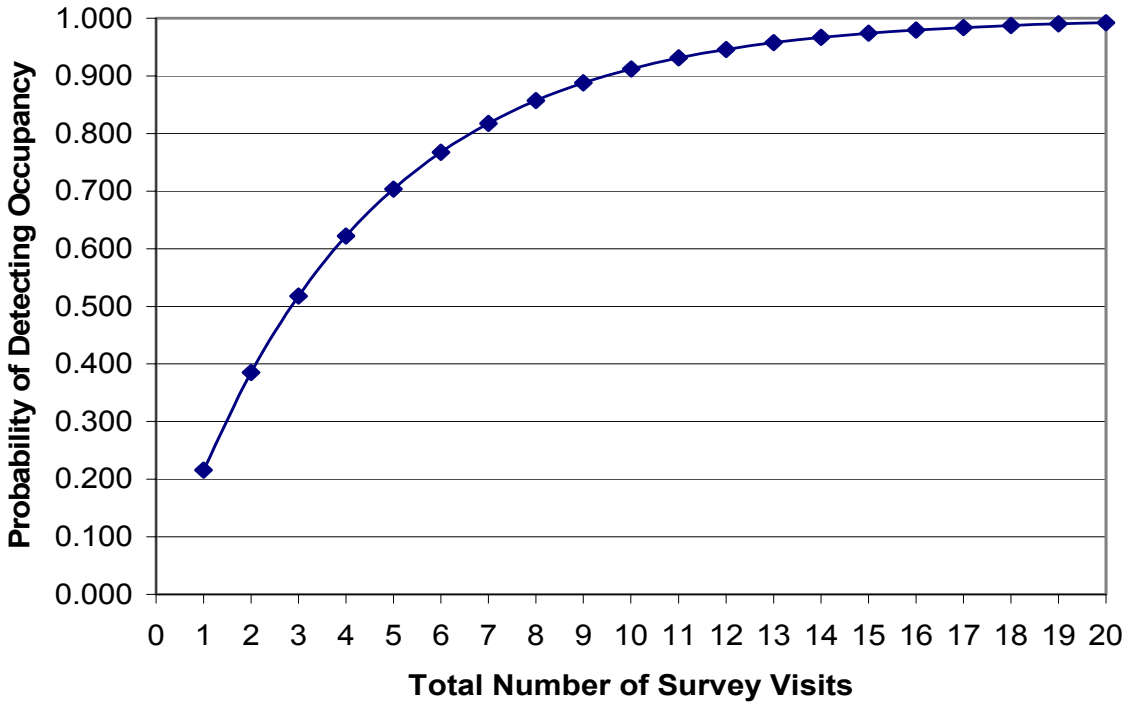


Figure 4.2-1. Relationship between the probability of detecting occupancy and increased survey effort using estimates of q from Baldwin (2002). Annual variation in q was not accounted for in these probabilities.

From 1992 through 1994, the inland survey protocol recommended 4 survey visits in each of 2 consecutive years to determine probable absence of murrelets (Ralph and Nelson 1992, Ralph et al. 1993, Ralph et al. 1994). The number of surveys visits required to detect occupancy was not defined until 1995, when the PSG issued a letter recommending 4 visits in each of 2 consecutive years to detect presence, and 10 surveys to detect occupancy (PSG 1995). Definitions of a site with presence and occupancy are defined in Evans Mack et al. (2003). Because the effect of year-to-year variability in detecting occupancy was unknown, PSG did not recommend how to distribute these surveys over the 2-year period, or the total number of surveys to conduct. For these reasons, it was likely that the majority of sites surveyed from 1992 through 1995 for forest management purposes were based on survey efforts of 4 visits in each of 2 consecutive years to determine probable absence (8 total) and 5 visits in each of 2 years to determine occupancy (10 total).

Using the maximum q values calculated by Baldwin (2002) (Table 4.2-2), the number of truly occupied sites misclassified as unoccupied for the period 1992-1995 can be estimated (Appendix A) using the survey effort guidelines prescribed by PSG.

Table 4.2-2. Estimates of the probability of detecting probable absence, presence, and occupancy during a single visit from sites with a true annual status of occupied.

Parameter	Estimate	Standard Error
Probability of Observing No Detections Absence (qo)	0.4244	0.02577
Probability of Observing Presence Presence (q1)	0.3416	0.01419
Probability of Observing Occupancy Occupancy (q2)	0.2341	0.02222

Source: modified from Baldwin 2002.

Although some proportion of sites likely had greater or lesser numbers of visits than those recommended by PSG, it is likely that, on average, survey efforts approached 4 visits in each of 2 consecutive years to determine probable absence (8 total) and 5 visits in each of 2 years to determine occupancy (10 total) as outlined in the PSG protocol. In addition, probabilities were adjusted for years where the probability of detecting occupancy was zero due to annual variability in occupancy status. Since a weighted average of 39% of occupied sites was found to change status within a 2-year time period (Baldwin 2001b), it was assumed that, on average, that 40% of occupied sites changed status over the 2-year sampling period (20% in Year 1 and 20% in Year 2), and 60% were occupied in both years (Evans Mack et al. 2003).

Results of the analysis indicated that out of every 1,000 sites that were surveyed that were truly occupied, the approximate misclassification rate would approach 15.15% over time. These truly occupied sites would have been misclassified as unoccupied (Appendix A) and possibly harvested, although it is unknown what proportion of these sites had some kind of management activity performed on them. This error rate is 3.3 times higher than the current 2003 PSG survey protocol, which has an approximate error rate of 4.53% over time (Appendix A).

To get an idea of the magnitude of the number of sites surveyed from 1992-1995, the number of survey site years in the 3-state survey database developed for the analysis conducted by Baldwin was calculated (Max 2001). A survey site year is a year of surveys conducted at a single site. There were 3,060 site years in the database from 1992 to 1995. These totals represent only a proportion of the surveys conducted during this period, since much of the data from private and Tribal survey efforts were not collected, and only a proportion of Federal survey data were gathered for the database. We believe that the estimated misclassification rate from 1992 to 1995 could have been lowered by improvements to the survey protocol during this period and from additional protocol improvements implemented from 1996 to 1998 (see Table 4.2-1). An analysis by Baldwin (2002) showed a tendency for an overall decline in q values from 1992 to 1997, although 95%

confidence intervals were large and overlapped for all estimates of q from 1992-1998 (Figure 4.2-2). The number of survey visits required to detect occupancy was not well defined between 1996 and 1997 due too uncertainty regarding the appropriate number of visits to recommend. Because of this confusion, the total number of survey visits to presence sites from 1996 to 1997 probably varied greatly, with some entities conducting the minimum number of survey visits and others the maximum. By 1998, a letter issued by the Service recommended 10 survey visits in each of 2 years to detect occupancy.

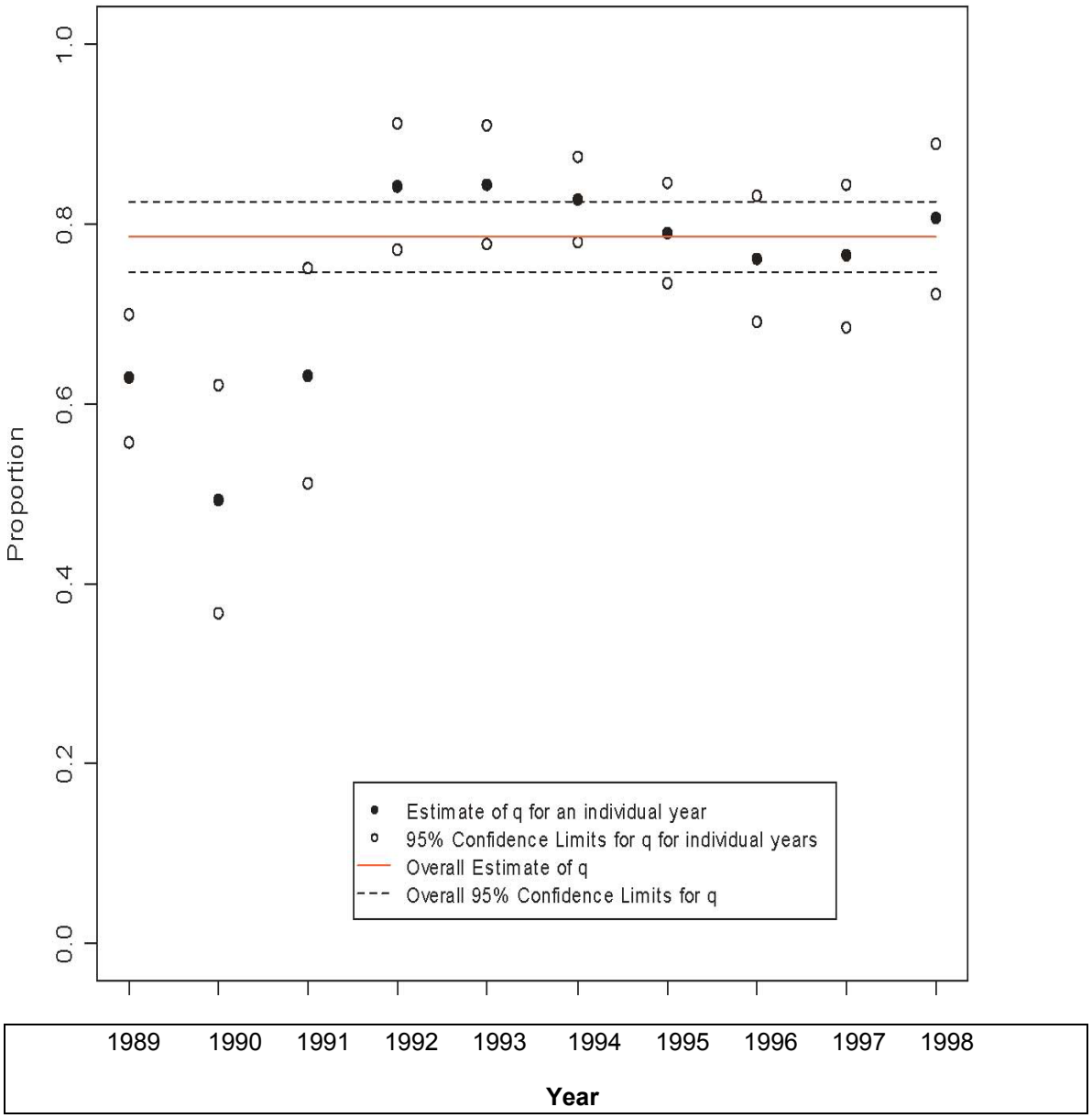


Figure 4.2-2. Estimates of *q* for occupancy along with 95% confidence intervals for each year (Baldwin 2002).

For sites surveyed from 1998 to 2002 using a protocol of 4 survey visits in each year for absence and 10 survey visits in each year for presence sites, the approximate rate of site misclassification approaches 4.21% over time (Appendix B). This is similar to the estimated error rate in the current 2003 PSG protocol.

4.3 Habitat Characteristics

Studies to characterize marbled murrelet habitat have been conducted on a number of scales: at the landscape level, at the nest plot or stand level, and at the tree scale. The following sections describe murrelet habitat characteristics at these different scales.

4.3.1 Habitat Characteristics at the Landscape Level

Habitat characteristics at a landscape scale include climate; elevation; slope; aspect; forest type, age class, structural complexity, and height class; and canopy closure. Studies to determine the characteristics of marbled murrelet nesting habitat at a landscape scale have been conducted using a variety of methods, including predictive models, radiotelemetry, audio-visual surveys, and radar. Results of each of these types of studies are summarized in the following sections.

4.3.1.1 Predictive Models of Habitat Suitability at a Landscape Level

Unlike habitat association studies, which often compare use and non-use areas to describe habitat relationships, studies involving predictive models of habitat suitability create a statistical model using a suite of habitat variables. Predictive models are then typically tested against an independent set of observations to determine how well they perform. On a landscape level, however, the micro-habitat characteristics of the forest typically associated with murrelet nests, such as the density of large branches, epiphyte cover, and mistletoe density, are not available in most forest inventory databases or mapping systems (Lank et al. 2003). Therefore, relating forest structural characteristics required by nesting murrelets to landscape level forest inventory data or GIS data to predict habitat suitability across a landscape has been a priority for some researchers. These efforts have led to the development of habitat suitability models that can be used to predict the presence of murrelets across a large landscape for areas that have not been surveyed and where use is not known. Landscape level mapping of murrelet habitat is important to the management of forest resources within the murrelet's range, as discussed in Section 4.1.

Unlike habitat suitability mapping, habitat capability mapping considers the potential for forest polygons (specific areas of the landscape shown on the map) to develop suitable murrelet habitat in the future (Burger 2002). Once developed and tested, predictive habitat capability models based on forest inventory or other data can also provide estimates of the amount of suitable habitat that has been lost in an area from timber harvest operations and provide information on where best to concentrate habitat restoration activities. Comparing the difference between the amounts of habitat estimated from capability and suitability mapping techniques can be valuable in identifying areas that have experienced significant decreases in murrelet habitat and would therefore be expected to have reduced murrelet populations (Burger 2002).

Over the past 10 years or so, a variety of models have been developed for the marbled murrelet to estimate or predict habitat at the landscape scale. Many of the models at this scale have been developed for murrelet habitat in British Columbia. However, because of differences in vegetation characteristics, biogeoclimatic influences, and landscape conditions between different geographic regions, model development and application may be best restricted for use at regional levels. The findings of the more recent landscape modeling efforts are summarized below.

- Harper et al. (2001) - On Vancouver Island, Harper et al. (2001) developed a model to identify and map suitable murrelet habitat to help a forest products company delineate habitat protection areas (Burger 2002). Field sampling identified the relationship between nest habitat parameters (such as platform density) with mappable parameters. Habitat was graded habitat into 5 suitability categories: (1) very high; (2) high; (3) moderate; (4) low; and (5) coarse filter. The coarse filter was used to eliminate habitat that was too young to provide nesting platforms. Tree height and site productivity was then used to rank the remaining habitat. The last step was to adjust high elevation zones unlikely to provide platforms using a Biogeoclimatic Ecosystem Classification system to down-grade high elevation sites (Burger 2002). He then went on to compare platform densities in non-contributing landbase (Provincial Parks, ungulate winter range, steep inoperable terrain) with land included in the timber harvesting land base. Platform densities were significantly higher in non-contributing forest compared to the timber harvesting landbase, and differences were largely due to elevation differences (Burger 2002).

- McLennan et al. (2000) - McLennan et al. (2000) developed a predictive habitat model to create landscape-level habitat suitability maps for the

marbled murrelet on the Queen Charlotte Islands/Haida Gwaii. They used mappable parameters and ranking criteria for each parameter developed by Burger (2002) specifically for these islands. The objective of the study was to evaluate the relationships between existing forest cover maps and stand structural attributes (measures of murrelet habitat suitability). The study examined the relationship between qualitative and quantitative measures of habitat suitability (stand structure and composition) derived from field sampling and forest cover attributes and broad biogeoclimatic units (similar to a site series) derived from existing forest inventory data. From field data, 4 categories of platform densities were developed as indices of habitat quality: low (0-20/acre [0-50/ha]), medium (21-61/acre [51-150/ha]), high (62-121/acre [151-300/ha]), and very high (>121/acre [>300/ha]). The final model used forest age class, height class, canopy closure, elevation, and slope as variables to predict habitat suitability. A preliminary algorithm to predict habitat suitability for the Queen Charlotte area was developed, and habitat suitability maps were generated with the model. Using the model, each forest polygon was scored for each of these measures and the scores added together to rate polygons into 5 suitability classes. No surveys were conducted as a part of this study; therefore, habitat suitability maps developed by the study did not show how local populations of murrelets utilized the habitats mapped or how these populations were distributed across the landscape (McLennan et al. 2000).

Based on both the field sampling data and forest cover data, potential nest platform density was unaffected by slope and aspect differences, but positively related to stand age class (81-251+ years) and height class (33-210 feet [10-64 m]). As expected, stands with larger, taller trees had increased densities of potential nest platforms. Nest platform densities were highest (65-73/acre [160-180 platforms/ha]) for canopy closures between 28-46% and decreased for higher and lower canopy closures. Tree species showed no association with measures of platform densities and were not used in the model.

The ability of the model to predict habitat suitability (described by platform density classes) was higher for the field sampling data (53%) compared to the existing forest cover data (40%) for all habitat suitability classes except for the lowest class. The authors stated that the higher success rate of the model using the field sampling data was likely due to the error associated with existing forest cover maps. The overall low success rate of both the forest cover type and field sampling models indicated that there was considerable error in predicting nest platform density, even if the forest cover data were completely accurate. Trends in

error of the forest cover data were then used to improve the model. Classification success increased to 61% and 46% for the field sampling data and forest cover data, respectively (McLennan et al. 2000). In most cases, the misclassification was either by 1 rating class or 1 platform density class. The relatively high misclassification rate of the new model emphasized the problem of using forest cover variables to predict murrelet habitat suitability. Forest cover variables by themselves did not accurately account for stand structural attributes. A separate analysis also showed a strong relationship between biogeoclimatic site series and nest platform density (habitat suitability) and stand structure. If these relationships can be further developed, it could provide a tool to assess the quality of the habitats that have been lost, predict the habitat suitability of stands at different levels of succession, and help prioritize habitat to be protected for recovery.

- Bahn and Newsom (2002b) - Bahn and Newsom (2002b) went on to develop a habitat suitability model for all of Clayoquot Sound which evaluated forest polygons from resource inventory maps that contained detailed land cover information focusing on forest cover. Forest variables associated with marbled murrelet nesting habitat were selected for sampling based on information from the literature. They sampled these variables using vegetation plots in randomly selected polygons. Regression analysis was used to relate the sampled habitat variables to the mapped variables. Based on these regressions, 7 mapped variables were selected for inclusion in the habitat suitability model: tree height, tree age, basal area, vertical canopy complexity, canopy closure, distance to ocean, and elevation. The 3 most important variables in their model included tree age, tree height, and basal area, and these were highly inter-correlated.

- Huettmann et al. (2003a) - Huettmann et al. (2003a) report on a study using radio-tagged murrelets to locate a sample of 121 nest sites in Clayoquot Sound and Desolation Sound, British Columbia. In a comparison of landscape level features at nest sites versus random sites, marbled murrelets selected nest sites with steeper slopes, lower elevations, and colder aspects (north), even after taking into account the current distribution of old-growth forests in the study areas. Nest site selection models with these 3 variables had the lowest AIC (Akaike's Information Criterion) scores in the largest number of cases. In a comparison of nests that were deemed successful at the mid-fledgling stage to those that were unsuccessful, they found that nesting success models were not driven by large-scale landscape features analyzed in their study but by smaller scale features that they did not examine (Huettmann

et al. 2003a). The study confirmed the common finding that murrelets overwhelmingly selected old-growth forests as nesting habitat (Nelson 1997).

- Bradley (2002) - Using the same sample of nests studied by Huettmann et al. (2003a) in Desolation Sound, Bradley (2002) applied univariate logistic regression models to study murrelet nesting success. He found that nest success increased with earlier breeding and with increased commuting distance, slope, and elevation (Burger 2002). Further analyses examining these variables did not determine which variable was a dominant influence on the model. Reduced predation at nests that were found farther inland and at higher elevations was suggested as a possible explanation for model results (Burger 2002).

- Hobbs (2003) - Hobbs (2003) assessed the accuracy and reliability of a "Mid Coast" Habitat Model developed by Spencer et al. (2002) for the Mid Coast Forest District in British Columbia. This area is located in the central coast planning area and includes 3 eco-regions: Kitimat Ranges, Northern Pacific Ranges, and the Hecate Lowlands. The results were used to assess the suitability of 46 candidate Wildlife Habitat Areas being proposed to protect marbled murrelets. Data were collected on 11 key habitat attributes for use in modeling and thought to be important in identifying suitable murrelet habitat. These variables included percent of large trees present, percent of emergent trees with platforms, percent canopy cover, vertical canopy complexity, topographic complexity, slope position, slope grade, stand age class, dominant tree species present, percent composition of 6 classes of habitat quality, and a 4-category site habitat quality assessment (superior, good, fair, none). Data were collected for each habitat attribute by flying above each survey area and visually estimating each of the attributes within the polygon. Two models were developed, but the model with the best accuracy included the variables age class, vertical complexity, crown closure, elevation, slope, and tree species present. The predictive accuracy of the model improved notably when "tree species" was included in the model attributes. Tree species was considered a surrogate indicator of site productivity, and not necessarily a selective preference exhibited by marbled murrelets. When tested against the data collected at 80 sample points, the Mid Coast model had an accuracy rate of 64-71%. An even higher level of accuracy (95.5%) was apparent for high quality habitats (good to superior). The model's ability to distinguish different habitats of varying quality was considered a great improvement over previous models that were unable to rank the relative suitability of the habitats identified.

- Meyer et al. (2003) - In a study in central and northern California, Meyer et al. (2003) evaluated whether multi-scale models were more predictive than single-scale models, and tried to determine which scale was most predictive for the murrelet. They used logistic regression models comparing occupied and unoccupied sites for their across-scale comparisons. Classification accuracy of the independent plots was assessed using 4 spatial scales (patch, landscape, subregional, and regional) and 2 time periods (present and previous decade). Many of the variables selected for the best models at each scale were the same as those selected in the multi-scale model. On a regional scale, occupied sites were located in the fog zone, indicating that the redwood zone primarily delineates the boundary of the geographic nesting range in California. On a subregional scale, occupied sites were close to marine areas with high primary productivity. At the landscape scale, occupied sites were in less fragmented old-growth and less isolated from other occupied sites.

  Accuracy in predicting occupancy was high (>80%) for all scales. Classification accuracy was highest by 10% for the model that incorporated all spatial scales. Of the individual scales, the landscape scale was the most accurate because it may have contained the most limiting factors for the murrelet, which were the degree of old-growth forest fragmentation, and isolation. Adding a time-lag factor improved the model's accuracy by 4%, indicating that, for the temporal scale, there was a time lag before birds showed a negative response to fragmentation. Accuracy of the multi-scale model applied to geographic subsections of the nesting range was high, and varied from 86-100%. The authors concluded that logistic regression could be a useful tool for predicting animal occupancy when variables are measured at multiple temporal and spatial scales.

In summary, at the landscape scale, most models indicated that the probability of murrelet occupancy or nesting was associated with stand age, tree height class, vertical canopy complexity, basal area (larger tree diameters), canopy closure, slope, distance to marine areas, fragmentation level, and elevation. In some cases, models improved when tree species composition was included as a variable.

4.3.1.2 Landscape Analyses Using Data from Telemetry Studies

Nearly all of the large-scale studies using telemetry to characterize murrelet nesting habitat have been conducted in British Columbia and are included in

this Evaluation Report because there is relatively little other research that has investigated the effects of topography (slope, elevation, and aspect) on nest placement and success. Some of the results of these studies are not necessarily applicable to areas south of British Columbia that do not have topography similar to this area, such as fiords and very steep slopes punctuated by frequent avalanche chutes.

Murrelet nests have been located at a variety of elevations from sea level to 5,020 feet (1,530 m) (Burger 2002). In early research, most marbled murrelet nests were located in low elevation coniferous forests, on the lower two-thirds of forested slopes (mean 1,089 feet [332 m], range 46-3,399 feet [14-1,097 m]) (Hamer and Nelson 1995, Chapter 4 this document). In Washington, Oregon, and California, nests continue to be found at lower elevations primarily because in some areas high elevations (>2,625 feet [>800 m]) are not present and in other areas suitable habitat does not occur at high elevation (e.g., Hamer et al. 1994). Recent telemetry research in British Columbia has shown that murrelets will nest on steep slopes at high elevation (Manley et al. 2001, Bradley 2002, Burger 2002, Huettmann et al. 2003a,b). Overall, 84% of nests (n=119) in British Columbia were below 3,281 feet (1,000 m), but they occurred up to more than 4,593 feet (1,400 m). Nests were at highest elevations in Desolation Sound. Outside the Desolation Sound area, most nests (75%) were below 1,969 feet (600 m) and all were below 2,953 (900 m). In a study by Kaiser and Keddie (1999) on the central mainland coast of British Columbia, 42% of murrelet nests were found in areas below 984 feet (300 m) in elevation, even though this only accounts for 3% of the total forested area. The remainder of the nests were found at elevations below 3,281 feet (1,000 m) (34% of the forested area), suggesting that murrelets are choosing to nest in sites at lower elevations relative to availability (Kaiser and Keddie 1999).

Murrelet nests have been found in a variety of conditions with respect to aspect and slope. In general, there is no evidence to suggest that they prefer a specific aspect, but a relationship between aspect and epiphyte abundance may be affecting habitat use in some areas. For example, Huettmann et al. (2003b) found that birds in Desolation Sound, British Columbia selected north aspects more often and that birds in Clayoquot Sound, British Columbia avoided flat aspects. They suggested that north aspects may have more suitable platforms as moss may be more abundant in these areas (although this was not tested). In inland areas in northern California and southwestern Oregon, murrelets have been found to be absent in dry areas that do not possess the moss or other substrates important for murrelet nesting (Dillingham et al. 1995, Hunter et al. 1998). Burger (2002) suggested that the effect of aspect on suitable murrelet habitat may be only important

in drier areas, as moister areas seem to have moss and suitable nesting platforms irrespective of aspect. Some local variation in the availability of platform moss with respect to aspect has been noted in moist areas in Oregon, however (e.g., Nelson and Wilson 2002). Murrelet nests have been located on a variety of slopes, and no research has demonstrated a preference for slope. However, extensive telemetry research in British Columbia demonstrated that murrelets frequently nested on steep slopes, and in Desolation Sound breeding success increased with slope (see below) (Bradley 2002, Manley 2003, Huettmann et al. 2003a,b). Other research in British Columbia on murrelet occupancy rates has shown negative or non-significant associations with slope (Manley 1999, Burger 2002).

The high nesting success on the steep slopes recorded in the Desolation Sound study area occurred at inland sites along avalanche chutes. These steep slopes appeared to have lower densities of nest predators (Huettmann et al. 2003b). Bradley (2002) was not convinced of the importance of slope in nest success in Desolation Sound, stating that nest inaccessibility or some other feature associated with the cliffs and avalanche chutes in the area may be what was actually influencing nest success.

Huettmann et al. (2003b) hypothesized that because of the murrelet's high wing loading, steep slopes would allow easier and safer access to their breeding sites. However, as Burger (2002) pointed out, gaps in the forest canopy provide access to limbs in the canopy regardless of slope. In addition, nesting along edges provides easy access to nest sites. Therefore, slopes do not appear to be an essential topographic component of murrelet habitat.

4.3.1.3 Landscape Level Analyses Using Audio-Visual Detection Data

A number of studies using audio-visual detection data have been used to characterize murrelet nesting habitat at a landscape scale in the 3-state area. These studies are summarized below.

- Meyer and Miller (2002) - Meyer and Miller (2002) used intensive audio-visual surveys (4,033 surveys at 3,609 stations) in the Siskiyou National Forest of southern Oregon and northern California to compare plots with evidence of murrelet occupancy to those with no occupancy. Ground-proofed Landsat thematic mapper imagery was utilized for the vegetation database. Habitat associations were tested at spatial scales of 1,312, 2,625, 5,249, and 10,499 feet (400, 800, 1,600, and 3,200 m) radii. They found that occupied landscapes had less fragmented and isolated

old-growth forest than unoccupied landscapes. Old-growth was defined as large, multilayered coniferous forests with trees ≥2.7 feet (≥82 cm) diameter-at-breast height (dbh) comprising at least 10% of the canopy cover and total canopy cover ≥40% (Hamer and Nelson 1995). They also found that occupied landscapes were relatively close to the coast and associated river mouths, and were mostly restricted to the fog-influenced western hemlock zone. Overall, occupied landscapes tended to have: (1) large core areas of old-growth; (2) old-growth patches with greater lengths of edge relative to area; (3) high contrast between the edges of old-growth and adjacent areas; and (4) low amounts of edge overall (contiguous matrix).

- Ripple et al. (2003) -Ripple et al. (2003) studied landscape patterns around 41 documented nest sites found during audio-visual surveys in the Oregon Coast Range between 1990 and 1998, representing all the nest sites found during this time period. They compared landscapes around nest trees (n=41) to a set of randomly located points (n=41) at a spatial scale of 0.62 mile (1 km). Random sites were limited to 32.3 miles (52 km) from the coastline, on public lands, and within stands of mature and old-growth trees. Habitat composition was based on field-checked photo interpretation, where vegetation within each circle was classified into 1 of 6 possible habitat classes. They found that all murrelet nests were located in mature-old-growth conifer forests (dominated by conifers with an average dbh ≥1.64 feet (50 cm) and having an understory of hardwood or conifer). Landscapes around the nest were best distinguished from random sites by the combination of greater amounts of pole-young (dominated by conifers with an average dbh between 13 and 50 cm) and mature-old-growth forests, less edge (perimeter density in 0.62-mile [1-km] radius plots and high contrast edge at nest patches), and more cohesive nest-patch shape.

- Meyer et al. (2002) - Meyer et al. (2002) investigated habitat suitability, as determined by occupancy and abundance, for murrelets in southern Oregon and northern California at multiple scales, from patch through landscape and regional. They used 2 vegetation layers: a fine resolution (0.15-acre [0.06-ha] minimum mapping unit) map of vegetation in northern California and southern Oregon based on Landsat TM imagery, and a course-scale (4.9-acre [4-ha] minimum mapping unit) map for northern and central California based on aerial photography that was ground-truthed. Analyses were conducted separately for fine- and coarse-scale vegetation maps. Old-growth was defined as coniferous forest having ≥40% canopy coverage and a mean dbh of trees ≥2.5 feet (≥77 cm) for Oregon and ≥ 3.0 feet (≥91 cm) for California. Habitat

associations were tested at spatial scales of 1,312, 2,625, 5,249, and 10,499 feet (400, 800, 1,600, and 3,200 m) radii. Overall, they found that the variables indicative of nesting that were common to both spatial scales at the landscape level were low elevation, proximity to marine areas with high chlorophyll, proximity to other plots with murrelets, low old-growth fragmentation in the mid-1980s, and location within the fog zone.

- Hunter et al. (1998) -Hunter et al. (1998) examined the presence or absence of murrelets in the inner north coast range of California, south of the Klamath Mountains section within Conservation Zone 4. A stratified random sampling design was utilized to survey within the 2 coniferous forest habitat types most likely to be used by murrelets: late mature and old-growth Douglas-fir and late mature and old-growth tanoak (*Lithocarpus densiflora*). Within the study area, 30.8% of the Douglas-fir sampling units were surveyed in 1995 and 1996, and 58.6% in the tanoak stratum were surveyed in 1995. Murrelets were not detected in either habitat. Due to the high power associated with their findings, the authors concluded that their study area was not within the current range of the marbled murrelet. These conclusions have the potential to be biased as there was only 1 year of surveys conducted in the tanoak sampling units; however, recent studies (Meyer and Miller 2002, Meyer et al. 2003) corroborate their findings.

- Raphael et al. (1995) - Raphael et al. (1995) analyzed landscape habitat selection at 2 scales: a broad-scale analysis within major river basins, and a more site-specific analysis on the influence of landscape characteristics immediately adjacent to survey sites on occupancy status. They generated statistical measures for both scales of analysis using GIS and landscape pattern programs at 261 previously surveyed locations throughout western Washington, including the Olympic Peninsula. The survey locations were grouped into 3 classes: those with detections and occupancy, those with detections but not occupancy, and those without detections.

  Proportions of old-growth forest and large-saw timber were greater at sites that were occupied by murrelets compared to sites where they were not detected. Mean size of patches of old-growth and large saw timber were also greater among occupied sites compared to sites where occupied behaviors were not observed or no detections were recorded. In general, occupied sites had more complex landscape patterns with more edge, a greater variety of cover types, and more complex shapes (greater lengths of edge relative to area of patches). The 3 categories did not differ in their distance from the ocean (mean=19.0 miles [30.6 km]) or

elevation (mean=1,581 feet [482 m]), but differed significantly in the proportions of old-growth forest surrounding each survey sites.

- Hamer (1995) – In a study in western Washington, using a sample of 262 surveyed stands, Hamer (1995) found that 98% of all detections occurred within 39.1 miles (63 km) of the ocean and below 3,500 feet (1,067 m) elevation, with rapid drop-offs in detections and percent of stands verified as occupied beyond these points. The author used logistic regression to contrast stand attributes between occupied (n=64) and unoccupied (n=87) stands. Probability of occupancy was positively correlated with increasing slope and negatively correlated with increasing elevation. Slope was likely correlated to occupancy because much of the low elevation old-growth in the study area that would have existed on gentler slopes had been harvested. Increasing elevation likely had a negative effect on occupancy due to a higher proportion of silver fir (*Abies amabilis*) in the stands with increasing elevation and higher lichen cover (displacing moss cover). Silver fir trees typically have few platforms, low mean tree diameters, and steeply sloping branches to shed snow making level potential nest platforms less likely.

In summary, studies using audio-visual detection data to characterize murrelet nesting habitat at a landscape scale in the 3-state area have often found murrelet use to be associated with:

- The presence of mature and old-growth forests;
- Larger core areas of old-growth;
- Low amount of edge (with 1 exception) and lower fragmentation levels; and
- Proximity to the marine environment.

In some studies, murrelet nesting was also associated with lower elevations, more complex landscape patterns (i.e., old-growth within a matrix of mature second growth), and areas that were close to other similar stands on the landscape. In California, areas used for nesting were associated with the fog-influenced coastal areas.

4.3.1.4 Landscape Analysis Using Data from Radar Studies

Radar has been widely tested in several studies, and the advantages and limitations are becoming well documented (Cooper et al. 1991, 2001; Hamer et al. 1995; Burger 1997, 2001, 2002; Cooper and Hamer 2003; Bigger and

Chinnici 2003). To date, however, there has been only 1 study conducted using radar in the 3-state area that has investigated landscape-scale relationships between abundance of marbled murrelets and distribution of nesting habitat; there have been numerous studies conducted in British Columbia.

- Raphael et al. (2002a) – Between 1998 and 2000, Raphael et al. (2002a) used radar to count numbers of marbled murrelets flying inland within 10 river drainages on the Olympic Peninsula, Washington. They utilized a mosaic forest-cover map created for the Service to support designation of nesting habitat for the northern spotted owl. This map was then limited to an elevation below 3,500 feet (1,067 m), where the majority of use by murrelets was found, in studies conducted in Washington (Hamer and Nelson 1995), and was combined with a GIS layer from the Olympic National Forest, which defined watersheds by natural topographic features and streamflow patterns. Late seral forest (defined as >70% crown closure from trees >1.7 feet [53 cm] dbh) was used to represent potential murrelet habitat.

  In each of the 3 years sampled, radar counts of murrelets entering watersheds increased linearly with increasing amount of late-seral forest, greater amounts of core area of late-seral habitat, and decreasing distance between late-seral patches to similar patches. Numbers decreased with increasing amounts of edge created by the juxtaposition of late-seral patches with other land cover types.

- Burger (2001) - In Clayoquot Sound, Vancouver Island, British Columbia, Burger (2001) examined landscape-level habitat associations in 18 watersheds. Radar surveys were conducted from 1996-1998. Habitat features for each watershed were derived from overlays of 3 GIS digital databases: 1:250,000 Biogeoclimatic Ecosystem Classification (BEC) from British Columbia Ministry of Forests (MOF); 1:250,00 Baseline Thematic Mapping from British Columbia Ministry of Environment, Lands and Parks; and the 1:20,000 Clayoquot Sound Watershed Atlas from MOF. To account for murrelets using the shortest access routes by crossing low ridges into neighboring watersheds, Burger (2001) adjusted counts within 5 of the watersheds to ensure that murrelet counts were matched to the appropriate watershed areas.

  Radar counts at the 18 watersheds were significantly correlated with total watershed area, areas of mature (>140 year old) forest, and most strongly associated with areas of mature forest below 600 m. After controlling for total watershed area, significant negative correlations

were found with logged and immature areas. Three of the 5 watersheds with extensive logging of low-elevation mature forest had fewer murrelets per area than unlogged watersheds or those that were <10% logged, but these differences disappeared once remaining low-elevation mature forests were considered. Burger (2001) suggested that murrelets were responding to the loss of low-elevation mature forest by leaving heavily impacted watersheds rather than nesting at higher densities within the remaining habitat.

- Cullen (2002) - On the Sunshine Coast, British Columbia, Cullen (2002) conducted radar counts in 2000 and 2001 in 21 watersheds to investigate landscape-scale habitat associations for marbled murrelets. Habitat features for each watershed were derived from overlays of 6 GIS digital databases: 1:50,000 Watershed Atlas from British Columbia Ministry of Environment, Lands, and Parks; 1:250,000 BEC from MOF; 1:20,000 Trim Coastline from GeoData, British Columbia; and 1:20,000 Forest Cover from the British Columbia Ministry of Forestry, Weyerhaeuser, and Interfor.

  Marbled murrelet numbers were found to be related significantly to the size of the watershed, and most significantly with measures of Old Forest (>250 years of age). After controlling for watershed area, there were negative correlations to alpine tundra (>5,249 feet [1,600 m]) and mature (140-250 yrs) aged forest. In terms of forest elevation, higher BEC zones (>2,123.5 feet [650 m]) correlated more strongly to increased murrelet numbers than lower BEC zones (<2,123.5 [650 m]). This differs from studies conducted by Burger (2001) and Manley et al. (2001), where counts were more strongly correlated to lower BEC zones. Cullen (2002) suggests that this difference is likely explained by the paucity of forest at low elevations, as commercial forestry has operated in the study area for over a century and the majority of low elevation old-growth stands have been removed. Densities of marbled murrelets did not significantly increase with greater percentages of immature and recently logged habitat within a watershed, or in relation to the amount of habitat originally available before industrial timber extraction. This concurs with other studies that propose that marbled murrelets do not pack into higher densities when nesting habitat is removed (Burger 2001, Manley et al. 2001).

- Steventon and Holmes (2002) – In 2001, Steventon and Holmes (2002) conducted a preliminary analysis of radar counts and habitat at 26 watersheds on the northern mainland coast of British Columbia. Habitat feature summaries were derived in a similar manner to Burger (2001) and

Cullen (2002). Like other studies, they found that the age-class, height-class, and canopy closure were positively associated with murrelet density; higher elevations and generally steeper-sloped biogeoclimatic variants were negatively associated with density. Age-class and elevation had the strongest correlation with density (positive and negative, respectively).

- Drever and Kaiser (1999) - On the central mainland coast of British Columbia, Drever and Kaiser (1999) compared radar counts with habitat measures at 15 sites. They used somewhat different methods than those in the other radar studies reviewed here; rather than placing the radar stations at the mouths of discrete watersheds, stations were placed at constrictions and mouths of inlets, and therefore could count murrelets entering several watersheds. Habitat features were derived from GIS data for all watersheds that had some part within 9.3 miles (15 km) of the ocean.

  Multiple regression models using mean dawn counts selected areas containing fir species (*Abies* spp.), alpine habitat, and slope from 0-30° as significant predictors of variation of counts. Maximum dawn counts were found to be strongly associated with fir and hemlock species and negatively associated with areas between 0 and 984 feet (0 and 300 m) elevation. In contrast to other studies in British Columbia, Drever and Kaiser (1999) found that none of the variables traditionally viewed as measures of good habitat correlated with radar counts of incoming murrelets, but put forth no explanation for their absence. Two possible factors in this lack of correlations are the seasonal timing of surveys and the methodologies used. The counts in this study occurred primarily in April, when inland murrelet detection activity is generally at lower levels (Burger 2000, Manley 1999); therefore, the total numbers of murrelets entering each watershed were likely underestimated. Second, the location of the radar survey stations have the potential to obscure murrelet associations with a particular watershed, as murrelets were counted entering several watersheds from a single radar station.

In summary, studies in Washington and British Columbia have often shown radar counts of murrelets to be:

- Positively associated with total watershed area;
- Positively associated with increasing amounts of late-seral forests;
- Positively associated with increasing age class and height class of forests; and

- Negatively associated with increasing edge and areas of logged and immature forests and alpine tundra.

Although radar counts have been shown to be positively associated with the amount of low elevation late-seral forests, exceptions exist (Drever and Kaiser 1999, Cullen 2002). Raphael et al. (2002a) only examined habitat below 3,500 feet (1,067 m) in elevation, thus excluding any elevation effect. Cullen (2002) suggested that a positive correlation to habitat at higher elevations was due to the removal of a majority of the old-growth habitat found at lower elevations. Drever and Kaiser (1999) found radar counts to be negatively associated with areas at lower elevations, but provided 2 possible factors explaining this relationship, including differences in the seasonal timing of surveys and the methodologies used compared to other radar studies.

In addition, several studies concluded that marbled murrelets do not pack into higher densities within remaining habitat when nesting habitat is removed (Burger 2001, Manley et al. 2001, Cullen 2002). In his review of radar studies of marbled murrelets, Burger (2002) concludes that several studies show evidence that logging portions of watersheds has a detrimental effect on murrelet numbers.

4.3.2 Habitat Characteristics at the Stand/Nest Plot Level

Numerous studies have been conducted to correlate murrelet nesting with forest characteristics on a stand scale. Variables include tree density, mean height and diameter, and mean number of platforms. Murrelet habitat characteristics at the stand/nest plot scale have been identified primarily through predictive models and audio-visual surveys. Results of these studies are summarized in the following sections.

4.3.2.1 Predictive Models of Habitat Suitability at a Stand/Nest Plot Scale

Predictive models of habitat suitability at the stand/nest plot level have been developed throughout the range of the marbled murrelet and are summarized below.

- Waterhouse et al. (2002) - Waterhouse et al. (2002) examined stand-level habitat associations of murrelet nest sites located by radio telemetry in the Sunshine Coast area of south coastal British Columbia. They used aerial photographs to estimate forest structural attributes for 45 nest polygons. Using logistic regression, they found that of the 5 attributes

tested (mean tree height, mean basal area, mean age, mean crown closure, and canopy vertical complexity class), vertical complexity best predicted which forest polygons would be used for nesting and which polygons were likely to contain a successful nest. Models using variable stand age and vertical complexity as covariates showed improved probability of murrelet nesting as forest age increased beyond 140 years, but the probability appeared to increase at a decreasing rate for polygons >200 years old that were vertically complex. Polygons that contained murrelets that had reached mid-chick stage (a measure of nest success, but actual outcome not known) were a minimum of 150 years old (Burger 2002). The authors stated that vertical complexity has been related to micro-habitat features associated with higher quality habitat and nest sites, such as platform size and epiphyte cover. Vertical complexity was defined as >20% height differences between canopy dominants and the average tree canopy layer, and the presence of visible canopy gaps (Waterhouse et al. 2002). Models including tree height and vertical complexity as covariates indicated that murrelet nesting probability increased in polygons with taller trees that were more vertically complex.

- Bahn and Newsom (2002a) - In Clayoquot Sound, Vancouver Island, British Columbia, Bahn and Newsom (2002a) tested whether the use of nesting habitat by marbled murrelets could be predicted from models of habitat suitability on a large scale using mapped forest information. Instead of testing a variety of multi-variable models of habitat suitability, they tested predictions from a simple 1-variable model based on a single mapped forest characteristic available from resource maps. In an analysis of habitat data from 118 vegetation plots, tree height was found to be the most useful variable to predict habitat suitability. They then compared audio-visual detections of murrelets at 11 pairs of stands having trees on average >115 feet (35 m) tall and stands with trees <85 feet (26 m) tall. Higher numbers of occupied detections and sub-canopy detections were recorded in the stands with greater tree heights, indicating that breeding activity could be predicted based on a mapped forest characteristic (Bahn and Newsom 2002a). There were no differences found using total detections as a comparative factor. Height of dominant tree species was correlated to tree basal area, vertical canopy complexity, and the age of the dominant tree species.

- Conroy et al. (2002) - Conroy et al. (2002) used intensive tree climbing techniques to locate murrelet nests in habitat rated as 'Excellent', 'Good' and 'Sub-optimal' by the Bahn and Newsom (2002b) habitat suitability model. Five nests were found in habitat rated as Excellent and no nests were found in habitat rates as Good or Sub-optimal. They also found

that habitat rated as Excellent had higher epiphyte thickness on trees, greater tree height, greater variability in tree heights, and larger tree diameters than Good or Sub-optimal habitats. Tree density was lower and canopy closure was higher in Excellent habitat compared to Good or Sub-optimal habitats. Good and Excellent habitats had higher densities of platforms and higher densities of trees with platforms than Sub-optimal habitat. Another evaluation of the model examined the distributions of 36 nest locations found by radio-telemetry (Lank et al. 2003). Nest densities were approximately 3 times higher in the highest ranking habitat suitability classes, although several nests were located in the lowest ranked Unsuitable category.

- Hamer (1995) - Using stepwise logistic regression, a predictive model of habitat suitability in Washington was developed by Hamer (1995). In a comparison of stand characteristics between occupied and unoccupied stands in western Washington, the probability of stand occupancy by murrelets was positively associated with platform densities, stem density of dominant trees, moss cover of limbs, slope, and the presence of large diameter western hemlock. The probability of occupancy decreased with increasing lichen cover (dryer and colder, high elevation forests), elevation, and canopy closure. The study found that the presence of potential nesting platforms was a better predictor of occupancy than tree size since other factors can help produce nesting platforms including mistletoe, heavy moss cover, and tree defects. The positive relationship between occupancy and slope was likely due to the harvest of much of the low elevation old-growth in the study area, leaving more mature stands on steeper slopes. A negative association of occupancy to increasing elevation was explained by the abundance of Pacific silver fir (*Abies amabilis*) and mountain hemlock (*Tsuga mertensiana*) at higher elevations. These 2 tree species were found to have a lower mean tree diameter and lower platform abundance compared to tree species found at lower elevations such as Douglas-fir, western hemlock, Sitka spruce (*Picea sitchensis*), and western red-cedar (*Thuja plicata*). Stands with high canopy closures may have had less access for murrelets to nesting platforms. Overall, the model correctly predicted occupancy in 74% of the sites.

- Perez-Comas and Skalski (1996) - In 1996, Perez-Comas and Skalski compared several stand characteristics (tree density and platform density for four dbh size categories) between a sample of occupied and unoccupied sites located in the western Olympic Peninsula (Washington). They used analysis of variance to determine the importance of each habitat variable and to estimate the variance of each component. They

then used discriminate analysis to estimate the discriminatory power of different combinations of habitat variables and constructed simple models to predict the probability that a forest site was occupied. The most parsimonious models included the following 3 variables: total number of nesting platforms on trees>3.0 feet (90 cm) in diameter; density of trees >3.0 feet (90 cm) in diameter; and the total number of platforms on all trees within each plot. Due to habitat differences between geographic regions sampled, better discrimination was achieved by performing separate analyses on the northern and southern portions of the study area. Maximum correct classification accuracies for the northern and southern study areas were 79.6% and 89.4%, respectively. These models were later used to develop a predictive model for the Washington State Forest Practices Board permanent State rule protecting marbled murrelet habitat (Pierce 1997).

- Hamer (1996) - In a comparison of habitat characteristics at a sample of occupied and unoccupied stands in southwest Oregon, logistic regression was used to determine which combination of habitat variables best predicted occupancy of stands by marbled murrelets (Hamer 1996). Cross validation procedures were used to test the model on an "independent" sample of stands. Forty-one forest variables were measured from a sample of 21 occupied and 21 unoccupied sites. T-test results from independent samples indicated that 7 variables were significantly different between occupied and unoccupied sites.

  Occupied sites had higher potential nest platform densities, higher densities of platform trees, higher percent moss cover on tree limbs, increased moss depths, more canopy layers, and were found on more moderate slopes. A predictive model consisting of platform density and percent slope had one of the highest classification accuracies, preformed well under cross validation, and made biological and intuitive sense. Overall classification accuracy was 78.6%. Moss depth and moss cover variables were most likely not included in the model because platform diameters and platform counts in the field were made with moss cover included in the estimate of diameter. Thus, platform density took into account the amount of moss cover and depth of moss on tree limbs and was positively correlated to both variables. Slope was negatively correlated with occupancy because sites on steeper slopes were typically located near ridgetops and away from the lower valley bottoms. These sites typically had lower mean tree diameters, lower density of trees with platforms, and platform density was either low or platforms were absent.

- Nelson et al. (2002) - In a study by Nelson et al. (2002) at 6 sites in the North Cascades and Olympic Peninsula in Washington and 34 sites in the Oregon Coast Range and Klamath Mountains in western Oregon, intensive tree climbing in 2 to 13 randomly located circular plots (131-feet [40-m] radius; n=215) per site was used to find murrelet nests. All trees with potential nest platforms in each plot were climbed. Once nests were located, detailed habitat characteristics were collected at the platform, tree, and micro-site (plot) scales. A variety of characteristics at each nest and non-nest or random site were measured. They located 49 murrelet nests in 16 stands and 39 plots after climbing 3,385 trees.

  Models were developed to explore the potential relationship between the probability of murrelet nesting and each of the selected explanatory variables at the platform, tree, and micro-site scales. They developed 22 a priori models thought to include the most likely factors or combinations of factors for distinguishing between nest and non-nest sites. These models were analyzed using logistic regression. AIC (small sample variant of Akaike's Information Criterion) were used to select the best approximating models at each scale; other models were ranked by differences in AIC values, where lower values indicate a better model. Models from each scale are summarized below.

  - Site Scale – Tree climbing found that nest trees were located in portions of the study sites with many platform trees, high platform density, and increased flight access (or "open flight corridors" based on slope, distance to closest edge, and number of canopy layers). Access was considered important for adult murrelets when flying into and out of their nest tree and nesting area and to facilitate the fledging of young. Of the 7 a priori models developed at this scale, the model with density of platforms (number/ha) and number of canopy layers had the lowest AIC value.

  - Platform Scale. Tree climbing results showed that nests were located on large tree limbs with extensive moss and overhead cover. Of the 8 a priori models developed at this scale, the model with platform width (cm), percent moss cover, substrate depth (cm), and vertical cover (%) had the lowest AIC value.

  - Tree Scale -Tree climbing results found that nest trees were large in diameter, contained numerous platforms, and were covered with extensive moss and often had infestations of dwarf mistletoe. Of the 7 a priori models developed at this scale, the

model with number of platforms, percent moss cover, and percent mistletoe infestation had the lowest AIC value.

- Meyer et al. (2003) – Using a multi-scale model in northern California, Meyer et al. (2003) found that at the patch scale, occupied sites were lower in elevation and contained larger trees compared to unoccupied sites. For the time scale, birds showed a lag in response to fragmentation, with birds occupying fragmented forests only if the fragmentation was recent.

In summary, at the stand and patch scale, most model results included platform density, higher epiphyte thickness and percent cover of epiphytes, greater tree heights and canopy complexity (including number of canopy layers), larger tree diameters, densities of large trees, elevation, and slope as predictive variables.

4.3.2.2 Nest Stand/Plot Analyses Using Audio-Visual Detection Data

Data from audio-visual studies have shown that murrelets tend to occupy forest stands with a complex structure, relatively large conifers, and a relatively large number of platform trees with epiphyte cover. In their summary of all nests found in North America by 1995, Hamer and Nelson (1995) found that nest stands had a total mean tree density of 73.7 trees per acre (182 trees/ha) in the Pacific Northwest, with the highest densities found in Alaska and the lowest in Washington (Table 4.3-1).

In general, murrelets tend to choose stands with larger platform trees with a high percentage of epiphyte cover. Many of the differences seen in nest site

Table 4.3-1. Summary of nest stand characteristics[1].

	Mean total tree density (#/ha)	Mean canopy height (m)
Alaska	575 (295-978)	23 (16-30)
British Columbia	297 (20-100)	ND
Washington	90 (78-100)	54 (44-59)
Oregon	120 (48-282)	59 (48-75)
California	235 (92-504)	88

[1] *Source: Hamer and Nelson (1995)*

selection studies between use and non-use areas at the landscape or at the nest tree scale; however, some studies have shown differences at the stand scale.

- Hamer and Meekins (1999) - Nest plots surveyed by Hamer and Meekins (1999) in western Washington had an average of 21.1 trees per acre (52 trees per ha) that were greater than 2.6 feet (80 cm) in diameter. Nest plots had an average canopy closure of 89% and an average of 3.5 canopy layers. Within these plots, platform trees had an average diameter of 3.5 feet (108 cm), with a range of 1.2 to 10.3 feet (37 to 315 cm). Percent moss cover in platform trees in plots averaged 66%, with an average depth of 1.2 inches (3.1 cm). However, only 2 characteristics were significantly different between plots with and without nests. Nest plots had greater mean platform diameters and lower rodent activity than non-nest plots. Many other variables, including number of trees >2.6 feet (80 cm) diameter per ha, mean tree dbh, number of trees/ha with platforms, and number of platforms on western hemlock, were not statistically significant between plots with and without nests.

- Nelson and Wilson (2002) – A study in western Oregon found that nest plots tended to have more platform trees, platforms, and canopy layers than non-nest plots. Also important in nest-site selection was distance to nearest natural opening; murrelets tended to select plots that were closer to openings, as this may improve access to the nest site. Non-nest plots also tended to be on steeper slopes than nest plots (Nelson and Wilson 2002).

- Rodway and Regehr (2002) - In Clayoquot Sound, British Columbia, researchers used audio-visual detections of occupied behavior to characterize murrelet nesting habitat (Rodway and Regehr 2002). They found that the density of large trees (>2.6 feet [80 cm] dbh) was positively related to occupied detections within 164 feet (50 m) and subcanopy detections, as was density of trees with platforms. Total detections were found to be positively related to density of all trees.

- Miller and Ralph (1995) - In California, audio-visual detections were used to explore the relationship of murrelets with habitat characteristics at inland sites (Miller and Ralph 1995). Researchers found that mean detection levels were positively related to the density of combined old-growth dominant and codominant tree cover and presence of redwood trees.

- Grenier and Nelson (1995) - In Oregon, Grenier and Nelson (1995) found that occupied stands tended to have older, large diameter dominant or remnant trees than random sites.

In California, it has been reported that marbled murrelets may be found in very small residual stands, and younger stands with residual trees (Hunter and Bond 2001); however, the nesting success of these birds is not known. It has been suggested that a murrelet is more likely to use a stand with residual trees if the residuals are clustered within the stand, and if the stand is located near (within 656 feet [200 m]) another stand of old-growth (LeValley and Brown 2001).

4.3.3 Habitat Characteristics at Nest Trees

Few models have examined habitat attributes at the tree and nest platform scale. Those that did found that total platform number, moss cover and depth, and percent mistletoe infestation as predictive variables at the nest tree scale. At the nest platform scale, models included limb size, moss cover, and overhead cover as predictive habitat attributes.

Several characteristics seem to be important for murrelet selection of nest trees (Burger 2002):

- Located near openings in the canopy for access to site;
- Large potential nest platforms (branches or deformities);
- Substrate for nest cup;
- Horizontal and/or vertical cover over nest site; and
- Sufficient height to allow jump-off departures and stall landings.

Old-growth conifers tend to be the only trees that provide these requirements for murrelet nesting. Exceptions have been noted; for example, Nelson and Wilson (2002) found nests in 18 young and mature trees (66-150 years in age) that were distinguished by the number of platforms provided by mistletoe infections, and 1 nest has been found in British Columbia in a red alder (*Alnus rubra*) tree (Bradley and Cooke 2001). Habitat characteristics of nest trees have been investigated by a number of audio-visual and tree climbing studies and are summarized in Table 4.3-2 and below, by location.

4.3.3.1 Alaska

Nesting habitat requirements in Alaska are not clearly understood as studies have positively identified a limited numbers of nests (Mendenhall 1992). In Alaska, nest sites have been located in stands composed of mountain hemlock and stands composed of western hemlock (*Tsuga heterophylla)* and Sitka spruce. These forested regions are variable and may only occur 656 ft (200 m) above sea level and a few kilometers inland (Kuletz et al. 1995a). The choice of nesting habitat for marbled murrelets appears superficially to be broader in Alaska, where murrelets nest both in trees and on the ground (Kuletz et al. 1995a). Given the variability in forest cover, a very small proportion of marbled murrelets, approximately 3% (Piatt and Ford 1993), nest on the ground. This usually occurs on barren, inland slopes and to the west of the major rainforests along the Alaskan gulf coast (Ralph et al. 1995).

Studies in Alaska have located 20 nests in 3 tree species including 9 mountain hemlock, 6 western hemlock, and 5 Sitka spruce (Hamer and Nelson 1995, Naslund et al. 1995). Nest trees tended to be large in diameter, contain a high number of potential platforms, and have a relatively high amount of epiphyte cover (Table 4.3-2). Of all tree species in Alaska, it is thought that Sitka spruce most likely exhibits the qualities that seem important to murrelets for nest selection (Naslund et al. 1995). No mean ages of nest trees have been reported for Alaska; however, Marks and Kuletz (2001) detected murrelet tree nests in mature and old-growth forests only. To date, no comparisons between nest tree/platforms and non-nest tree/platforms have been published for Alaska.

There have also been at least 15 nests found on the ground, in crevices, and on cliffs. It is suggested, however, that the apparent importance of ground-nesting in Alaska is partially an artifact of survey effort as ground nests are more easily discovered than tree nests (Kuletz et al. 1995a). Most ground nests have been found on islands or on mainland areas within 0.5 mile (1 km) of the ocean (maximum 3.2 miles [6 km]) with no large trees nearby (Nelson 1997, Marks and Kuletz 2001). Comparisons between nesting in forested areas versus non-forested areas were explored by Marks and Kuletz (2001); see Section 4.3.4 below.

4.3.3.2 British Columbia

Burger (2002) reports that with the exception of the red alder discovered in British Columbia, all nest trees in this region would be classified as old-growth conifers, with a likely age >140 years. Nest stands and nest trees are

Table 4.3-2. Mean murrelet nest tree and site characteristics[1].

	Nest Tree Characteristics			Nest Site Characteristics						
	Tree Age (Years)	dbh (cm)	Height (m)	Nest Height Above Ground (m)	Nest Limb Diameter at Trunk (cm)	Limb Diameter at Nest (cm)	Distance from Trunk (cm)	% Over-head Cover	Mean Moss Depth (cm)	Duff & Litter Depth (cm)
Alaska (N=14)	ND	63 (30-104)	23 (30-104)	13 (10-17)	15 (9-27)	19 (12-28)	62 (0-224)	89	3.9 (20-6.0)	ND
British Columbia2 (N=9)	>140	212 (90-370)	58 (30-80) *45*	33 (18-44) *27*	32 (15-38)	27 (15-38)	134 (0-340)	100	4.8 (2.7-7.0)	ND
Washington3 (N=5-6)	879 (450-1736)	152 (88-220) *110*	57 (45-65) *32*	37 (23-53) *20*	36 (14-49)	29 (11-46)	94.5 (0-325)	90 (70-100)	2.7 (2.0-3.5)	2.9 (2.0-3.8)
Oregon4 (N=20-22)	209 (180-350)	192 (127-279) *116* *(49-212)*	67 (36-86) *51* *(33-58)*	51 (18-73) *28* *(10-75)*	31 (14-56)	34 (10-81)	101 (1-762)	79 (40-100)	5.1 (0.6-12)	3.4 (3.0-3.8)
California (N=10)	NA	278 (139-533)	73 (61-86)	47	35 (21-61)	34 (16-61)	47 (0-184)	90 (5-100)	2.9 (0.8-8.1)	7.4 (2.5-20.0)

Primary source: Hamer and Nelson 1995

[1] *Ranges shown in parentheses.*

[2] *Values in italics are from Burger (2002).*

[3] *Values in italics are from Hamer and Meekins (1999), N=22, for western Washington.*

[4] *Values in italics are from Nelson and Wilson (2002) for western Oregon.*

summarized by region, and nest tree characteristics of a sample of these nests are summarized in Table 4.3-2.

- Queen Charlotte Islands – The Queen Charlotte Islands are characterized by 2 biogeoclimatic zones in British Columbia's coastal hypermaritime region, the Coastal Western Hemlock (CWH) zone and the Mountain Hemlock (MH) zone. The CWH zone, in which most murrelets appear to nest throughout British Columbia (Burger 2002), is dominated by western hemlock, western red cedar, and Sitka spruce, while yellow cedar (*Chamaecyparis nootkatensis*) is found in higher elevations. Lodgepole pine (*Pinus contorta*) is found in wet lowlands often associated with bog conditions. The MH zone occurs at higher elevations above the CWH zone. Dominant trees in this zone are mountain hemlock and yellow cedar. Pacific silver fir is associated with both zones on the mainland of British Columbia and Vancouver Island; however, it does not occur on the Queen Charlotte Islands.

  The 1 nest located to date in the Queen Charlotte Islands was found in a western red cedar (Dechesne and Smith 1997). However, high densities of detections have been recorded in stands of large Sitka spruce and western hemlock (Rodway et al. 1993).

- Vancouver Island – Vancouver Island is characterized by 3 biogeoclimatic zones. The CWH zone encompasses the greatest amount of area, with the MH zone in higher elevations, and a small component of the dry Coastal Douglas-fir (CDF) zone covering the southeastern coast and Gulf Islands. Douglas-fir is the dominant tree in the CDF zone, with a small component of grand fir (*Abies grandis)*, western red cedar, western hemlock, Sitka spruce, and lodgepole pine. There has been no research on marbled murrelets in this zone, but nesting is likely because the birds are often seen nearby on the ocean (Burger 2002).

  Across Vancouver Island, 25 nest trees have been identified. The majority of these have been located on southwest Vancouver Island, with the majority in Sitka spruce, followed by western hemlock, western red cedar, and 1 each in mountain hemlock, yellow cedar, and Pacific silver fir. Surveys on southeast Vancouver Island (Burger et al. 2000a) have positively identified 3 nests in Douglas-firs (Burger 2002).

  In the Ursus Valley, Clayoquot Sound, it was found that trees containing nests were significantly larger in diameter than trees that did not contain nests (5.6 vs. 3.2 feet [171 vs. 99 cm] dbh, respectively). Though not statistically significant, trees with nests had, on average, more platforms (9.8 vs. 7.2), higher percent epiphyte cover (72% vs. 54%), thicker

epiphyte cover (1.4 vs. 1 inch [3.6 vs. 2.6 cm), and were taller (137.8 vs. 114.8 feet [42 vs. 35 m]) than non-nest trees (Conroy et al. 2002).

- Sunshine Coast – The Sunshine Coast in southwestern British Columbia is characterized by 3 biogeoclimatic zones including the CWH, MH, and a small component of CDF. Whereas studies on Vancouver Island have been focused on areas of little habitat loss, the largest numbers of nests located on the Sunshine Coast have been found in high elevation yellow cedar stands subject to intensive logging (Manley and Jones 2000, Demarchi and Button 2001).

  Along the Sunshine Coast, Douglas-fir and yellow cedar tend to be used for nesting more often than would be predicted based on their availability (Manley 2003), as well as Sitka spruce (Burger 2002); however, nests have also commonly been found in western hemlock and western red cedar in this area (Burger 2002, Manley 2003).

  Intensive studies on the Sunshine Coast (Manley 1999, Bradley and Cooke 2001) have discovered 97 nests in 7 tree species. These were mainly found in yellow cedar, followed by western hemlock, Douglas-fir, western red cedar, mountain hemlock, and 1 Pacific silver fir. The single hardwood nest tree located in this region was a red alder (Bradley and Cooke 2001). Ground nests have also been confirmed in British Columbia. Bradley and Cooke (2001) suggest that at least 3% of nests in their study area (Desolation Sound and Mussel Inlet) occur on the ground.

4.3.3.3 Washington

Occupied behavior in potential nest stands has been found in 4 physiographic provinces throughout Washington (Hamer 1995). These are the North Cascades, South Cascades, Olympic Mountains, and Southwest Coast. Nest stands are primarily dominated by western hemlock, Douglas-fir, Sitka spruce, and western red cedar. Pacific silver fir and mountain hemlock found at higher elevations make up a smaller component of nest stands in Washington (Hamer and Nelson 1995).

Throughout Washington, 29 nest trees have been identified in 4 tree species. All nests found by Hamer and Meekins (1999) in western Washington were found in conifers. Eighty-three percent (n=24) were in western hemlock, 7% (n=2) in Sitka spruce, and 7% (n=2) in Douglas-fir. All nest trees surveyed by tree climbers exhibited a high percent cover of moss (mean=68%), with the exception of 1 Douglas-fir nest tree with a higher proportion of lichen (85%). Previously, 1 nest had been found in western red cedar (Hamer and

Nelson 1995). During this study, however, no nests were found in red cedar or in silver fir, even though both species were present in the study area. Nest trees had an average of 14 platforms (min.=1 and max.=32) in the 0.33 – 0.62 feet (10-19 cm) category and 14 in the 0.66+ feet (20+ cm) category (min.=1 and max.=43). For both categories combined, the minimum number of platforms per nest tree was 10, with a mean of 28. Researchers in this study suggest that in western Washington, western hemlock likely had the best combination of attributes preferred by nesting marbled murrelets.

Nest trees found in western Washington had a mean diameter of 3.6 feet (110 cm) (n=22) (Hamer and Meekins 1999). Two nest trees were found to have diameters <2.6 feet (80 cm) dbh, both of which were mistletoe-infected western hemlocks. These are the smallest diameters recorded for nest trees in Washington to date. All nest limbs were >65.6 feet (20 m) in height, at an overall average of 105 feet (32 m), within the middle to upper portion of the live crown. These values are comparable to those reported by Nelson and Hamer (1995) (Table 4.3-2).

Most (68%; n=15) nests were found in trees classified as alive, 32% (n=7) in declining, and none in dead trees (i.e., snags) (Hamer and Meekins 1999). Of these, 64% (n=14) had intact and alive tree tops, 32% (n=7) had broken tops, and 4.5% (n=1) had intact, dead tops. In a summary of 6 nests found in Washington, Hamer and Nelson (1995) found the mean age of stands containing nests to be 879 years, with a range of 450 to 1,736 years.

Hamer and Meekins (1999) report that nest cups were found on large limbs (>0.36 feet [11 cm] in diameter, >2.0 feet [0.6 m] in length), the majority of which were either primary limbs (41%, n=12), large areas where 2 limbs forked (28%, n=8), or mistletoe-infected limbs (28%, n=8). All nests were found on limbs covered with moss (>15% cover), with a mean moss thickness adjacent to the nest of 0.17 feet (5.2 cm) (min. of 0.02 feet [0.5 cm] and max. of 0.55 feet [16.8 cm]). Limb diameters at the nest averaged 0.85 feet (26 cm), and nest platform area averaged 2.2 feet^2 (2,044 cm^2) in size. The mean nest cup circumference was 1.22 feet (37.3 cm). All nests were located within 10.66 feet (3.25 m) of the tree bole, with an average distance of 3.1 feet (94.5 cm), and most were located in areas with high vertical and horizontal cover (mean=82% and 54%, respectively). These values are comparable to those reported by Nelson and Hamer (1995) (Table 4.3-2).

Nest platforms had greater limb diameters, platform surface areas, horizontal and vertical cover, and moss depths than randomly selected non-nest platforms (Hamer and Meekins 1999). Nest trees were taller, had greater canopy lift, and longer overall limb length than non-nest trees. When

compared with other non-nest platform trees, nest trees had a significantly greater number of platforms, with a threefold higher platform count for platforms 20 cm or greater in diameter. In addition, nest trees had significantly greater moss depths than other non-nest platform trees.

4.3.3.4 Oregon

In Oregon, marbled murrelet habitat is located in the Coast Range and Klamath Mountain physiographic provinces, while nesting may extend into the Willamette province. These forests are characterized by large diameter Douglas-fir, Sitka spruce, and western hemlock, as well as a small component of red alder and a mix of smaller diameter conifers and hardwoods (Grenier and Nelson 1995). Most nest stands in Oregon are dominated by Douglas-fir and western hemlock, with 1 site dominated by Sitka spruce (Hamer and Nelson 1995).

In a study of 37 nests located in western Oregon, the majority were found in western hemlock (n=25), followed by Douglas-fir (n=9), Sitka spruce (n=2), and western red cedar (n=1) (Nelson and Wilson 2002). Prior to this study, the majority of nests in Oregon were found in Douglas-fir, with 1 found in Sitka spruce and 1 in western hemlock (Grenier and Nelson 1995, Hamer and Nelson 1995). Nelson and Wilson (2002) report that the mean number of platforms on nest trees was 25.8, with a range of 4 to 92. All nest trees had moss cover (more than 50% for all but 3), with some containing lichen and mistletoe.

The 37 nest trees were found to have a mean dbh of 3.8 feet (116 cm) (range 1.6-6.96 feet [49-212 cm]); average tree height was 167.3 feet [51 m] (range 108.3-272.9 feet [33-85 m]) (Nelson and Wilson 2002). These values are considerably smaller than the averages of 6.3 feet (192 cm) and 219.8 feet (67 m) for dbh and height, respectively, reported by Hamer and Nelson (1995) (Table 4.3-2) and are likely related to the number of nests found in young western hemlock trees with mistletoe deformations. The majority of nest trees previously found in Oregon were in Douglas-fir; most of the nest tress in the study by Nelson and Wilson (2002) were western hemlock. The average height of nest limbs was 91.9 feet (28 m) (range 32.9-296.1 feet [10-75 m]) (Nelson and Wilson 2002).

In a summary of 20 nests found in Oregon, Hamer and Nelson (1995) found the mean age of stands containing nests to be 209 years, with a range of 180 to 350 years. In western Oregon, nests were found in stands that were either old-growth or mature with remnant trees, with the exception of stands on the Tillamook and Clatsop State Forests that were comprised of young and

mature hemlock trees with infestations of mistletoe. Nest trees ranged in age from 66 to 363 years, with an average age of 167 years. Younger nest trees were found in stands with young western hemlock trees that had abundant platforms formed by mistletoe infections (Nelson and Wilson 2002).

Nest trees had a mean nest platform diameter of 0.69 feet (21 cm) (range 0.39-1.28 feet [12-39 cm]), at a mean distance of 3.22 feet (98 cm) from the trunk (range 0-114.8 feet [0-35 cm]), and a mean platform length and width of 2.39 feet (79 cm) (range 0.25-14.76 feet [7.5-450 cm]) and 0.72 feet (22 cm) (range 0.23-1.44 feet [7-44 cm]), respectively. The mean substrate depth on platforms was 0.06 feet (1.7 cm) (range 0-0.2 feet [0-6 cm]), comprising of an average of 81% moss and 3% lichens. Cover above the nests had a mean value of 55% (range 15-88%) (Nelson and Wilson 2002).

At the plot scale, murrelets in western Oregon tended to select trees that were larger in size, with more and larger platforms, more moss, and more horizontal cover over platforms than other non-nest trees (Nelson and Wilson 2002). Murrelets appear to select nest platforms with a larger diameter and width with more horizontal and vertical cover than available platforms in the nest plot. Of these characteristics, platform width was the best predictor of nest platform selection by murrelets. Selected limbs also tended to have more substrate (i.e., moss) than non-nest limbs (Nelson and Wilson 2002).

4.3.3.5 California

In California, nest stands have been found in the Coast and Klamath physiographic provinces. These are dominated by coast redwood (*Sequoia sempervirens*) and Douglas-fir, while a small component of stands contain western hemlock and Sitka spruce. In both central and northern California, all nest sites had a higher percentage of redwood trees than Douglas-fir (Hamer and Nelson 1995).

Throughout the Californian range of the marbled murrelet, a total of 14 nests have been located in 3 tree species, with the majority in coast redwood, followed by Douglas-fir and 1 in western hemlock (Burger 2002). In a review of 10 nests in California, Hamer and Nelson (1995) report 5 nests found in coast redwood, 4 in Douglas-fir, and 1 in a western hemlock. One of the greatest heights ever recorded to date of a murrelet nest occurred in Humboldt County, on a redwood limb 80 m above the ground (Spickler and Sillett 1998). Average nest site characteristics in California appear similar to Oregon and Washington (Table 4.3-2). While data are currently being

collected in California to compare nest trees with non-nest trees, no clear trends have yet to be reported (Suddjian 2003).

4.3.4 Habitat Characteristics at Ground Nests

In Alaska, 15 ground nests have been found in a variety of locations, including mossy cliff ledges, among tree roots on cliffs or steep slopes, in rock crevices, under crevices, and in the open (Degange 1996, Nelson 1997). Four of these nests were found in forested areas near the ocean in south-central and southeastern Alaska (Degange 1996); however, the majority were found within 1 km of the ocean with no large trees nearby (Nelson 1997, Marks and Kuletz 2001). In this area, the low incidence of predators may facilitate using treeless areas for nesting (Marks and Kuletz 2001). It has also been suggested that the use of unforested areas for nesting may be both a product of limited forest cover, as well as limited prey availability near forested areas (Marks and Kuletz 2001). In Degange's (1996) review of nests found in Alaska, 17 of 33 nests were ground nests; however, the author felt this may have been a reflection of the higher probability of finding ground nests in Alaska rather than a true estimate of percentage of ground-nesting versus tree-nesting murrelets.

On the basis of marine distribution during the breeding season, Piatt and Ford (1993) estimated that approximately 3% of the Alaskan murrelet population may nest on the ground. In Alaska's Kenai Fjords National Park, Marks and Kuletz (2001) found that inland murrelet detections were appreciably and significantly higher in forested areas than unforested areas. Within unforested areas, murrelet land use may be related to the amount of low vegetation and shrubs, as stations with more vegetation had higher dawn activity. They suggest that although marbled murrelets nest on unforested ground in south-central Alaska, many of these nests share the same traits as tree nests, such as a moss or litter substrate and overhead cover, and most importantly, are located on steep slopes to allow for a drop take-off. Because of the similarities at the microhabitat scale between ground and tree nests, it has been suggested that ground nests only serve as an alternative to tree nests when forested areas are lacking (Marks and Kuletz 2001).

Bradley and Cooke (2001) documented 1 confirmed and 2 probable cliff nests used by radio-tagged birds in southwestern British Columbia. Although marbled murrelets had not previously been observed nesting on the ground south of Alaska, the "structure" of these nest sites (heavy epiphyte cover, large platform, vegetative cover, and flyway opening) is similar to conifer nest trees. Bradley and Cooke (2001) proposed that the selection of ground nests by marbled murrelets in their study area may be

due to heavy modification of the original old-growth forest. In all 3 of their sites, moss-covered ledges and crevices seemed to provide many more potential nesting platforms than any adjacent trees. Ground nests comprised 3% (1/30) of confirmed nest sites and 3% (2/78) of suspected sites. No ground nests have been found south of British Columbia (Nelson 1997), and suitable habitat for ground nesting is extremely limited in close proximity to the ocean.

There is little genetic or morphological divergence between tree- and ground-nesting populations of marbled murrelets in Alaska (Pitochelli et al. 1995). Congdon et al. (2000) concluded that population genetic structure of British Columbia and Alaska birds shows no selection associated with different nesting habits. There is no information on whether individual birds switch from tree to ground nests with the disappearance of suitable tree habitat. The transition from 1 habitat to another might not be readily made, if there is strong habitat imprinting of nestlings. There have been >15 records of birds returning to the same tree in successive years; however, it is not certain as to whether they are the same birds (Nelson and Peck 1995, Singer et al. 1995, Manley 2000). Marbled murrelets exhibit behavior that is consistent with that of other adult alcids, a high level of philopatry, returning to the same nesting area and often the same nest site over many successive years.

4.4 Marbled Murrelet Densities

There have been relatively few studies that have estimated the density of marbled murrelets. However, the results of some recent radar studies have been used to estimate density on a watershed level. Intensive tree climbing techniques have also been used to locate old and active visible nest sites, allowing an estimate of historic and active nest densities.

4.4.1 Estimated Densities of Marbled Murrelets from Radar Counts

Evidence for significant relationships between murrelet numbers and areas of suitable nesting habitat from radar studies suggests that densities derived from these studies (birds per ha of habitat) would be useful as a management tool. A density measure could be used to calculate the area of forest needed for a specific population of murrelets or, conversely, to estimate the numbers of murrelets likely to be using a specific area of forest.

Table 4.4-1 lists density estimates from 5 separate radar studies. The major problems in deriving and applying densities are in correctly defining and measuring suitable habitat. Several different measures of habitat are listed in

Table 4.4-1, as there was no common measure in all studies. This could be corrected by reanalysis of the data using 1 or more measures applied to all the watersheds. Burger (2002) states that the application of these density estimates in management decisions should be done with caution. For example, densities derived from general measures of habitat (e.g., all mature and old-growth at all elevations) cannot be applied to selected patches of habitat known to contain a high proportion of likely nesting habitat. Ideally, the habitat criteria used to derive the density measures should be the same from watershed to watershed, which would allow for more accurate comparisons of densities.

Table 4.4-1. Densities of marbled murrelets (birds/ha) estimated from radar counts of birds entering watersheds and areas of habitat derived from GIS.

Study Area	Density Estimates		Radar Est.	Habitat Measure	# of Water-sheds	Reference
	Mean	SD				
Sunshine Coast, BC	0.021	0.015	Mean 2 yrs	Forest age >250 yrs, all BEC zones below Alpine Tundra	21	Cullen and Manley et al. 2001 and 2001 surveys
	0.029	0.022	Max 2 yrs			
Clayoquot Sound, BC	0.067	0.024	Mean 3 yrs	Forest age >140 yrs, BEC zones <600 m elevation	18	Burger 2001
	0.083	0.030	Max 3 yrs			
Northwest Vancouver Island, BC	0.083	0.043	Mean 1 yr	Forest age >140 yrs, BEC zones <600 m elevation	20	Manley 2000
Central Coast, BC	0.042	0.012	Mean 1 yr	Forest age >140 yrs, all BEC zones below Alpine Tundra	22	Schroeder et al. 1999
Olympic Peninsula, WA	0.005	0.001	Mean 3 yr	Late Seral Forest, <1.067 m elevation	10	Raphael et al. 2001

Source: Cullen 2002
SD=standard deviation

4.4.2 Estimates of Nest Density Using Intensive Tree-Climbing Methods

In western Washington, Hamer and Meekins (1999) calculated an average density (visible nests, both new and old) of 0.4 nests per acre (0.9 nests per ha) in surveyed stands, with a minimum density of 0.08 nests per acre (0.2 nests per ha) and maximum of 0.8 nests per acre (2 nests per ha). Nest densities were calculated by dividing the area that was climbed by the number of nests found during tree-climbing searches. In the Ursus Valley on the west coast of Vancouver Island, British Columbia, tree-climbing searches resulted in tree nest density estimates of 0.26$\pm$0.12 visible nests per acre (0.66 $\pm$ 0.29 visible nests per ha), and 0.05$\pm$0.05 active nests per acre (0.13 $\pm$ 0.13 active nests per ha) per year (only 1 active nest was found in the study area)

(Conroy et al. 2002). In fragmented forests along the Sunshine Coast, British Columbia, nest densities were found to range from 0.12 to 0.28 nests per acre (0.3 to 0.7 nests per ha), with a maximum density of 1.7 nests per acre (4.2 nests per ha). No distinction was made between old and active nests in these calculations (Manley 1999).

In Oregon, tree climbers calculated visible nest densities at the plot level (Nelson and Wilson 2002). They found that nest density ranged from 0.1 to 3.0 nests per ha; however, because habitat patches tended to be clustered, they suggested that densities may be lower at the stand level.

4.5 Terrestrial Threats

The primary ongoing threats to marbled murrelets in the terrestrial environment include:

- Loss of suitable nesting habitat from 1992-2003;
- Effects of current land ownership on the amount of existing nesting habitat, and habitat distribution and quality;
- Projected future trends of suitable habitat;
- Natural disturbances such as wildfire, insect/disease, and windthrow effects on nesting habitat;
- Land management practices affecting nesting habitat availability and quality and survival and reproductive rates;
- Fragmentation and edge effects on forest habitat; and
- Effects of noise from recreational activity, human development, and other disturbances on adults and chicks.

These factors are discussed in the following sections as they pertain to the recent past, current, and projected future habitat conditions for the marbled murrelet.

4.5.1 Loss of Suitable Nesting Habitat

In the past 11 years, there have been many estimates of the amount of old-growth and/or suitable murrelet habitat occurring throughout the listed range (Table 4.5-1). Since 1992, estimates of suitable habitat have been adjusted down largely based on improved knowledge of the species' inland

Table 4.5-1. Estimates of old-growth/suitable murrelet habitat within the listed range, 1992-2003.

Year and Source	Estimates of Suitable Murrelet Habitat (ac)		
	Washington & Oregon	California	Total
1992 Source: the Service (1992), based on acres of old growth in WA & OR	3,400,000	No estimate provided	
1994 Source: USFS and BLM (1994), based on spotted owl habitat requirements, which have been shown to be less rigid in respect to stand age & tree size than murrelets	N/A	N/A	2,500,000
1995 Source: Perry (1995)	1,542,996	819,472	2,362,469
1996 Source: the Service (1996), based on acres of old-growth forest in WA & OR and acres of old-growth coastal redwood in CA	3,400,000	70,000	3,470,000
2003 Source: Table 4.1-1, this report	1,829,462	393,586	2,223,048

N/A – not applicable.

distribution, and more refined queries of available vegetation databases, rather than on actual loss of habitat.

Although there are no credible estimates of historic or existing murrelet habitat, the trend in habitat availability since the species was listed in 1992 has likely been negative due to continued harvest, natural disturbances, and lack of sufficient time for old-growth characteristics to develop in forested habitats of the 3-state area.

While it is important to estimate the current amount and distribution of suitable murrelet habitat throughout Washington, Oregon, and California, it is also important to the 5-Year Status Review to understand the trend in habitat since listing. For the marbled murrelet, it is especially relevant to examine changes in the amount of habitat throughout the listed range because the historic loss and continued threat of inland forest habitat loss was a key factor in the consideration to list the species in 1992 and in developing the Recovery Plan.

As part of evaluating trends in murrelet nesting habitat, the Expert Panel considered the possibility of new suitable habitat developing through natural succession since 1992. Based on the short timeframe – 11 years – it is unlikely that few, if any, forest stands would have had sufficient time to develop trees with large lateral limbs and heavy moss cover, if they had not already had these characteristics in 1992. In general, the earliest possible recovery time for nesting habitat is estimated to be 100-200 years (Service 1997). In 1997, the Recovery Plan indicated that the next 50 years would be

the most critical to murrelet conservation efforts because populations would continue to decline, primarily as a result of loss of nesting habitat. The plan predicted that while forest stands considered mature in 1992 may become suitable nesting habitat over the next 50 years, most young forests would probably not develop suitable habitat characteristics for 100-200 years. Some stands that were on the brink of being habitat in 1992 would have had 11 years' worth of growth and may have some of the habitat characteristics needed for nesting.

There are no data that truly quantify the net change in suitable habitat acreage that has occurred since the 1992 listing. However, the recent Supplemental Environmental Impact Statement (SEIS) on Survey and Manage Species (USFS and BLM 2004) provides an estimated rate of increase for late-successional forest of approximately 600,000 acres (242,812 ha)/decade for the entire area covered by the NWFP. Based on NWFP GIS data, approximately 40% of the late-successional reserves (LSR), Administratively Withdrawn Areas (AWA), and Congressionally Withdrawn Areas (CWA) currently protected by the NWFP are within the listed range of the murrelet. Thus, it is possible that up to 240,000 acres (97,125 ha) of forest old enough to achieve late-successional status according to USFS and BLM criteria (USFS and BLM 1994) developed between 1994 and 2003. However, this estimate does not reflect a similar gain in murrelet habitat because the definition of late-successional used by the USFS and BLM (1994) includes stands much too young (>80 years) for most forest types to have characteristics suitable for murrelet nest sites. There are also several other problems with the estimated gain of 600,000 acres (242,812 ha)/decade. First, there are no citations or analyses presented in the SEIS (USFS and BLM 2004) to determine how this value was derived and the validity of the methods used. In addition, there is no information on specific locations where this new late-successional forest might have developed. Clearly, more information is needed to accurately estimate the amount of potential murrelet habitat developed since 1992, but overall it is unlikely to have substantially increased.

The evaluation of murrelet habitat loss was largely based on 2 types of data provided by the Service: (1) information submitted to the Service (primarily from other Federal agency land managers) in response to the April and July 2003 request for information regarding known loss of habitat from management activities and other natural events in their local management areas, and (2) data from approximately 11 years of ESA Section 7 consultations that anticipated murrelet habitat removal and other impacts to the species. The following sections summarize the results of the analyses of these 2 datasets.

4.5.1.1 Losses of Suitable Habitat Summarized from Information Received During the Public Request

Information on murrelet habitat loss was available from 20 Federal agency land managers (USFS, BLM, and National Park Service) and 1 state agency. These agencies are responsible for the management of 2,089,248 acres (845,492 ha), or 94% of the 2,223,048 acres (899,639 ha) of suitable murrelet habitat (see Table 4.1-1). The habitat estimates were derived at the scale of an entire National Forest, BLM District, or National Park. One submission from the WDFW contained an estimate of habitat loss on private lands in Washington. These data were based on the submitter's interpretation of what constituted suitable habitat for murrelets. Data quality is variable, as some land managers have more refined information than others. For some lands, definitions of northern spotted owl habitat were used as a substitute for murrelet-specific information, which is likely a source of error. These estimates do not include losses due to consultations by the Service, which are summarized in the next section.

Habitat loss since 1992 was from 2 sources—timber harvest and natural events (i.e., wildfire, insect outbreak, and windthrow). Based on available data, the combined loss of suitable murrelet habitat over the 3-state area was estimated to be 22,398 acres (9,064 ha) with 5,364 acres (2,127 ha) resulting from timber harvest and 17,034 acres (6,893 ha) resulting from natural events (Table 4.5-2). Sixty-six percent of the entire regional loss of suitable habitat occurred from a single wildfire (the Biscuit fire, 2003) in southwestern Oregon. Habitat loss due to timber harvest accounted for 24% of the total loss with the majority (75%) of timber harvest occurring in Oregon, and lesser amounts in California and Washington.

Within individual administrative units that provided both habitat availability and loss data, the percent of suitable habitat loss ranged from 0 to 16.6%. (Table 4.5-2). Over the 3-state area, the percent loss of suitable habitat was estimated at 1.1%. The greatest proportional habitat loss occurred on the Siskiyou and Rogue River National Forests, primarily due to the Biscuit wildfire.

4.5.1.2 Suitable Habitat Losses Summarized from Consultations

- The Service conducted a second analysis to summarize information from ESA Section 7 and California Department of Fish and Game (CDFG) consultations on timber harvest plans from 1992 to 2003. Service field

Table 4.5-2. Loss of suitable murrelet habitat, 1992-2003 based on information from land managers.[1]

Cons. Zone	Ownership	Unit	Suitable Habitat (ac)[2]	Habitat Loss (ac)			% Loss	Data Source
				Harvest	Natural[3]	Total		
California								
4	Federal-USFS	Arcata NF	25,640	0	0	0	0.	BLM CA State Office submission (Acting State Director, pers. comm.)
4	Federal-USFS	Six Rivers NF	217,553	7	173	180	0.1	R5 FS submission updated by Q. Youngblood
4	Federal-USFS	Klamath NF	85,976	690	1,667	2,357	2.7	R5 FS submission
4	Federal-NPS	Redwood NP	41,280	0	0	0	0.0	NPS submission (H. Sakai, pers. comm.) - <1 ac of habitat degraded
5	Federal-NPS	Golden Gate NRA	269	0	0	0	0.0	NPS submission (H. Sakai, pers. comm.)
5	Federal-NPS	Muir Woods NM	343	0	0	0	0.0	NPS submission (H. Sakai, pers. comm.)
		California Total	**371,061**	**697**	**1,840**	**2,537**	**0.7**	
Oregon								
3	Federal-BLM	Salem	111,650	185	0	185	0.2	BLM Oregon State Office submission (E. Borg, pers. comm.)
3	Federal-BLM	Eugene	41,440	154	66	220	0.5	BLM Oregon State Office submission (E. Borg, pers. comm.)
3&4	Federal-BLM	Coos Bay	100,359	1,217	0	1,217	1.2	BLM Oregon State Office submission (E. Borg, pers. comm.)
3&4	Federal-BLM	Roseburg	90,928	1,089	0	1,089	1.2	BLM Oregon State Office submission (E. Borg, pers. comm.)
4	Federal-BLM	Medford	5,751	464	0	464	8.1	BLM Oregon State Office submission (E. Borg, pers. comm.)
3	Federal-USFS	Siuslaw NF	270,343	0	0	0	0.0	R6 FS submission (L. Goodman, pers. comm.)
4	Federal-USFS	Rogue/Siskiyou NFs	95,165	927	14,878	15,805	16.6	R6 FS submission (L. Goodman, pers. comm.)
		Oregon Total	**715,636**	**4,036**	**14,944**	**18,980**	**2.7**	
Washington								
1&2	Federal-USFS	Olympic NF	259,731	7.5	0	8	0.0	R6 FS submission (L. Goodman, pers. comm.)
1	Federal-USFS	Mt. Baker Snoqual. NF	227,314	0	250	250	0.1	R6 FS submission (L. Goodman, pers. comm.)
1	Federal-USFS	Gifford Pinchot NF	16,071	0	0	0	0.0	R6 FS submission (L. Goodman, pers. comm.)
1&2	Federal-NPS	Olympic NP	402,785	0	0	0	0.0	
1	Federal-NPS	San Juan NHP	39	0	0	0	0.0	
2	Federal-NPS	North Cascades NP	60,265	0	0	0	0.0	NPS submission (H. Sakai, pers. comm.)
1	Federal-NPS	Mt. Rainier NP	25,278	150	0	150	0.6	NPS submission (H. Sakai, pers. comm.) - 150 ac degraded by hazard tree removal
1&2	Private[4]	multiple lands	11,068	473	0	473	4.3	WDFW submission (J. Buchanan, pers. comm.)- 473 ac is combined harvest and blowdown
		Washington Total	**1,002,551**	**631**	**250**	**881**	**0.1**	
		Total	**2,089,248**	**5,364**	**17,034**	**22,398**	**1.1%**	

[1] *Data summarized by the Service.*
[2] *From Table 4.1-1.*
[3] *Natural loss includes fire and windstorm.*
[4] *Estimated from orthophotos, 1992-1998; includes only lands with orthophoto coverage, not all private lands.*

offices compiled available records on formal and informal consultations that resulted in removal, degradation, or disturbance of suitable murrelet habitat. Because Section 7 consultation is an interagency process, the consultation records primarily addresses projects on Federal lands and those projects on non-Federal lands where there is a Federal nexus. The information does not include loss or degradation of suitable habitat on State or private lands in Washington or Oregon, unless the removal was included in an HCP. Some information was available for private lands in California from CDFG and Service records on technical assistance to private landowners on timber harvest plans. Nonetheless, consultation records do provide an indication of how much suitable and occupied habitat has been threatened by timber harvest. This is particularly important information for non-Federal lands, particularly losses from ownerships covered by HCPs.

Although consultation records from 1992-2003 are thought to be relatively complete, it is possible that some are missing consultations; double-counting may have occur in a few cases as well. The quality and quantity of the information contained in consultations is variable, generally improving over time as understanding of the species increased. Implementation of the NWFP in 1994, the creation of critical habitat units (CHUs) in 1996, and the issuance of the Marbled Murrelet Recovery Plan in 1997 all increased the quality of information provided on murrelets during consultation. Since Conservation Zones did not exist prior to the 1997 Recovery Plan, analysis of habitat loss from consultation records could only be accomplished on a state level.

In general, the loss of suitable habitat summarized from consultation records presents a worst-case evaluation for the following reasons:

- Information on project implementation was largely unavailable, so it was impossible to validate whether the consulted-on habitat loss actually occurred as planned.

- Habitat loss authorized under HCPs may occur over a long period of time (e.g., several decades); however, under Section 7, this habitat is removed from the environmental baseline at the time the consultation document is issued.

- The estimates of habitat loss do not take into account the terms and conditions that minimize the take.

- The information regarding the quality of the affected habitat was not consistent among consultation documents; therefore, this analysis considers all habitat to be of equal value to the murrelet.

- In California and Oregon, some of the habitat impacts included in consultation records are outside of what is currently considered the range of the murrelet.

The following sections summarize: (1) loss of suitable habitat, (2) loss of occupied habitat, (3) degradation of habitat, and (4) loss of potential nest trees based on consultation records since 1992.

Loss of Suitable Habitat

Since 1992, the Service has consulted on the removal of at least 203,113 acres of suitable murrelet habitat throughout Washington, Oregon, and California (Table 4.5-3). An additional 5,933 acres (2,401 ha) were consulted on by CDFG as part of timber harvest plans in Conservation Zones 4, 5, and 6, resulting in a total of slightly more than 209,000 acres (84,580 ha) that were anticipated to be removed. Based on this analysis, acres covered by consultation or CDFG technical assistance for habitat removal were greatest in Washington and least in California. Approximately 62% of the total habitat covered by consultation in the 3-state area was accounted for by a single HCP in Washington (WDNR HCP). It is, however, important to emphasize that not all suitable habitat addressed by consultation has been removed to date, particularly the acreage included in HCPs that cover long time periods.

Loss of Occupied Habitat

Approximately 7,370 acres (2,983 ha) of suitable murrelet habitat anticipated for removal through Section 7 consultations were surveyed and found to be occupied by murrelets (Table 4.5-3). Another 92,505 acres (37,436 ha) of suitable murrelet habitat were either not surveyed or unknown to be surveyed (Table 4.5-3). Thus, the estimated anticipated loss of 7,370 acres (2,983 ha) of known occupied murrelet habitat should be considered a minimum estimate. Approximately half of the suitable habitat proposed for removal (109,172 acres [44,181 ha]) was surveyed and determined not occupied. In addition to the acres of occupied murrelet habitat authorized for removal, degradation was authorized for a small amount (approximately 40 acres [116 ha]) of known occupied habitat.

Of the estimated 7,370 acres (2,983 ha) of known occupied murrelet habitat anticipated for removal, most occurred on lands covered under HCPs (about 70%) and on Tribal lands (about 25%) (Table 4.5-3). Sixty-four percent (4,696 acres [1,900 ha]) of the total loss of known occupied habitat occurred in California as a result of the PALCO HCP (Table 4.5-3). In Washington,

Table 4.5-3. Acres of suitable marbled murrelet habitat anticipated to be removed between 1992 and August 20, 2003, based on Section 7 consultation and CDFG technical assistance on California Forest Practices permits records.[1,2]

Group	Known Occupied[3] (ac)	Not Occupied[3] (ac)	Not Surveyed (ac)	Unknown if Surveyed (ac)	Total Suitable (ac)
Washington					
NWFP Lands*	0	758	487	69	1,313
Tribal Lands	1,616	475	226	329	2,646
Other Federal Agencies/Lands**	0	38	34	2	74
Habitat Conservation Plans	413	53,715	77,173	0	131,301
Private or State Lands	0	0	40	0	40
Washington Total	2,029	54,986	77,960	400	**135,374**
Oregon					
Unknown Landowner	0	1,454	11	0	1,465
NWFP Lands	205	29,972	153	736	31,066
Tribal Lands	298	982	37	0	1,317
Other Federal Agencies/Lands	0	0	0	0	0
Habitat Conservation Plans	0	0	2,380	0	2,380
Private or State Lands	142	7,552	2,374	11	10,079
Oregon Total	645	39,960	4,955	747	**46,307**
California					
NWFP Lands	0	571	1,362	565	2,498
Tribal Lands	0	3,579	107	0	3,686
Other Federal Agencies/Lands	0	0	0	0	0
Habitat Conservation Plans	4,696	4,107	6,409	0	15,212
Private or State Lands (except CDFG)	0	36	0	0	36
Private Timber Harvest Plans (CDFG)	0	5,933	0	0	5,933
California Total	4,696	14,226	7,878	565	**27,365**
3-State Total	**7,370**	**109,172**	**90,793**	**1,712**	**209,046**

[1] *Service, unpublished data.*
[2] *Acres presented in table represent the area included in individual consultations and likely over-estimates actual habitat loss from each project.*
[3] *Occupancy determined from surveys.*
* *USFS and BLM lands covered by the NWFP.*
** *USFS and BLM lands not covered by the NWFP and NPS lands.*

approximately 2,000 acres (809 ha) of anticipated removal occurred as a result of consultations for timber harvest on Quinault Indian Nation lands and the City of Tacoma's Green River Watershed HCP (Table 4.5-3).

Of the more than 209,000 acres (84,580 ha) of suitable habitat in the 3-state area covered by consultation records for the murrelet, nearly half have not been surveyed for occupancy. Consequently, the estimate of total occupied habitat is uncertain. Nonetheless, known occupied habitat has continued to be lost since 1992. The additional habitat loss may result in reduced reproductive success, which is a major factor in the decline of the species (see Chapter 3.0 for discussion of demographics).

Degradation of Suitable Habitat

Information from the Section 7 consultations also provided an assessment of impacts to murrelet habitat where habitat was not entirely removed, but degraded in condition or function. Habitat degradation generally resulted from activities such as hazard tree removal, maintenance of recreation facilities (e.g., campgrounds and trails), and commercial thinning of forests.

These activities had the potential to reduce the quality of habitat or reduce nesting opportunities for murrelets.

Throughout the 3-state area, a total of 28,199 acres (11,412 ha) were consulted-on for activities that potentially degrade murrelet habitat. These acres are in addition to the 209,046 (84,598 ha) acres of suitable habitat loss. Virtually all of the degradation occurred on Federal or Tribal lands, and approximately 93% occurred on Federal lands in Oregon. On Federal lands in Oregon, anticipated degradation was primarily associated with commercial thinning operations around or adjacent to remnant trees with nest platforms.

Loss of Potential Nest Trees

Section 7 consultations throughout the 3-state area anticipated the removal of 10,537 potential nest trees and degradation of another 50 trees. All but 12 of these trees were removed or degraded in Oregon and Washington, and all but 28 were removed or degraded on USFS and BLM lands covered by the NWFP. Approximately 500 of the potential nest trees were surveyed and determined not occupied. A majority (9,800) of these trees were not surveyed prior to removal or degradation and were therefore considered as occupied in the Section 7 consultations. The survey status of 215 trees was unknown, and these trees were therefore considered occupied.

The majority of individual trees removed or degraded in Washington were a result of programmatic consultations with 2 National Forests and 1 National Park and were all related to hazard tree removal associated with campgrounds, trails, and roads. All but 1 tree removed or degraded in Oregon were a result of consultations with the USFS or BLM, and were generally associated with commercial thinning operations.

4.5.2 Effects of Land Ownership Patterns

Over the last 11 years, timber harvest levels have varied considerably by ownership. Similarly, the Section 7 and CDFG consultation record indicates that loss of suitable habitat was greatly influenced by land ownership (Table 4.5-3). Federal lands accounted for 34,951 acres (14,144 ha) (17%) of the total habitat anticipated for removal under Section 7 consultation. However, the estimate for Federal lands is known to be high for at least 2 reasons: (1) projects are often modified or dropped due to funding limitations or public participation; and (2) large areas in California and southwest Oregon, once considered in the range of murrelet, are now believed to be outside the inland distribution of murrelets, and those areas have only recently been excluded from consultations involving murrelets.

Non-Federal lands accounted for 168,162 acres (68,053 ha) (80%) of the total habitat anticipated for removal under Section 7 consultation. The largest single area of murrelet habitat loss was on ownerships covered by HCPs and accounted for 148,893 acres (60,255 ha) (71%). Because some of these HCPs are long-term plans, all the acres of habitat removal consulted on may not have been harvested at present, but it is anticipated that harvest of the permitted acres will occur in the future. Tribal lands accounted for 7,649 acres (3,095 ha) (4%), the fourth largest loss of habitat by ownership. A total of 5,933 acres (2,401 ha) of suitable murrelet habitat were estimated to have been released for timber harvest on private lands in California not covered by an HCP (Table 4.5-3). Since these lands were in private ownership and the timber value was high (many acres included old-growth redwood or Douglas-fir trees), it is likely that most, if not all, of these acres were actually harvested.

4.5.2.1 Lands in Federal Ownership

At the Federal level, the adoption of the NWFP and signing of the "Record of Decision for Amendments to Forest Service and Bureau of Land Management Planning Documents within the range of the North Spotted Owl" (USFS and BLM 1994) caused a dramatic decline in logging of older forests in the 3-state area. The goal of the Forest Ecosystem Management

Team (FEMAT) Plan (FEMAT 1993) was to conserve 89% of suitable murrelet nesting habitat on Federal land in the 3-state area. The Northwest Forest Plan eliminated a substantial source of habitat loss in the 3-state area.

The reduction in Federal timber harvest that occurred since 1994 is very evident in Washington State. In 1992, the harvest from national forests in western Washington was 256 million board feet (MBF); by 2001, the harvest on USFS lands had dropped 97% (WDNR 2004) (Figure 4.5-1). Similar to Washington, Oregon's statewide timber harvest fell almost 60%, from 8,615 MBF in 1988 to 3,542 MBF in 1998 (Gebert et al. 2002). Though harvest fell on all Oregon timberlands during this time, the largest effect came from the decline in harvest on Federal lands, which decreased more than 89%. In western Oregon, 13.5 million acres of Federal lands produced an average of 2,660 MBF per year in the 1980s, (Azuma et al. 2002). In the 1990s, that number dropped to an average of 190 MBF between 1995 and 1997 – this represents a 93% decrease. Between 1990 and 1993, the public lands in the 4-county North Coast region of California (Del Norte, Humboldt, Mendocino, and Sonoma) had annual timber harvests that averaged 48,865 MBF; in 1994, the harvest dropped to 8,491 MBF (Waddell and Bassett 1996). No data were readily available for the time since 1994, but the harvest on public land has remained at low levels.

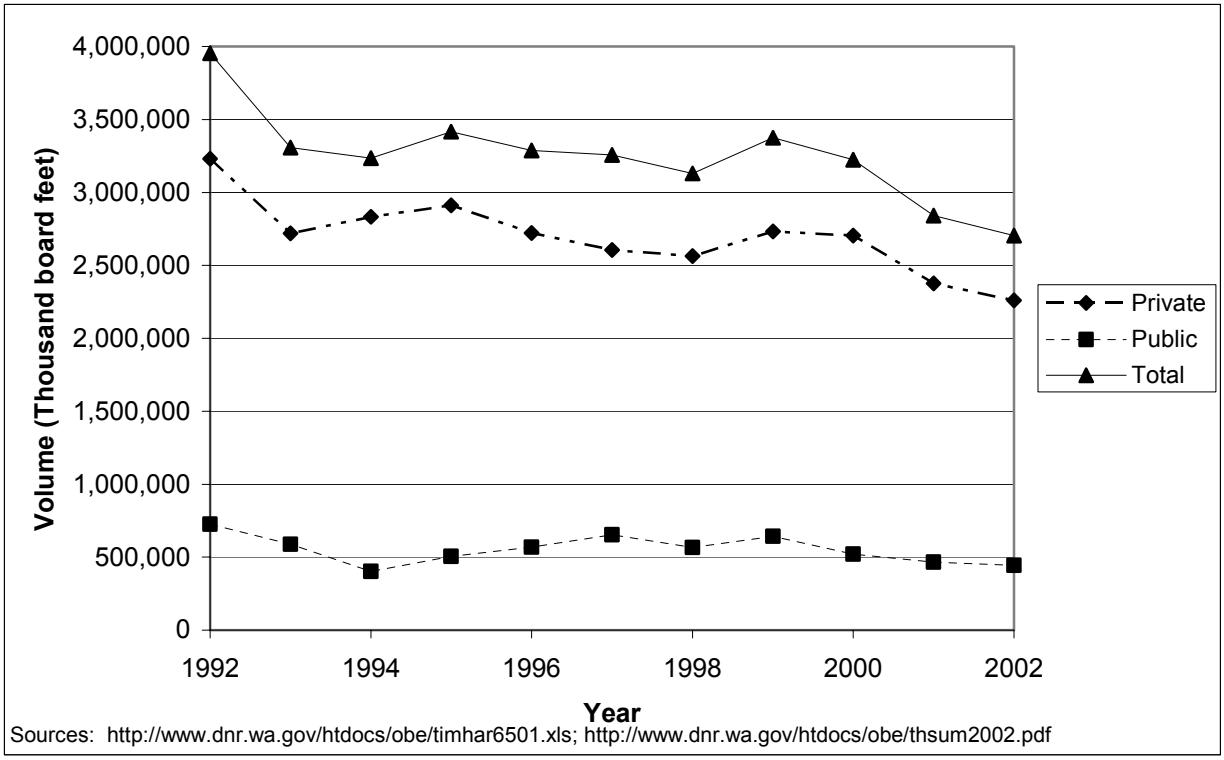


Figure 4.5-1. Recent trend in western Washington timber harvest.

4.5.2.2 Lands in Private and State Ownership

State regulations that address murrelets are not specifically discussed in this report. However, it is clear that the trend in habitat loss over the last 11 years has been affected by the State requirements. Occupied murrelet habitat on non-Federal land in California is protected under the California Forest Practice Rules and the California Endangered Species Act (CESA). There, each timber harvest plan is reviewed by CDFG biologists for impacts to murrelets. Landowners proposing harvest of suitable murrelet habitat must conduct surveys to demonstrate that the species is not occupying a stand prior to harvest.

In 1997, the WDNR Forest Practices Board adopted rules to protect murrelet habitat (occupied sites and non-surveyed habitat) on privately owned land within 50 miles of the coast that is not covered by an HCP. In Washington, State Environmental Policy Act (SEPA) checklists provide some assurance that actions on private land consider threatened species such as the murrelet. The Timber, Fish, and Wildlife (TFW) rules provide standards and guidelines for riparian habitats that likely protect some murrelet habitat. In Washington, harvest levels on State trust lands have remained relatively stable between 1992 and 2002, fluctuating between 274 and 573 MBF (WDNR website). In Oregon, little is currently required by State regulations in terms of murrelet habitat protection on private land. Between 1980 and 1991, annual harvest from private land in western Oregon averaged 2.78 MBF; between 1992 and 1997, the harvest averaged 2.71 MBF, a drop of 2.5% (Azuma et al. 2002).

A significant change that has occurred since 1994 is the adoption of HCPs on State and private lands. HCPs have set aside some occupied and suitable murrelet habitat. However, timber harvest allowed under HCPs has continued to eliminate habitat in each of the 3 states.

In Washington State, existing HCPs cover approximately 745,000 acres (301,492 ha) of private land (WFPA 2003). However, it is not known how much of this land is actually within the range of the murrelet or exactly how much murrelet habitat is being protected by the HCPs. The WDNR implemented an HCP that covers 1.6 million acres (647,500 ha) of State land in 5 planning units and an experimental forest that overlap the murrelet range. Using minimum structural characteristics of murrelet nest stands in Washington, the WDNR estimated that between 55,773 and 63,614 acres (22,571 and 25,744 ha) represent suitable murrelet habitat. In addition to the WDNR HCP, major HCPs in Washington that address murrelets include:

- City of Seattle's Cedar River Watershed
- City of Tacoma's Green River Watershed
- Simpson Lumber on the Olympic Peninsula
- Plum Creek HCP

Since 1990, there have been several land exchanges between private timber companies and the USFS in the state of Washington (WFPA 2003). Several of these land exchanges are within the range of the murrelet. In the long-term, the transfer of ownership may increase connectivity of older forests by reducing the "checkerboard" ownership pattern in some areas. However, in the short-term, the transfers are not likely to provide any additional murrelet habitat or reduce the overall risk of habitat loss from timber harvest. In many cases, the land transferred to the USFS is not old-growth, while any older forests now in private ownership will likely be subjected to timber harvest in the near future.

Unlike the state of Washington, where more than 131,300 acres (53,135 ha) of murrelet habitat have been consulted on by the Service as part of HCPs, only 2,380 acres (963 ha) were included in Oregon HCPs. The only Oregon HCP that addresses murrelet habitat is for the Elliott Forest.

In California, several State and Federal land acquisitions and private HCPs provide protection for murrelet habitat in northern California. These include the following:

- PALCO Headwaters Agreement of 1996, which established marbled murrelet conservation areas (MMCAs) encompassing 7,500 acres (includes 1,772 acres [717 ha] of uncut old-growth, and 2,733 acres [1,106 ha] of residual redwood) while allowing harvest of 6,909 acres (2,796 ha) of potential murrelet habitat.
- State purchase of habitat in the Owl Creek and Grizzly Creek MMCAs.

4.5.3 Future Habitat Trend

Future habitat trends for the murrelet are difficult to predict and dependent on the time scale of consideration. Future losses and gains in habitat are discussed below.

4.5.3.1 Potential Future Habitat Losses

In the near future, habitat loss is expected to continue, but it is uncertain at what rate. The Service continues to consult through Section 7 on adverse effects to murrelets from proposed actions and the removal of suitable habitat. Consultations with some non-Federal parties are ongoing in Washington and Oregon, and these involve the future removal of hundreds of acres of suitable murrelet habitat. Some of the habitat involved in these consultations is known to be occupied by murrelets. It is important to note that in Oregon private parties do not typically consult with the Service when harvesting suitable habitat unless there is a Federal nexus that requires a Biological Assessment.

The projected future of timber harvest in the 3-state area has not specifically been analyzed, although there are a number of studies for each state or region that can be used to describe general future trends in forestland habitat. Total area of timberland in the Pacific coast region of the United States is projected to decline from the 72.2 million acres (29.2 million ha) that were present in 1997 to 69.3 million acres (28 million ha) in the year 2050 (Haynes 2003), which represents a 4% decline. Kline and Alig (2001) predict that forestland in western Oregon and western Washington will decline 1% by the year 2050 due to conversion to urban land uses. Most of this conversion is expected to occur on lands that are close to existing cities, although several residential areas and destination resorts have recently been developed in forestland. The continued conversion of forestland will reduce the acreage of land that could develop into murrelet habitat in the long-term future.

In Oregon, future harvest projections for private forest lands not owned by timber companies begin at or above peak historical volumes (Adams et al. 2001). Currently most of these lands support second- and third-growth forest stands that do not represent suitable murrelet habitat. All projections suggest that these lands in western Oregon could maintain harvests near recent historic peaks for at least the next 50 years. Thus, private forest lands not owned by timber companies are not likely to contribute murrelet habitat in the future.

Under a recent court order, the BLM is currently evaluating the removal of all LSR designations on land they administer. The outcome of this evaluation could significantly reduce the acreage of late-successional forest. BLM lands account for 283,000 acres (114,527 ha) or 14% of the suitable murrelet habitat in the listed range and 38% in Oregon alone (see Table 4.1-1).

4.5.3.2 Potential Future Habitat Gains

A significant increase in the acreage of stands with old-growth characteristics necessary for murrelets is expected to take at least 100-200 years to develop (Service 1997). Development of suitable nesting habitat is generally dependent on trees attaining a size that supports large lateral branches. Manley (1999) reported that in British Columbia, only 4% of trees of all species <20 inches (51 cm) dbh had suitable nesting platforms; no Douglas-fir trees <20 inches (51 cm) dbh had suitable platforms. In general, Douglas-fir have to be >30 inches (76 cm) dbh before some large lateral branches begin to develop.

The nature and suitability of regenerated habitat from clearcut harvesting is not fully understood, but a delay in the onset of suitable platform development and structural complexity is expected due to high initial tree densities in planted stands. In the Oregon Coast Range, a study on the development of old-growth forests indicated that specific characteristics attributed to old-growth trees, such as large diameter branches and deep large crowns, are likely to develop much sooner at low initial stand densities (trees/acre) than at high stand densities (Tappeiner et al. 2003). The study indicates that density thinning of these stands may be useful in increasing diameter growth rates and producing habitat complexity and diversity more quickly in otherwise homogenous young forests. Evidence from other related studies investigating the effects of thinning on biodiversity (Muir et al. 2002) suggests that thinning in these forests also can speed the development of other older forest characteristics, including development of diverse understories of trees and shrubs and multiple canopy layers.

In addition to large trees, murrelets typically require limbs with a dense moss cover. The development of canopy bryophyte communities is strongly influenced by the age of the forest stand and the continuity of forest cover. In younger forest stands, light typically declines rapidly with distance into the canopy from the upper surface, due to the relatively dense spacing of individual trees. Low light conditions reduce bryophyte growth, whereas variable density thinning has been shown to significantly contribute to the development of epiphyte communities in 20 to 50 years (Neitlich and McCune 1997, Rolstad et al. 2001).

Most future gains of suitable murrelet nesting habitat are expected to occur on Federal lands. If the NWFP remains in effect and is not altered substantially from its current form, the projected acreage of USFS and BLM lands in the Pacific Northwest that support stands older than 200 years—a

lower limit of old-growth definitions—is expected to increase substantially by the year 2050 under various management scenarios (Mills and Zhou 2003).

Under the assumptions of the NWFP, the existing 1.1 million acres of late-successional forest in the Matrix and Adaptive Management Areas would be harvested over the next 40 to 50 years (USFS and BLM 2004). During this same 40- to 50-year timeframe, the BLM and USFS (2004) estimate that the overall amount of late-successional forest in the NWFP area will increase by 2.7 million acres (1,092,656 ha) (approximately 600,000 acres [242,812 ha] per decade) and reach a total of 10.7 million acres (4,330,154 ha). Based on NWFP GIS data, approximately 40% of the LSR, AWA, and CWA are within the listed range for the murrelet. Thus, if it is assumed that 40% of the 2.7 million acres (1,092,656 ha) of new late-successional forest is within the murrelet's range, it would mean that approximately 1 million additional acres (404,687 ha) of late-successional forest would be present 50 years from now. This assumes that late-successional forest has a generally equal probability to develop in the LSRs and withdrawn areas across the NWFP area. There are, however, several difficulties in basing the amount of late-successional forest and murrelet habitat development expected over the next 50 years on the rate of 600,000 acres (242,812 ha) provided by the USFS and BLM (2004). First, there are no citations or analyses presented by the USFS and BLM (2004) to determine how this value was derived and the validity of the methods used. Second, the NWFP defines late successional forests as >80 years old (USFS and BLM 1994), while stand and tree characteristics suitable for nesting murrelets can take 100-200 years to begin to develop, depending on forest type. Third, there is no information on specific locations where this new late-successional forest might develop. And finally, the NWFP may be revised from its current form.

The analysis of the NWFP acreage projections based on the USFS and BLM (2004) estimates generally corresponds with the trend in forest ages reported for National Forests in western Washington and Oregon compiled by Curtis et al. (1998). Using their age distribution data, between 170,000 and 240,000 acres (68,796 and 97,125 ha) of forest on National Forest land could attain 200 years of age during each of the 5 decades from 1998 to 2038 (Table 4.5-4). Thus, by the end of 50 years, all acreage in their 155-195 year classes – about 1.06 million acres (428,968 ha) in western Washington and Oregon – could attain 200 years. It is assumed that some of this forest will develop the conditions suitable for murrelets. However, there are also some problems with using Curtis et al. (1998) projections of old-growth forest development over the next 50 years to estimate increases in murrelet habitat. A large portion of National Forest lands in what Curtis et al. (1998) considered to be western Washington and Oregon is outside of the murrelet's inland range.

Table 4.5-4. Overall distribution of forest age classes in western Washington and Oregon on National Forest and industrial forestland, 1997.

10-year class mid-point	Approximate Acreage (1,000 ac.)
5	2,300
15	2,200
25	1,500
35	1,600
45	1,800
55	1,500
65	950
75	700
85	550
95	410
105	410
115	350
125	400
135	300
145	290
155	230
165	240
175	230
185	195
195	170
200+	1,350
Total	**17,675**

Source: Curtis et al. (1998).

This is especially true in Oregon where none of the Cascade Mountains are in the murrelet's range. In addition, the actual occupancy of the new habitat could vary greatly depending on numerous factors, such as distance from ocean, elevation, availability of canopy openings, size of habitat patch, colonization time, type of adjacent habitat and distance to nearest active nesting stands. And finally, old-growth characteristics necessary for murrelet nesting may take much longer to develop due to natural and man-made disturbances, tree density, and site-specific conditions, substantially reducing the acreage that actually represents suitable nesting habitat (USGS 2003). Future increases in suitable murrelet habitat, on private and State lands is even more difficult to project than on Federal lands. However, several of the larger HCPs have estimated future development of potential murrelet nesting habitat. The Simpson Lumber Company HCP (Washington) estimates the development of nearly 1,400 acres (567 ha) of potentially suitable murrelet habitat during the next 50 years within the 261,575-acre (105,856-ha) Plan Area (Simpson Timber Company 2000).

The Green River Watershed HCP (Washington) has the potential to promote the development of old-growth in riparian areas and late-successional forest conditions on approximately 8,349 acres (3,379 ha) (Tacoma Public Utilities 2001). Also in Washington, the Cedar River Watershed HCP, which covers 90,500 acres (36,624 ha), predicts the development of nearly 60,000 acres

(24,281 ha) of mature and late-successional forest. However, current murrelet use in all 3 of these Washington HCP plan areas is low to non-existent, so it is not certain if the increased acreage of suitable habitat would be occupied by murrelets. In addition, these areas may not be maintained in the long-term and therefore do not provide permanent habitat for murrelets. In California, the PALCO HCP predicts that an additional 2,535 acres (1,026 ha) of late-successional forests may develop suitable habitat structure in the next 50 years. There are also approximately 50,000 acres (20,234 ha) of second-growth in California Redwood National and State Park that could develop into murrelet nesting habitat sometime in the future.

Overall, it is difficult to estimate suitable the amount of newly available murrelet habitat in the near or distant future. Timber harvest will continue, and while the general trend is for acreage of late-successional forest on Federal lands to gradually increase, there are no projections that predict significant increases in murrelet habitat acreage with the necessary structural characteristics for nesting during the next several decades. The modeling and mapping efforts now underway as part of the Effectiveness Monitoring Plan for the NWFP (Madsen et al. 1999) may provide substantially better habitat projections for most Federal lands.

Regardless of the amount of suitable murrelet nesting habitat that may develop in the future, use of this new habitat may lag for a considerable period. The process of colonization is poorly understood, but high philopatry (see Section 2.9) likely leads to low rates of immigration, even in larger populations. Therefore, a large lag time may be expected between when new habitat becomes available and when use of this habitat occurs. Colonization probability, degree, and timing at individual sites will likely be dependent on proximity to occupied habitats, distance from ocean, population sizes, and other factors.

4.5.4 Effects of Wildfire, Windthrow, and Insect/Disease on Habitat Availability and Habitat Condition

Natural disturbances have the potential to affect the amount and quality of inland habitat for the marbled murrelet, particularly at the local and watershed scales. The effects of wildfire and windthrow result in immediate loss of habitat and may also influence the quality of adjacent habitat. The effects of insects and disease on habitat are typically slower but may by more pervasive.

4.5.4.1 Wildfire

Historically, wildfire was the primary large-scale natural source of disturbance in the conifer forests near the Pacific Ocean (Agee 1993). Other sources of natural disturbance included windstorms, vegetation removal by floods on rivers, forest diseases, and the rare volcanic eruptions. Fire effects varied with severity, vegetation community, and weather conditions. High-severity fires tended to eliminate all trees, while low-severity fires often left patches of trees and allowed a stand to retain the typical uneven characteristics of old-growth (Agee 1993). Natural fire frequency varied greatly throughout the murrelet's range in the 3-state area; an example fire frequency was 271 years along the Oregon Coast Range (Wimberly et al. 2000).

There has not been an extremely large fire in the Douglas-fir region of the Pacific Northwest since the 1940s (Curtis et al. 1998). The Yacolt Fire in 1902 and the Tillamook fire in 1933 each burned nearly 240,000 acres (97,125 ha) and were human caused (Tillamook County 2003; History Link 2003). The 2003 Biscuit fire in southern Oregon, however, is estimated to have eliminated nearly 15,000 acres of murrelet habitat (Table 4.5-2). The risk of large-scale wildfires may be reduced by modern fire detection and suppression techniques (ODFW 1995). Some changes in land use may reduce risks (e.g., highways, agriculture, urbanization), but others, such as residential expansion into forest areas, increase both the risks and the potential magnitude of damage. It is possible that global warming, combined with the long-term fire suppression on Federal lands, could result in a higher incidence of stand-replacing fires in the future, increasing the threat to murrelet habitat in the future.

4.5.4.2 Windthrow

Windstorms periodically eliminate potential murrelet habitat; most damage, however, is at a local scale. Windstorms probably have a positive effect on currently unfragmented habitat by creating small openings in the forest canopy, which have been shown to be important for murrelets. Habitat losses can be high in shelterwood stands, in partial cuts in dense stands that have developed without thinning, and along edges of fragmented stands (Curtis et al. 1998). There have been occasional large storms (e.g., 1962 Columbus Day) that have affected large portions of the westside forests (ODFW 1995). In general, Oregon is less susceptible to large-scale windstorms than Washington and California (ODFW 1995). On the Olympic Peninsula of Washington, at least 10 storms with hurricane-force winds have occurred in the last 200 years (Hamer and Meekins 1998). As

forest fragmentation increases from timber harvest, the threat of habitat loss due to windthrow is likely to increase.

4.5.4.3 Insect/Disease

Diseases and insect outbreaks represent a threat to murrelet habitat because they can kill complete stands of trees, either reducing existing suitable habitat or limiting the development of suitable habitat in the future. Insects and disease pathogens can cause significant tree mortality and growth loss, damaging large volumes of potential wood products each year. Forest pathogens can reduce management options for landowners and contribute to hazardous forest fire conditions (USFS 2002). Many forest diseases and insects cause only minor damage to trees and actually increase the availability of the tree deformities that may help create murrelet nesting platforms and beneficial canopy openings. However, forestland managers actively manage forests to reduce the impact of the major diseases and insect infestations that can kill significant numbers of conifers.

Root disease continues to be the most significant ongoing forest health problem in Washington (USFS et al. 2001). Laminated root rot is the most widespread and destructive root disease in the Oregon. Swiss needle cast and native root diseases such as black stain root disease, Armillaria root disease, and Annosus root disease cause considerable damage in certain areas, particularly in southern Oregon. Since the 1980s, Swiss needle cast has become increasingly evident in Douglas-fir forests of the Oregon Coast Range. In both Oregon and Washington, acreage affected by Swiss needle cast has increased over the last 5 years (USFS et al. 2001). The area from southern Washington to southern Oregon has shown increasingly severe damage from Swiss needle cast (Campbell et al. 2003). In 1999, about 500,000 acres were mapped as being infested with Swiss needle cast. Infection causes premature death and shedding of needles, resulting in reduced growth and occasional mortality if infections are repetitive or severe. Current management strategies in the Oregon Coast Range (e.g., Tillamook and Clatsop State Forests) rely on replacing Douglas-fir with other tree species such as hemlock, cedar, spruce, noble fir, and alder.

Two long-established non-native diseases, Port Orford cedar root disease and white pine blister rust, continue to cause extensive damage to Port-Orford cedar (*Chamaecyparis lawsoniana*) and five-needle pines, respectively. Production and planting of genetically resistant trees is offsetting some of the losses to these diseases. Port Orford Cedar root disease, caused by a fungal species (*Phytophthora lateralis*), is causing Federal agencies to alter harvest schedules of affected stands in southwest Oregon and northwest California.

It is not known how much of this habitat represents suitable murrelet habitat. Because spread of this fungus is limited to high risk sites, the annual new infestation rate varies substantially depending on presence of roads and streams that serve as vectors for the disease (USFS and BLM 2003).

4.5.5 Effects of Forest Management Practices on Murrelet Habitat

Habitat loss from timber harvest was one of the primary reasons for listing the marbled murrelet as threatened in 1992 (57 Federal Register [FR] 45328). In addition to the direct loss of old-growth forest from timber harvest, various silvicultural treatments also influence the quality of remaining habitat for murrelets. Studies on specific effects of various forest management practices on murrelets are limited, but there has been considerable recent research on effects on forest birds.

Compared to historical conditions, timber management has resulted in vastly different patterns of forest disturbances. Logging over the last century has typically used rotation cycles of 30 to 100 years, while fire-return intervals have been made substantially longer, resulting in a decreased proportion of older forest in managed landscapes. Clearcuts in western forests tend to be smaller and more evenly distributed than historical fires, leading to smaller old-growth and mature patch sizes and more edge effect (Wimberly 2000). Various silvicultural treatments have been shown to affect numerous species of forest-nesting birds through alteration of habitat structure, changes in predation rates, and competition for nest sites (e.g., Barber et al. 2001). Some of these, and their effects on murrelets, are described below.

- Wildlife Tree Retention - Retention of a small number of "wildlife trees" has long been used as a way to provide wildlife habitat in clearcuts. In general, wildlife trees surrounded by younger forest are not suitable for murrelets due to high nest predation (Manley 1999). In California, however, murrelets have nested in residual large redwoods that occur in a matrix of second-growth (Hunter and Bond 2001), but nests have not always been successful. Several private timber companies in northern California have implemented measures to retain large trees during clearcut harvests as a means of retaining possible nesting opportunities in the short term.

- Forest Edge Treatment – Edges of clearcuts can be abrupt or feathered. Feathering provides a more gradual transition to adjacent, older forest stands. There is little specific information on the effects of forest edge treatment on murrelets. In general, however, predation rates on forest

birds have been found to be higher at abrupt edges than at feathered edges (edges with partial harvests or a different forest type [e.g., Ratti and Reese 1988]). Explanations for differences in predation risk with edge type are likely related to differences in vegetation or nest concealment cover, use of the habitat by predators, or landscape context. A more open vegetation structure could allow greater visibility of open cup nests and thus facilitate higher predation from predators using visual cues. An increased risk of predation was found with horizontal visibility of forest bird nests along anthropogenic edges (Huhta et al. 1998). Similarly, abrupt edges may act as travel corridors for predators increasing the likelihood of incidental predation on forest bird nests (Ratti and Reese 1988, Song and Hannon 1999). In contrast, non-abrupt or feathered edges seem to provide more vegetative complexity, which may reduce the efficiency of some predators.

- Buffers - There are very little data on the tolerance of murrelet chicks to radiation and thermal stress that can result near forest edges. Kremsater and Bunnell (1999) reported that microclimate effects can extend up to 2 to 3 tree heights (328-492 feet [100-150 m]) into the forest. Establishment of forest buffers is the primary management action directed at reducing edge-related predation and environmental effects such as windthrow (Luginbuhl 2003). Current regulation-mandated buffers adjacent to suitable unsurveyed habitat average 300 ft (91 m) wide in Washington (Washington Forest Practices Board 2002). There are no data on the effectiveness of these buffers. Occupied habitat in California receives a 300-ft (91-m) buffer.

- Salvage Logging - Salvage logging operations in recently burnt areas have the potential to reduce future suitable murrelet habitat at relatively large scales if living trees are removed. The recent wildfires (e.g., the Biscuit fire of 2003 in southern Oregon) has led to the recent signing of the "Healthy Forests Initiative" by the Executive Branch of the Federal government that may increase "fuel-reduction" timber harvests in many areas of the west. This may further reduce the availability of murrelet nesting habitat.

- Selective Harvest Practices - Curtis et al. (1998) summarized silvicultural measures that can be used to increase structural diversity in Pacific Northwest forests, hastening the development of suitable murrelet habitat. These measures include using small (<10 ac [4 ha]) clearcuts, shelterwood, advance regeneration, group selection, green tree retention, and long harvest rotation. In addition to retention of old-growth, land managers can implement silvicultural systems such as small group

selection or selective logging with variable retention in advanced second-growth to facilitate development of suitable murrelet habitat in the future (Manley 1999). Several studies have recommended heavy thinning in young Douglas-fir stands to accelerate the development of late-successional attributes (Garman et al. 2003, McComb et al. 1993, Tappeiner et al. 1997). Regimes that thin heavily at ages 40 and 60 reduce species diversity and vertical structure of the initial stand, but these attributes quickly recover due to the rapid understory regeneration (Garman et al. 2003). Thinning and regulation of overstory density can be used to produce large trees quickly, develop stand structure, and generally aid the development of mature forest and old-growth-like characteristics (Curtis and Marshall 1993, Newton and Cole 1987). Advance regeneration can be released to produce multiple layers. However, the rate of at which murrelet habitat characteristics in younger aged stands could be developed using these methods is unknown. Thus, there is still the risk that loss of habitat will continue to be a threat even with improved silvicultural techniques.

4.5.6 Fragmentation and Edge Effects in Forest Habitat

Forest fragmentation occurs when contiguous areas of forested habitat are reduced to small, isolated fragments through logging or other forms of habitat modification. This results in a spatial and temporal mosaic of forest patches with quantitative and qualitative effects, including:

- Reductions in the amount and heterogeneous nature of habitat;
- Changes in the context and configuration of forest stands;
- An increase in the number of small habitat patches;
- Reduced forest patch sizes;
- Reductions in the amount of interior or core habitat;
- An increase in the amount of edge;
- Isolation of the remaining habitat patches;
- Creation of "sink" habitats; and
- The introduction of a myriad of edge effects related to patch size and isolation (e.g., Andren 1995, Murcia 1995).

The short and long-term ecological consequences of these changes to wildlife can include effects on population viability and size, local or regional extinctions, displacement, fewer nesting attempts, failure to breed, reduced fecundity, reduced number of nests, lower nest success, increased predation and parasitism rates, crowding in remaining patches, and reductions in adult survival (e.g., Andren 1995, Raphael et al. 2002b).

A large volume of literature from the eastern United States and Europe documents the effects of forest fragmentation on birds in agricultural or suburban landscapes (e.g., Temple and Cary 1988, Andren 1992, Paton 1994, Robinson et al. 1995, Soderstrom et al. 1998, Heske et al. 1999). Many of these studies found that the abundance of forest interior species declined with increasing isolation and decreasing size of forest patches. In addition, forest/field edges and small forest fragments were generally characterized by lower nesting success (but see Murcia 1995). In contrast, in western forested landscapes, the effects of forest fragmentation on species abundance and predation tend to vary with location, the type of predators and their habitat associations, the level of fragmentation, the quality of forest edge and cover, the associated matrix of habitats (landscape context), scale, and study design (e.g., Ratti and Reese 1988, Hannon and Cotterill 1998, Sieving and Willson 1998, Tewksbury et al. 1998, Song and Hannon 1999, Marzluff et al. 2000, De Santo and Willson 2001, Raphael et al. 2002b).

Despite the variation in recent research results from the western United States, some patterns are beginning to emerge:

- Higher nest predation was documented in areas with a high abundance of nest predators (e.g., Ratti and Reese 1988, Sieving and Willson 1998, Tewksbury et al. 1998, Song and Hannon 1999, De Santo and Willson 2001);

- The abundance or diversity of predators increased with the variety and complexity of habitats (Andren 1995, Marzluff et al. 2000, Raphael et al. 2002b);

- Abundance of some corvid species increased along edges or in forest fragments near human activities (e.g., Marzluff et al. 2000, De Santo and Willson 2001, Masselink 2001, Raphael et al. 2002b);

- High nest predation by corvids occurred along edges near human activities or in areas of low forest cover (Hannon and Cotterill 1998, Sieving and Willson 1998, De Santo and Willson 2001, Raphael et al. 2002b); and

- High predation by small mammals occurred in a variety of habitats including interior forests and along anthropogenic edges (Ratti and Reese 1988, Hannon and Cotterill 1998, Sieving and Willson 1998, Tewksbury et al. 1998, De Santo and Willson 2001).

Additional research in the western United States, with a concerted effort to develop research designs that can be replicated across habitat types with large sample sizes, will further elucidate the relationships between birds, fragmentation, edge, species abundance, and risk of predation.

The effects of forest fragmentation on marbled murrelets are just beginning to be explored, and additional research will be required to more clearly determine the extent and impacts of the effects of fragmentation and edge on murrelet populations. However, the amount of old-growth forests in Washington, Oregon, California, and some areas of British Columbia has been reduced from historic levels by more than 80% (Service 1997, Burger 2002), leaving murrelets with small, isolated stands of older trees for nesting. Murrelets are thought to be highly sensitive to forest fragmentation (Hansen and Urban 1992), and changes in their distribution and abundance have occurred in association with habitat loss and forest fragmentation (Service 1997). For example, murrelets no longer occur in areas without suitable forested habitat (e.g., Marin County in California) (Service 1997), and they appear to abandon some highly fragmented areas over time (areas highly fragmented before the late 1980s generally did not support murrelets by the early 1990s) (Meyer et al. 2002). In addition, detections of murrelets at inland sites and densities offshore were found to be higher in or adjacent to areas with large patches of significant old-growth, and in areas of low fragmentation and isolation of old-growth forest patches (Raphael et al. 1995, 2002a,b; Burger 2002; Meyer and Miller 2002; Meyer et al. 2002; Miller et al. 2002). Similarly, murrelet densities (determined with radar) increased with increasing amounts of suitable habitat, core-area old-growth, and proximity of patches within specific watersheds (Burger 2002, Raphael et al. 2002a). Smaller patch size and reduced core area could also have an effect on the number of murrelet nests and on nest success (Burger 2002, Raphael et al. 2002a).

Marzluff and Restani (1999) and Raphael et al. (2002b) reviewed the potential impacts of forest fragmentation on marbled murrelets. They suggested that reduced amount of nesting habitat would have long-term impacts on the number of nests and short-term impacts on nest success, both of which would affect population size. As with other alcids, adult marbled murrelets are believed to have high site fidelity and return to the same nest site in successive years (Nelson and Peck 1995, Nelson 1997). If their nest site is

lost to logging or development, it is suspected that some murrelets will lose breeding opportunities in successive years or not breed again, and others may be displaced (if possible) to nearby or disjunct suitable or marginal habitat (Divoky and Horton 1995, Service 1997). If murrelets are forced to move into marginal habitat, nesting success could decline over time, leading to low nesting density and small populations (Raphael et al. 2002b). The fecundity rates of remaining pairs could also decline. Because marbled murrelets nest solitarily, successful nesting is dependent upon the murrelets' ability to remain hidden at the nest site to avoid predation. A cryptic plumage (both adults and juveniles) and secretive behaviors, such as limiting activity primarily to the low light levels of dawn and dusk, visiting the nest infrequently during chick rearing, and minimizing loud vocalizations from the nest, decrease their chances of being discovered by predators (Nelson and Hamer 1995, Nelson and Peck 1995, Nelson 1997). As suitable nesting habitat becomes scarce and more isolated, and predator populations increase (see below), remaining hidden at the nest may become increasingly more difficult.

Smaller patch size, associated with reduced interior habitat and increased edge, will likely affect murrelet nest success and the number of nests in both the short and long term, which will ultimately have long-term consequences for population size (Raphael et al. 2002b). The increase in isolation of patches will also affect nest success (short term) and number of nests (long term), and ultimately population size. In addition, adult murrelets may be at greater risk while commuting to and from isolated forest patches. Miller et al. (2002) demonstrated that suitable forest patches >5 km from known nesting areas were less likely to be occupied by murrelets, and no occupied patches were more than 11 km from other occupied sites.

Murrelets are thought to be highly vulnerable to increased levels of nest predation associated with forest edges (Service 1997). While the extent of the effects of fragmentation and edge on murrelet nest success is not known (see below), predation has consistently been the most significant cause of nest failure at marbled murrelet nests. Most active murrelet nests have failed (43-85%), and most failures have resulted from predation (78%, 29 of 37 nests with known outcomes) (Table 4.5-5) (Nelson and Hamer 1995, Hamer and Meekins 1999, Manley 1999, Manley and Nelson 1999, Bradley 2002, Hebert and Golightly 2003, Nelson and Wilson 2002, Manley 2003, Peery et al. in prep.). Recent radio telemetry research found failure rates of 54% in British Columbia (Bradley 2002), 68-86% in northern California (Hebert and Golightly 2003), and 84-100% in central California (Peery et al. in prep.). In a study of artificial nests in Washington and Oregon, 81-86% were disturbed or depredated (Marzluff et al. 1999, Luginbuhl et al. 2001). Known predators

Table 4.5-5. Number of successful and failed murrelet nests by state and province[1].

Location	Number of Successful Nests	Number of Failed Nests			Total Nests	Nest Failure Rate
		Eggs Lost	Chicks Lost	Total Failed Nests		
British Columbia 2	20	17	6	23	43	53%
Washington 3	4	1	2	3	7	42%
Oregon 4	9	4	8	12	21	57%
California 5	8	32	13	45	53	85%
Total	**41**	**54**	**29**	**83**	**124**	**67%**

[1] Includes only nests with known outcomes and known stage of failure.
[2] Sources: Manley 1999, Bradley 2002; Jones, unpubl.
[3] Sources: Sources: Hamer and Meekins 1999, unpubl.; Washington Department of Fish and Wildlife, unpubl.
[4] Sources: Hamer and Nelson 1995, Nelson and Wilson 2002, unpubl.
[5] Sources: Singer et al. 1995, Singer et al. 1997, Hebert and Golightly 2003, Peery et al. in prep.; Burkett unpubl.

of active and artificial nests are listed in Section 2.10. The effects of edge and fragmentation on murrelet nest success, predator populations, and predation risk are summarized below.

4.5.6.1 Placement of Nests Relative to Edges

Marbled murrelets are known to locate their nests throughout forest stands and fragments, including along various types of natural and man-made edges (Hamer and Meekins 1999, Manley 1999, Bradley 2002, Burger 2002, Nelson and Wilson 2002, Nelson et al. 2003). Nests on natural edges occur along streams, wetlands, forest gaps, large natural openings, or avalanche chutes, whereas nests on anthropogenic edges are adjacent to clearcuts, roads, or regenerating forest. The number of nests along natural or man-made edges and within the forest interior is summarized in Table 4.5-6. Edge nests were considered to be within 164 feet (50 m) of the forest edge.

Most of the nests occurred along edges (76%), but in most cases these were natural edges (59%). Edge nests are more readily discovered through ground

Table 4.5-6. Number of edge and interior nests by state and province.

Location	Number of Interior Nests	Number of Edge Nests1			Total Nests
		Natural Edges	Man-made Edges	Total Edge Nests	
British Columbia [2]	38	43	23	66	104
Washington [3]	3	12	11	23	26
Oregon [4]	5	30	20	50	55
California [5]	2	1	5	6	8
Total	**48**	**86**	**59**	**145**	**193**

[1]Defined as within 50 m of an edge.
[2]Sources: Jordan et al. 1997, Bradley 2002, Conroy et al. 2002.
[3]Sources: Hamer and Meekins 1999, unpubl.; WDFW, unpubl.
[4]Sources: Manley and Nelson 1999, Nelson and Wilson 2002, Nelson et al. 2003, unpubl.
[5]Sources: Singer et al. 1995, Singer et al. 1997, Burkett unpubl.

surveys or climbing trees in areas of murrelet activity, whereas nests found using random tree climbing plots or radio telemetry are likely less biased. When considering only the nests found by random tree climbing and radio telemetry, the results were similar; most nests were located on edges (76% of 152 nests), and the most common type of edge was natural (69% of 115 nests) (Hamer and Meekins 1999, Bradley 2002, Nelson and Wilson 2002, Nelson et al. 2003). Nests on natural edges occur along streams, wetlands, forest gaps, large natural openings, or avalanche chutes, Nelson et al. 2003). Bradley (2002) found that murrelets often nest within 33-66 feet (10 to 20 m) of natural edges, but the range was more highly variable in other studies (6.6-2,297 feet [2-700 m]) (Hamer and Nelson 1995, Jordan et al. 1997, Manley and Wilson 1999, Conroy et al. 2002, Nelson and Wilson 2002). Distances to anthropogenic edges ranged from 9.8-1,125 feet (3 to 343 m) in these studies.

As Burger (2002) points out, there are no data on the proportions of available forest area relative to forest edge; therefore, we do not know whether murrelets are selecting for edge habitat. The frequency of murrelet nests on edges may be solely a reflection of the prevalence of forest fragmentation and natural edges. However, murrelets have high wing loading and thus have limited maneuverability during aerial flight. Edges may provide murrelets easier access at their nests, both for adults during visits to and from the nest, and for chicks during fledging.

The replacement of native forest with small, isolated patches and abundant edge can create changes in microclimate, vegetation species, predator-prey dynamics, and other edge effects. Unfragmented, older-aged forests have lower temperatures and solar radiation and higher humidity compared to clearcuts and other open areas (e.g., Chen et al. 1993, 1995). Edge habitat is also exposed to increased temperatures and light, high evaporative heat loss, increased wind, and decreased moisture. Fundamental changes in the microclimate of a stand have been recorded at least as far as 787 feet (240 m) from the forest edge (Chen et al. 1995). The changes in microclimate regimes with forest fragmentation could stress a cold-water adapted seabird (Meyer and Miller 2002) and affect the distribution of epiphytes that murrelets use for nesting. While there are little data on the effects of radiation and thermal stress on murrelets, both chick and adults have been observed panting when exposed to direct sunlight (Binford et al. 1975; Nelson, unpubl.). Exposure to increased temperatures could cause heat stress for adults and chicks and eventually cause adults to abandon the site. Likewise, increased winds at the forest edge could trigger cold stress. The effects, however, would vary with aspect, slope, elevation, and topography. Marbled murrelets do not build nests. Instead, they utilize pre-existing structural tree branch formations and lay their single egg on epiphytes or

other substrate covering a large tree limb. Branch epiphytes or substrate have been identified as a key component of murrelet nests (Nelson et al. 2002, 2003, Chapter 4 this volume). The substrate is important for insulating the egg and protecting it from falling. While there are no data on the specific effects of microclimate changes on the availability of murrelet nesting habitat at the branch and tree scales, the penetration of solar radiation, wind, and warm temperatures into the forest could change the distribution of epiphytes or blow moss off nesting platforms. In some portions of their range, murrelets may not nest in areas that lack epiphytes, such as along forest edges or in areas of extreme temperature (Hunter et al. 1998).

4.5.6.2 Nest Success and Distance to Edge

The effects of distance to edge habitat are equivocal, with some studies showing no difference in predation rates (Tewksbury et al. 1998, Rodewald 2002) and others showing an effect primarily within 164 feet (50 m) (Paton 1994, Nelson and Hamer 1995, Manley and Nelson 1999), but this relationship sometimes varied with the structure of the adjacent forest and proximity to human activity (e.g., Raphael et al. 2002b). Type of edge (artificial vs. natural, abrupt vs. feathered, suburban vs. forested) also created varied results, but in general predation rates were higher at abrupt edges than at feathered edges (edges with partial harvests or a different forest type) (e.g., Ratti and Reese 1988) and suburban edges than forested or natural edges (e.g., De Santo and Willson 2001, Raphael et al. 2002b).

Early research on the success of marbled murrelet nests with respect to edge showed that successful nests were significantly farther for forest edges than failed nests (Nelson and Hamer 1995). More recent research demonstrates somewhat mixed results. At active murrelet nests in British Columbia, Washington, Oregon, and California, successful nests were again found to be farther from forest edges (x=463 feet [141 m]) than failed nests (x=184 feet [56 m]) (Manley and Nelson 1999). Nest success was 38% (n=29 nests) within 164 feet (50 m) of the forest edge and 55% (n=29 nests) at distances greater than 164 feet (50 m) from the edge, but the differences were not statistically different (χ^2=4.55, P > 0.05). Most of these nests failed because of predation (60%), and predation was higher within 164 feet (50 m) of an edge than within the forest interior. No murrelet nests >492 feet (150 m) from the edge failed because of predation. In addition, successful nests were in areas with more cover at the micro-site scale. Some evidence suggested that nests in larger stands (>741 acres [300 ha]) were more successful, but successful nests have occurred in stands of all sizes. In contrast to these studies, Bradley (2002) found no relationship between nesting near forest edges (at 50 and 100 m) and nest success in Desolation Sound, British

Columbia (n=37 nests). However, few of his active nests were found near anthropogenic edges (n=2); therefore, conclusions about the data were difficult to make.

Table 4.5-7 shows nest success of active nests in relation to location (edge and interior) and type of edge (artificial vs. natural). From the limited research projects with data on nest location and edge type, a similar number of successful and failed nests were found at edge (within 164 feet [50 m]) and interior sites. In addition, there was no difference in the number of successful and failed nests on artificial or natural edges. However, in a larger scale GIS analysis looking at nest success within 656 feet (200 m) of an edge, Bradley (2002) found that nesting success differed significantly between sites adjacent to natural, artificial, and interior nests (n=98 nests). A significantly higher nest success at nests near natural edges (79%, n=42) compared to nests in the interior (48%, n=33) explained most of this difference. Nest success near artificial edges was 61% (n=23). However, he found no significant differences in individual comparisons of nests adjacent to artificial and natural edges, or at artificial edges and within the forest interior. From these results, Bradley (2002) was unable to determine the effect of natural and artificial edges on murrelet nesting success.

Research at artificial murrelet nests has been conducted in Oregon and Washington (Marzluff 1995; Marzluff and Raphael 2001; Marzluff et al. 1999, 2000; Raphael et al. 2002a; Luginbuhl 2003). Survival of simulated nests varied little among habitat types (old-growth, mature), structural complexity (simple, complex), and fragmentation levels (contiguous, fragmented), although proximity to human activity and structural complexity influenced rates of predation. The highest survival was in simple structured mature

Table 4.5-7. Nest success of active murrelet nests in relation to forest edge (within 50 m) and edge type by state and province.

Location	# Edge Nests		# Interior Nests		# on Artificial Edges		# on Natural Edges	
	Successful	Failed	Successful	Failed	Successful	Failed	Successful	Failed
British Columbia [1]	9	11	7	10	2	0	7	11
Washington [2]	4	2	0	0	3	2	1	0
Oregon [3]	6	11	3	1	3	10	3	1 [5]
California [4]	2	4	1	1	2	3	0	1 [6]
Total	**21**	**28**	**11**	**12**	**10**	**15**	**11**	**13**

[1] *Source: Bradly 2002.*
[2] *Sources: Hamer unpubl., WDFW, unpubl.*
[3] *Sources: Nelson and Wilson 2002, unpubl.*
[4] *Sources: Singer et al. 1995, Singer et al. 1997, Burkett unpubl.*
[5] *This nest was along a river, but a paved road was just on the other side of the river from the nest.*
[6] *This nest was along a river, but a campground was located directly on the other side of the river from the nest.*

forests, but only when unfragmented and near human activity or fragmented and far from human activity. The lowest survival was in simple, mature forests near human activities or in old-growth forests within 1 km of human activity. However, predation was highest within 164 feet (50 m) of the forest edge, especially adjacent to human activity. These edge effects were slightly more pronounced at nests containing simulated chicks than those with artificial eggs, with chick nests <656 feet (200 m) from an edge having significantly higher predation rates.

The results from these studies demonstrate that the relationship between fragmentation, edge effects, and rates of predation at murrelet nests is not completely known. While there appears to be an edge effect (within 164-656 feet [50-200 m]), especially in proximity to human activities, we need to determine how the suite of nest predators present, their habitat associations, and foraging efficiency in different habitats affect murrelet nest success. In addition, the behavior of nesting murrelets in response to predator populations, edge habitat, and human activities needs exploration. Additional research will be needed to further elucidate the complex interactions that could affect murrelet nest success in fragmented landscapes and edge habitat.

4.5.6.3 Nest Success in Relation to Slope, Aspect, and Elevation

Information on nest success with respect to elevation is limited. Burger (2002) reviewed information on the distribution of select predators (raptors, owls, corvids) with respect to elevation in British Columbia. Raptors and owls were found to be less abundant at higher elevations, but corvid distribution was more complex. Bradley (2002) found lower predator numbers at high elevations in Desolation Sound, but Steller's jays and small mammals were abundant at all elevations. Predator densities, however, were 3 times higher within 0.62 mile (1 km) of the ocean than farther inland, suggesting that the presence of potential murrelet nest predators is much more likely at low elevations. Burger et al. (2000b) also documented higher predator numbers at low elevation coastal sites. Perhaps in association with these results, murrelet nesting success was higher at higher elevations (and on steep slopes, see below) (Huettmann 2003a,b). Additional research that specifically addresses the risk of predation and the distribution of predators at various elevations will help determine the specific effects of elevation on murrelet nest success.

4.5.6.4 Forest Fragmentation and Edge Effects on Predator Populations

The alteration of forests and landscapes through fragmentation and other disturbances can change the composition and abundance of species and create new interactions. Opportunistic species have the ability to colonize these new habitats, and movement of opportunistic predators into these newly created habitats could increase rates of predation on nearby forest-nesting species. The significance of forest fragmentation and edge effects on nest predation, however, appears to be dependent on the suite of nest predators in the area and their abundance, edge type, the adjacent matrix of habitats, proximity to human activities, parental behavior (vigilance), nest density, and vulnerability of nests, based on type of nest and type of vegetation influencing nest concealment.

Predator species respond differently to edges, forest fragmentation, and scale. For example, the abundance and predation rates of mammalian predators (squirrels and mice) are not necessarily correlated with edge or patch size (e.g., Hannon and Cotterill 1998, DeSanto and Willson 2001), and they may be more affected by small-scale characteristics, such as food availability, within their small home ranges (Song and Hannon 1999). In contrast, the abundance and predation rates of avian predators, especially some species of corvids, can be affected by edge and areas of low forest cover (Hannon and Cotterill 1998, Tewksbury et al. 1998), and they may be more influenced by landscape features such as type of matrix habitat and proximity to human disturbance (Raphael et al. 2002b). The risk of predation by these species depends on their distribution and abundance within the forested landscape.

Corvids have been implicated as the primary predator of murrelet nests (Nelson and Hamer 1995, Raphael et al. 2002b). Corvids are extremely intelligent and efficient predators, using a highly refined search image for locating prey. They are visual predators that are capable of specializing on nests (Andren 1992). Corvid numbers have increased significantly throughout the west in the last century in response to habitat change and human development (Marzluff et al. 1994). The densities of many corvid species, and predators in general, have been found to be higher in landscapes fragmented by agriculture and human development than forested landscapes (Andren 1992, Rodewald 2002, Marzluff and Restani 1999, but see Tewksbury et al. 1998). However, within forested landscapes, their habitat relationships vary with landscape contiguity, proximity to humans, and stand structure. American and northwestern crows occur more frequently in logged or coastal sites (fragmented) than unlogged areas and are more abundant in areas close to human development (Neatherlin 2002, Neatherlin

and Marzluff 2002, Burger 2002, Marzluff and Neatherlin in prep., Rodway and Regehr 2002, Verbeek and Caffrey 2002, but see Schieck et al. 1995). Gray jays, on the other hand, are most abundant in continuous landscapes (Raphael et al. 2002b) and may not always be affected by human activity. Common ravens appear to be equally abundant in all landscapes (Burger 2002, Rodeway and Regehr 2002), probably because of their large home range size, but their relationship to human settlement and recreation varies from significant increases in some urban areas (Boarman and Heinrich 1999) to rarely visiting anthropogenic sites (Marzluff and Neatherin in prep.).

In general, Steller's jays occur more often in fragmented landscapes and are more abundant at stand edges compared with the interior forest (Brand and George 2001, De Santo and Willson 2001, Marzluff et al. 2001, Masselink 2001, Burger 2002, Rodway and Regehr 2002). For example, Masselink (2001) found Steller's jays to be most abundant along artificial edges within 164 feet (50 m) on either side of the forest-clearcut boundary. These jays also commonly forage within the forest canopy, indicating a high risk to murrelets nesting within 164 feet (50 m) of the forest edge. However, Steller's jays are found in a variety of habitat types, and their densities are not solely affected by the presence of edges, stand size, or landscape patterns (e.g., Lehmkuhl et al. 1991, Schieck et al. 1995, De Santo and Willson 2001, Raphael et al. 2002b). For example, while De Santo and Willson (2001) found jays more abundant along edges than within the forest, and higher densities occurred adjacent to human development where they may have been attracted to supplemental food at feeders. Steller's jays were also common along natural feathered edges of wetlands, which provided small and widely spaced trees for perching. In addition, Marzluff et al. (2001) found that jays preferentially used fragmented habitats and forest edges, especially if associated with human settlements and campgrounds. In contrast, Raphael et al. (2002b) determined that Steller's jays were affected by both fragmentation and proximity to human activity, and their abundance was highest in fragments (vs. continuous forest) far from human activity. Neatherlin (2002) also found that Steller's jays were unresponsive to human settlement and recreation, and did not expand their home ranges with forays to anthropogenic sites.

Landscape context or the type of adjacent matrix habitat appears to have a large impact on predator abundance. For example, Steller's jays in Oregon and Washington were found to be in highest abundance along forest edges bordered by early seral vegetation with berry-producing shrubs (Marzluff et al. 1999, 2000, Luginbuhl et al. 2001, Raphael et al. 2002b). Remote fragments surrounded by clearcuts with berry-producing plants also had high predation rates (Marzluff and Restani 1999). In contrast, simple structured

forests had the lowest corvid populations and lower predation rates (Marzluff et al. 1999, 2000, Raphael et al. 2002b). In addition, nest predation rates were not elevated when remote forest fragments were surrounded by regenerating forest (Raphael et al. 2002b, Ripple et al. 2003). Predators may use complex habitats and landscapes more often than simple (younger) habitats or homogeneous landscapes because of the differences in vegetative structure and food abundance. Younger forests and simple landscapes have fewer microhabitats for foraging, whereas complex stands and varied landscapes (such as farmland, suburbs, and forest/clearcut edges) provide a wider range of food sources and more opportunities for foraging (Huhta et al. 1998, De Santo and Willson 2001, Marzluff et al. 2000, Raphael et al. 2002b). A higher abundance of nest predators in these productive landscapes has often led to higher rates of nest predation (e.g., De Santo and Willson 2001).

The risk of predation on marbled murrelet nests from increased predator numbers and landscape fragmentation is not completely known. However, the risk of predation by avian predators appears to be highest in complex structured landscapes in proximity to edges and human activity, where many of the corvid species are in high abundance. The risk of predation by mammalian predators, on the other hand, may be equally high at forest edges and in the forest interior, wherever mice and squirrels are most abundant. From several artificial nests studies in Oregon and Washington, we know that the risk of predation by jays increased with jay abundance (Neatherlin 2002). In addition, Luginbuhl et al. (2001) found that the rate of artificial nest predation in contiguous and complex mature and old-growth forest landscapes was positively and significantly correlated with corvid abundance (but not at the plot scale). Overall, artificial nest survival decreased with increasing structural complexity of the forest stand, proximity to human activity, and declining amounts of mid- to late-seral stage forest and contiguous young forest in the landscape (Marzluff et al. 1999, 2000; Marzluff and Raphael 2001; Raphael et al. 2002b). The implications of this are as follows:

- In fragmented landscapes, murrelet nesting stands may be more productive if surrounded by simple structured forests and by minimizing the effects of human recreation and settlement; and

- In extensive mature forest landscapes, murrelet productivity will best be enhanced by maintaining large, complex-structured forest far from human activity.

While artificial nest studies can provide some preliminary information on trends in rates of predation, they do not necessarily provide accurate

information on predation risk or actual rates of predation (e.g., Ratti and Reese 1988, Whelan et al. 1994, Yahner and Mahan 1996, Ortega et al. 1998, Sieving and Willson 1998, Wilson et al. 1998, King et al. 1999). Further study within suitable murrelet habitat and with real nests will be required to resolve our understanding of the effects of fragmentation and predator abundance on murrelet nesting success.

4.5.6.5 Predator Increases in Areas of Human Activity

As discussed above, proximity to human activity and landscape context appears to have the greatest impact on predator densities. In many studies, significantly more predators, especially corvids, occurred in campgrounds, along suburban forest edges, and in other areas close to human development (e.g., Masselink 2001, De Santo and Willson 2001, Marzluff et al. 2001, Burger 2002, Neatherlin 2002, LeValley et al. 2003). However, the relationship with human activities varied with predator species. For example, common ravens and Steller's jays did not always respond to human settlement or recreation (e.g., Marzluff and Neatherlin, in prep., Raphael et al. 2002b). In addition, many mammalian predators, especially squirrels, reach their highest abundance in the forest interior away from human activity (e.g., red squirrels) (Ratti and Reese 1988, Hannon and Cotterill 1998, Sieving and Willson 1998, Tewksbury et al. 1998, De Santo and Willson 2001).

The highest risk of predation was documented in areas close to humans (within 1 km), including along suburban edges and in campgrounds, dumps, and other areas of development, where human food sources attract predators, especially corvids (De Santo and Willson 2001, Marzluff et al. 2000). Edge-dependent nest predation also occurred in forests near clearcuts and in clearcuts, but only when highly fragmented and near human activities (e.g., De Santo and Willson 2001). Predation by corvids declined at nests greater than 200 m from edges adjacent to human activity (Raphael at al. 2002b).

4.5.6.6 Predation Risk Relative to Fragmentation and Edge

The effect of forest edge and fragmentation on predation rates has not been consistent among landscapes. Given the variability in research results from these and other studies, it is difficult to summarize and generalize the impacts of habitat fragmentation on birds in a western forested setting. However, some patterns are beginning to emerge from these studies, including:

- Higher nest predation in areas with high predator densities;

- Increased abundance or diversity of predators with the variety and complexity of habitats;
- Increased abundance of some corvid species along edges or in forest fragments near human activities;
- High nest predation by corvids along edges near human activities or in areas of low forest cover; and
- High predation by small mammals in a variety of habitats including interior forests and along anthropogenic edges.

Understanding the effect of fragmentation on marbled murrelet populations is critical for creating and managing habitat for this species. The most important factors in the risk of predation of murrelet nests seem to be landscape context or composition (including proximity to human activities) and its effect on the type of predators present, their abundance, and foraging efficiency. Predation rates could also be influenced by within-stand habitat characteristics such as nest placement and habitat quality; these have not been studied with respect to marbled murrelets. While the relationship between forest fragmentation, edge effects, and murrelet nest success is not completely clear, it is apparent that the abundance of some predators is affected by anthropogenic factors, and the risk of predation may be higher in proximity to human activities. As fragmentation and human development increase over time, edge-related predation could become more pronounced. Therefore, lowering the risk of predation will require creating and maintaining abundant, large, complex-structured forest in areas isolated from human development. In addition, in a fragmented landscape, predation at murrelet nests could also be reduced by surrounding stands with simple structured forest and by minimizing the effects of human recreation and settlement (Marzluff et al. 2000, Ripple et al. 2003).

4.5.7 Effects of Noise Disturbance at Nest Sites from Human Activities, Including Research and Survey Efforts

Very little information is available on the effects of disturbance on murrelets, but studies on other wildlife have documented various impacts from noise and human activity such as increased energetic expenditure, stress levels, and susceptibility to predation (Knight and Gutzwiller 1995). There are also numerous examples of habituation to noise.

Information on the effects of noise disturbance at marbled murrelet nest sites is largely based on limited anecdotal evidence and empirical study of only a single chick or pair of nesting adults (Nelson 1997, Long and Ralph

1998, Golightly et al. 2002). Much of the anecdotal information has been provided by biologists reporting incidental disturbance observed during research. Thus, this section addresses disturbance from research and survey efforts, as well as other human activities (recreation, logging, vehicle use, etc.). Overall, anecdotal reports indicate that murrelets typically exhibit only a limited, temporary behavioral response (if any) to noise disturbance at nest sites and are able to readily adapt to both ambient and specific local auditory stimuli (Singer et al. 1995, Long and Ralph 1998, Golightly et al. 2002).

Corticosterone studies have not been conducted on murrelets, and it is not known at what levels and for what duration corticosterone levels need to be elevated before the likelihood of injury is created. Corticosterone is released by the hypothalamo-pituitary-adrenal gland to help animals respond to environmental stress. Chronic high levels may have negative consequences on reproduction or physical condition of wildlife (Marra and Holberton 1998). Male northern spotted owls whose home-range centers were within 0.25 mile (0.41 km) of a major logging road or recent (within 10 years) timber activity showed higher levels of corticosterone than those with home-range centers farther from logging roads or recent timber activity; females showed no such increase in hormone levels (Wasser et al. 1997).

The solitary nesting biology of the marbled murrelet and relatively limited number of known nests has precluded a systematic study comparing nest success and breeding pairs exposed to noise disturbance versus non-disturbed "controls." To date, summary studies on the effects of disturbance on marbled murrelets have documented no known instances of nest failure, abandonment, or chick mortality directly attributed to nest site noise disturbance (Singer et al. 1995, Hamer and Nelson 1998, Golightly et al. 2002). When noted, responses to auditory stimuli at nests sites are generally limited to modifications of posture and on-nest behavior (T. Hamer pers. obs., K. Nelson pers. obs., Simons 1980, Long and Ralph 1998). Hamer and Nelson (1998) found that pedestrian activity within 131 feet (40 m) and within line-of-sight of murrelet nests caused flushing behavior or aborted feeding 27% of the time. Adults do not appear to be affected by vehicle traffic or most loud noises, excluding blasting and helicopters (Long and Ralph 1998).

An indication of murrelet tolerance of noise disturbance is provided by 2 of the earliest marbled murrelet nests found. These sites were located in Big Basin State Park, a popular park located in Santa Cruz County, California (Singer et al. 1991). Much of the information on murrelet response to auditory disturbance stems from observation of birds in these 2 nests, as well as nests in subsequent years located close to areas heavily used by park

recreationists. The first 2 nests found in this area were located 33 and 82 feet (10 and 25 m) from hiking trails and were thus exposed to periodic low-level human noise disturbance during the daytime. One of these nests was also only 115 feet (35 m) from the park's sewage treatment facility, and would have been subjected to the relatively constant hum of machinery. Yet, Singer reports that birds incubating on these nests "only rarely showed behavior suggesting agitation from human presence or noise" (Singer et al. 1991). Nests monitored in subsequent years (1991, 1992, and 1993) were all located on branches overhanging a major hiking trail used by approximately 25,000 visitors a year. For adults at these nests, Singer reports observing "no visible reaction to loud talking [or] yelling" near nest trees and chicks successfully fledged (Singer et al. 1995). However, it should be noted that nests located in the redwood forest zone of California are usually much higher off the ground due to tree heights than nests in the Douglas-fir or western hemlock zone in Washington and Oregon; thus, nest disturbance may be less due to this factor alone.

Naslund, reporting on observations of marbled murrelet pairs nesting in Big Basin State Park in 1989, notes that incubating and attending adult murrelets might sit up and look around in response to loud noises or other unusual human-induced auditory stimuli, but otherwise appeared to be largely unaffected by people nearby (Long and Ralph 1998). Nelson, in a description of murrelet behavioral response to disturbance at these same nests (1989), reports that nesting birds exhibited no notable response to human activity in the vicinity of nest trees (Long and Ralph 1998).

Nesting murrelets appear to be more sensitive to auditory stimuli – especially concomitant to visual cues– in habitat where nests are less inaccessible and/or as disturbance stimuli approach in proximity (Simons 1980, Long and Ralph 1998). Researchers in Alaska commonly report flushing incubating birds from ground nests as they approach (Long and Ralph 1998). On Naked Island, Alaska, where the harsh climate restricts tree height increasing the susceptibility of nests to disturbance, Naslund reported flushing an incubating bird from a previously undetected nest as she approached the nest tree (Long and Ralph 1998). Simons (1980) reports being able to approach within 16 feet (5 m) of a ground nest in the Barren Islands, Alaska, by circling around and observing from an overhang. The incubating adult did not appear to notice the observer but did become alert in response to noise such as shuffling feet or camera clicking.

Hamer has reported changes in adult feeding behavior in response to his presence in immediate proximity to nests (i.e., in the nest tree or tree canopy) during monitoring (T. Hamer pers. obs.). While he was at a nest in the Mt.

Baker/Snoqualmie National Forest of Washington in 1991, an adult landed with a fish less than 3.2 feet (1 m) away on the nest limb. The bird watched Hamer for less than a minute and then left with the feeding fish and circled the nest tree. Before Hamer descended the tree, the bird again landed on the nest limb, and quickly flew off without feeding the chick. The incident did not result in nest abandonment, and the breeding pair successfully fledged a chick that year. In the Suislaw National Forest of Oregon in 1992, Hamer had climbed a tree and was less than 9.8 feet (3 m) from an active nest when an adult returned with a fish (T. Hamer pers. obs.). The bird did not land but circled the nest tree until Hamer could reposition himself in the tree out of sight of the nest. The adult then landed and fed the chick apparently without notice of the investigator still in the tree.

Jones noted a different behavioral response by a feeding adult murrelet to the presence of an investigator near an active nest (Long and Ralph 1998). Jones had climbed an adjacent tree and was within 33 feet (10 m) of the nest. Although the investigator's presence elicited no overt response from the chick, the feeding adult froze on the nest limb upon landing and did not attempt to feed the chick until the investigator descended from the tree. Once the investigator was grounded, the adult successfully fed the chick.

In comparison to nesting adults, murrelet chicks have generally been found to exhibit an even more muted behavioral response to noise disturbance and typically do not respond at all to auditory stimulus in the absence of accompanying visual cues (Long and Ralph 1998). Most information on nestling murrelet response to disturbance comes from anecdotal reports from investigators monitoring nests. Thus, nest-bound chicks are not necessarily responding to auditory disturbance alone, but to the presence of an investigator in proximity to a nest. In such situations, marbled murrelet chicks have generally been found to all but ignore investigators until in extreme proximity (<3 feet [1 m]) and to readily adapt to near-nest disturbance over time (T. Hamer pers. obs., Simons 1980, Hamer and Nelson 1998, Long and Ralph 1998).

In 1996, at a nest in Big Basin State Park, investigators installed and removed video equipment within 3.2 feet (1 m) of an active nest with no apparent reaction from the chick (Long and Ralph 1998). Although the chick was at first thought to be asleep, it exhibited defensive behavior when investigators approached within 1 foot (0.3 m). Through observation of 2 different nests in 1991 and 1992, Hamer found that chicks quickly habituated to his presence when he was maintaining video equipment 3 feet (1 m) from each nest (T. Hamer pers. obs.). Upon initial visits to a nest, Hamer found a chick would snap its bill at him. With each encounter, behavioral response became

progressively more muted until, by the end, a chick would not even turn to face him upon approach. However, chicks were not found to become habituated to the investigator reaching for them and continued to exhibit a defensive behavioral response (e.g., rearing up, snapping, pecking) to such a stimulus. Simons (1980) reports a similar response to his presence from a marbled murrelet chick at a ground nest. Upon approach by an investigator, the chick was found to alter its posture to keep the observer in view and rear up on its legs and peck with an open gape if an investigator approached "too close."

Golightly et al. (2002) conducted the only known empirical study of the effect of noise disturbance (in absence of visual stimuli) on a marbled murrelet chick (n=1). Investigators documented the behavioral response of a 28-day-old marbled murrelet chick in Redwoods National and State Park in Humboldt County, California to the sound of a chainsaw (average dB 86.3; n=5) located 82 feet (25 m) from the nest tree. In addition, the investigators conducted point counts for corvids prior to, during, and after exposure to the auditory stimulus to assess the effects of human disturbance on corvid behavior, and the subsequent risk of predation on marbled murrelets. The investigators noted no "overt signs of disturbance" exhibited by the chick in response to the chainsaw noise, though the duration of "resting bouts" by the chick was found to be significantly longer post-trial (F=10.6, df=2, P=0.001; Bonferroni multiple comparisons test). Results of point counts indicate that corvid presence "may have changed through the disturbance trial" although the authors were not specific. Human activities have been known to affect levels of predation, however (see Section 4.5.6.5).

Response to vehicular noise at nest sites has been noted to be minimal for both murrelet chicks and nesting adults (Nelson 1997, Hamer and Nelson 1998, Long and Ralph 1998). Singer et al. (1995) report observing no visible response by murrelets to vehicles transiting on a "well-traveled park road" located within 230 feet (70 m) of nests monitored in Big Basin State Park from 1992 to 1994. Nelson, too, documented no response to vehicular noise from birds associated with nests in this same location in 1989. Chinnici also noted little response by murrelets to vehicles driving on a "lightly used" logging road located 230 feet (70) m from a nest in Humboldt County, California observed over 11 days in 1992. Chinnici noted that the chick once opened its eyes and became alert at the approach of a vehicle but otherwise did not respond to vehicular noise (Long and Ralph 1998). Nelson reports observing no response from chicks or adult murrelets to vehicular noise or such prolonged and/or loud noises as chainsaws, logging machinery, trucks, and a boom whistle sounded in the general vicinity of monitored nests (Long and Ralph 1998). Low altitude aircraft and boating activity are known to

alter murrelet behavior and immediate distribution in marine environments (Kuletz 1996, Speckmen 1996, Nelson 1997, Hamer and Thompson 1997), though investigators have typically noted no (or minimal) response by murrelet chicks and attending adults at nest sites (Kerns and Miller 1995, Long and Ralph 1998).

Documented responses by murrelet chicks and adults to calls and auditory cues from corvids and other potential predators ranged from no response and alert posturing, to aggressive attack and temporarily leaving a nest (adults only). However, it is unclear if auditory stimuli from such potential predators alone can elicit this gamut of behavioral responses in both murrelet chicks and adults. The most typical behavior documented in both chicks and nesting adults responding to a stimulus (including auditory) indicating the presence of a potential predator is to flatten against a tree branch and remain motionless (Simons 1980, Naslund 1993, Nelson and Hamer 1995).

Without large-scale systematic study, the effects of sub-lethal disturbance on murrelet populations will likely not be overtly manifest. Though noise disturbance at nest sites is generally thought to minimally affect individual birds and nesting pairs, large-scale effects of increased energy expenditure at the population level may be significant. The unique breeding biology of the marbled murrelet is not conducive to comparisons of reproductive success of "populations" exposed to noise disturbance versus undisturbed "controls" as possible with colonial alcids and other seabirds. However, population studies on other alcid and seabird species have generally revealed the detrimental effects of disturbance to breeding success and the maintenance of viable populations. Researchers have noted decreases in population reproductive success with progressive increase in disturbance regimes for black guillemots (Cairns 1980), tufted puffins (Pierce and Simons 1986), and least auklets (Piatt et al. 1990). All 3 species breed colonially, thus, allowing the immediate effects of noise disturbance at the population level to be readily discerned. Although marbled murrelet breeding biology may preclude such ready detection of the effects of sub-lethal noise disturbance at the population level, the effect of noise disturbance on murrelet fitness and reproductive success should not be completely discounted.

4.6 Summary and Conclusions

This section summarizes the primary conclusions related to inland habitat for the marbled murrelet, as drawn from a review and evaluation of research and records from 1992-2003.

4.6.1 Current Amounts of Suitable and Occupied Habitat

Based on information provided by land managers in the 3-state area, the Service estimates that there are currently 2,223,048 acres (899,639 ha) of suitable marble murrelet habitat. Of this amount, about 91% is located on Federal land; State, County, and private lands account for about 8%; and Tribal lands contain about 1%. About 47% of the suitable habitat occurs in Washington, 35% in Oregon, and 18% in California. However, comparisons between states and Conservation Zones are difficult to make because definitions of suitable habitat vary among administrative units. Surveys since 1992 have resulted in a number of changes to the extent of the inland range of the marbled murrelet, particularly in northern California, southwestern Oregon, and southern Washington. Occupancy rates appear lowest in California (16%) and highest in Oregon (51%). However, comparisons between states and Conservation Zones are difficult to make because definitions of suitable habitat and survey coverage vary among administrative units. The higher rate of misclassifying actually occupied sites as unoccupied during the early and mid-1990s likely resulted in loss of murrelet nesting habitat. The improved survey protocol has reduced this error in recent years.

One of the drawbacks to the current estimate of habitat is that many administrative units use northern spotted owl habitat definitions as a surrogate for murrelet habitat. Because northern spotted owl habitat is often defined at 80 years old and murrelet habitat typically does not develop by that time (Service 1997), the current estimate of 2.2 million acres (890,312 ha) is almost certainly an overestimate of suitable habitat. Improved estimates of murrelet habitat are needed in the future to allow for more accurate estimates of amount, quality, and spatial distribution. There are no available maps of suitable and occupied marbled murrelet habitat for the 3-state area.

4.6.2 Inland Detectability

Inland survey methods for marbled murrelets have changed substantially since the species was listed in 1992. The protocol for audio-visual surveys was modified in 1993, 1994, 2000, and 2003, providing increasing guidance on the number and timing of surveys and identification of suitable habitat. The most recent protocol (Evans Mack et al. 2003) recommends the use of radar as well as audio-visual surveys. Both methods have their strengths and weaknesses and can also be used in tandem to accomplish particular objectives.

There are a variety of factors that affect the ability of observers to detect murrelets at inland sites, including weather, daily variation in detection rates, season, conditions at the survey site (tree canopy closure, amount of visible sky), and distance from marine foraging locations (O'Donnell 1995, Nelson 1989, Baldwin 2002). The error in classifying sites as unoccupied when they were indeed truly occupied (false negative) is estimated to approach 15.5% for surveys conducted prior to 1998. The higher rate of misclassification of occupied sites in the earlier period of surveys may have led to the loss of an unknown number of breeding sites. The current error rate is estimated to be 4.2%, indicating substantial improvement in survey methods.

4.6.3 Habitat Characteristics

Habitat characteristics for the murrelet have been studied on landscape, stand/nest plot, and nest site scales, as summarized below.

4.6.3.1 Landscape Scale

At the landscape level, areas with evidence of occupancy tended to have higher proportions of large, old-growth forest, larger stands and greater habitat complexity, but distance to the ocean (up to about 37 miles [60 km]) did not seem important. Elevation had significant negative effects in some studies but not all. Variability in the elevation of occupied nest stands has been recorded across the range of the murrelet, but most data suggest that there is a negative relationship between elevation and stand occupancy. Hamer and Nelson (1995) found the mean elevation in a sample of 45 nest trees in British Columbia, Washington, Oregon, and California to be 1,089 feet (332 m), while data from Alaska had a mean elevation of 315 feet (96 m) and a maximum of 853 feet (260 m). This likely reflects the lack of low-elevation suitable habitat in Washington, Oregon, and California. Burger (2002) documented most nests (84%) in British Columbia below 3,281 feet (1,000 m) and found reduced stand occupancy with increasing elevation.

Five radar studies in British Columbia and 1 on the Olympic Peninsula showed significant positive correlations between numbers of murrelets and area of suitable habitat per watershed, although the measures of habitat varied between the studies. These data provide the first strong evidence that the watershed populations of marbled murrelets are directly proportional to the areas of old-growth forest available, along with the relationship shown between numbers of murrelets at-sea and the amount of habitat available in each Conservation Zone. Evidence from radar studies suggests that murrelets do not pack into remnant old-growth patches in higher densities as

areas of old-growth are reduced by logging. Breeding populations of murrelets are predicted to decline as areas of old-growth decrease.

Landscape-scale models indicated that the probability of murrelet occupancy or nesting was associated with stand age, tree height class, vertical canopy complexity, basal area (larger tree diameters), canopy closure, slope, distance to marine areas, fragmentation level, and elevation. In some cases, models improved when tree species composition was included as a variable.

4.6.3.2 Stand/Nest Plot Scale

Throughout the marbled murrelet's range, suitable nest stands have the following characteristics:

- Old-growth trees (typically >250 yrs);
- Large diameter trees;
- Variation in tree size and variation in canopy structure and/or gaps in the forest;
- Epiphyte mats on branches;
- Availability of potential platform limbs; and
- Low elevation (Burger 2002).

At the stand scale, murrelets tend to choose sites with larger platform trees with a high percentage of epiphyte cover; however, many of the differences seen between nest site selection and rejection have been shown at the landscape or at the nest tree level. Models at the stand and patch scale showed correlations between occupancy and a number of variables, including platform density, higher epiphyte thickness and percent cover of epiphytes, greater tree heights and canopy complexity (including number of canopy layers), larger tree diameters, densities of large trees, elevation, and slope.

4.6.3.3 Nest Site Scale

Few models examined habitat attributes at the tree and nest platform scale. Those that did found that models of nest trees had total platform number, moss cover and depth, and percent mistletoe infestation as predictive variables. At the nest platform scale, models included limb size, moss cover, and overhead cover as predictive habitat attributes. Overall, several characteristics seem to be important for murrelet selection of nest trees:

- Located near openings in the canopy for access to site;
- Larger limbs or deformities for nest platforms;
- Substrate for nest cup;
- Horizontal and/or vertical cover over nest site; and
- Sufficient height to allow jump-off departures and stall landings (Burger 2002).

Regardless of location, these 5 characteristics seem to be important in nest site selection. While differences in habitat selection are seen throughout the range of the murrelet, they seem to be a result of differences in forest species composition and structure, whereas selection at the microsite level or nest site remains fairly consistent.

Thus far, all identified nest trees in North America have been in conifers, with the exception of a single nest found in a red alder in British Columbia (Burger 2002). Available tree species throughout the range include: yellow cedar, western hemlock, Sitka spruce, Douglas-fir, western red-cedar, mountain hemlock, Pacific silver fir, and coast redwood. The current nest tree data do not accurately represent the entire range of the marbled murrelet as sampling efforts have been limited, and in British Columbia and Alaska have generally been focused in areas with the greatest accessibility. Furthermore, there have been no nest trees identified on the northern mainland coast of British Columbia, which supports a large percentage of the breeding population in this province (Burger 2002). Therefore, nest tree preferences exhibited by the murrelet may be misleading, and caution should be used when analyzing the data.

4.6.3.4 Variation in Terrestrial Conditions

Across the range of the marbled murrelet, a significant source of variation at the stand level appears to be found when comparing stands from northern latitudes to those in southern latitudes. Data on the height and dbh of nest trees, and the height of nest limbs, also show clear regional differences (Burger 2002). In Alaska for example, total tree density was 3 times greater, stand sizes were generally smaller, tree diameters were smaller, and tree heights and nest branches were lower than when compared to nest stands in the Pacific Northwest (Hamer and Nelson 1995). Burger (2002) also reported that nests from Vancouver Island, Oregon, and Washington were in larger trees than those from the Sunshine Coast, Queen Charlotte Islands, and Alaska. He suggests that these differences are a result of higher latitudes

and harsher climates, low-elevation logging that has removed most of the larger trees in the Sunshine Coast, and studies on Vancouver Island that have primarily focused on low elevation areas which still have very large trees.

4.6.4 Terrestrial Threats to the Marbled Murrelet

Terrestrial threats to the marbled murrelet are primarily the loss and fragmentation of nesting habitat.

4.6.4.1 Habitats Trend Since Listing

Loss of nesting habitat from timber harvest was 1 of the major reasons for listing the marbled murrelet in 1992. To estimate habitat loss since listing, the Service analyzed ESA Section 7 consultation records and information received from Federal land managers. Analyses of these 2 data sources produce different estimates of the amount of murrelet habitat lost since the species was listed. However, this is not surprising given the sources of data. The information received from the data request represented primarily Federal lands, whereas the consultation database compiled by the Service includes information on some projects specific to non-Federal lands. There are limitations to each of these 2 sources of information, and although the habitat loss estimates are different, the trend in habitat is similar.

Based on an analysis of available data from Federal lands managers, the combined loss of suitable murrelet habitat over the 3-state area between 1992 and 2003 was estimated to be 22,398 acres (9,064 ha). This represents about 1.1% of the suitable habitat reported by Federal land managers. Of this amount, 5,364 acres (2,171 ha) resulted from timber harvest and 17,034 acres (6,893 ha) from natural events. Sixty-six percent of the entire regional loss of suitable habitat occurred from a single wildfire (the Biscuit fire, 2003) in southwestern Oregon. Habitat loss due to timber harvest accounted for 24% of the total loss with the majority (75%) of timber harvest occurring in Oregon, and lesser amounts in California and Washington. Between 1992 and 1999, Strong (2003a) documented a moderate to major decline in murrelet marine counts along the Oregon coast and hypothesized that the lack of old-growth conifer forest, along with prey species availability, may be responsible.

An analysis of consultation records, which includes private ownership as well as Federal lands, suggests that the amount of suitable habitat lost since listing is potentially greater. Since 1992, the Service and CDFG have consulted on the potential removal of at least 209,000 acres (84,580 ha) of suitable murrelet habitat in the 3-state area. Approximately 7,370 acres (2,983 ha) of

suitable murrelet habitat anticipated for removal through Section 7 consultations were surveyed and found to be occupied by murrelets. However, more than 98,000 acres (39,660 ha) of habitat were either not surveyed or unknown to be surveyed. Thus, the estimated loss of 7,370 acres (2,983 ha) of known occupied murrelet habitat should be considered a minimum estimate. The primary difficulty with the analysis of consultation records is the inability to verify whether all projects as proposed were implemented or over what time period they may be implemented.

The total loss of suitable murrelet habitat from the 3-state area from 1992-2003 can be approximated by combining the estimates obtained from the 2 analysis as follows: 17,034 acres (6,893 ha) (0.8%) of habitat lost from natural causes (as reported by Federal land managers) plus 209,046 acres (84,598 ha) (9.4%), as estimated from consultation records (and assuming that the 5,364 acres of suitable murrelet habitat reported as harvested by land managers were included in the 209,046 acres (84,598 ha) covered by consultation). Thus, the total loss of suitable murrelet habitat between 1992-2003, not counting degraded habitat (28,119 acres [11,379 ha]), is approximated at 226,080 acres (91,492 ha), or about 10% of the current estimate of 2.2 million acres (890,312 ha) of suitable habitat. In addition, more than 10,000 potential nest trees were consulted on for removal.

For a wide-ranging species like the marbled murrelet, it is difficult to track and keep tally on the loss of habitat through consultations by different offices across a 3-state area. In the future, potential loss or modification of habitat through consultations by the Service through the entire listed range should be periodically assessed in a cumulative fashion.

Overall, the data demonstrate that there has been a continued loss of suitable murrelet habitat due to timber harvest and wildfire since the species was listed in 1992 (Table 4.6-1). Further, the time since listing is too short to expect any measurable amount of habitat development. The rate of decline in habitat has varied by ownership. NWFP guidelines and critical habitat designations have greatly reduced risk of habitat loss and modification on Federal lands, which account for 91% of suitable murrelet habitat. Changes in the rate of decline in habitat on private and State ownership vary by region. The development of comprehensive State protection guidelines for habitat on private and State lands has likely greatly reduced the loss of occupied habitat in Washington, at least for habitat outside of consultations by the Service and HCPs. However, the rate and risk of habitat loss in Oregon and California on private lands have likely remained the same since listing (that is, high). In addition, habitat that is not occupied will continue to be harvested on State and private lands. Besides stochastic events, the

Table 4.6-1. Threat of habitat loss to murrelets in 1997 compared with 2003.

Recovery Plan – 1997	Status in 2003
Murrelets use forests that primarily include old-growth, but also use mature forests with old-growth component (remnant trees).	Similar understanding of habitat use. Further study has shown a more restricted inland distribution in northern California and southern Oregon than previously known.
Recovery time for habitat once lost is generally 100-200 years.	Similar understanding.
Population still suffers from large-scale loss of habitat in the past, primarily due to timber harvest.	Habitat loss has continued, without appreciable development of new habitat. The population is still likely to be suffering from the effects of habitat loss.
Habitat loss is a threat to the recovery of the species. Recovery actions recommend maintaining occupied sites and minimize loss of suitable habitat.	Habitat loss has continued. Known occupied habitat has been lost. Loss of suitable habitat is expected to continue in the future based on ongoing Section 7 consultations and full implementation of approved HCPs. Since 1994, the rate of habitat loss has declined substantially on Federal land under the NWFP, and the rate has likely declined in Washington where State protection guidelines have been developed. The relative threat of habitat loss has not changed on non-Federal land in Oregon, which has no protective measures, or California, which protects the murrelet under state ESA.
Recommend increasing habitat amounts, but will likely take 50-100 years or more.	No estimated appreciable development of new habitat.
Historic loss of habitat resulted in reduced vigor of the species that now makes the murrelet more vulnerable to environmental fluctuations and catastrophes. Chance events such as floods, fire, oil spills, and windstorms could now cause or facilitate the extirpation of the entire listed species or specific zone populations.	Since development of the Recovery Plan, the largest single natural stochastic event resulting in murrelet habitat loss was the Biscuit fire (14,878 acres of murrelet habitat lost) in southwestern Oregon. This chance event has caused further habitat reduction that may further increase the vulnerability of the species to future stochastic events or management projects.

greatest "loss" of habitat in the 3-state area has resulted from consultations on individual harvest units, individual trees, and large amounts of suitable habitat through HCPs. Unpredictable stochastic events like the Biscuit fire also represent a great risk of habitat loss, both now and in the future. The analysis indicates a continued downward trend in available inland nesting habitat, and most importantly, it shows a loss of known murrelet breeding sites. The continued protection of nesting habitat on Federal land could be severely compromised if the NWFP is altered. The BLM is currently considering the elimination of LSR designations.

4.6.4.2 Effects of Forest Fragmentation

Murrelets are thought to be highly sensitive to forest fragmentation (Hansen and Urban 1992), and changes in their distribution and abundance have occurred in association with habitat loss and forest fragmentation (Service 1997). If murrelets are forced to utilize marginal habitat, nesting success could decline over time, leading to low nesting density and small populations (Raphael et al. 2002b). The fecundity rates of remaining pairs could also

decline. Smaller patch size may also affect murrelet nest success and the number of nests, which may ultimately have long-term consequences on population size.

By far the greatest threat to marble murrelets from forest fragmentation is increased levels of nest site predation associated with forest edges (Service 1997). Predation has consistently been the most significant cause of nest failure, and corvids are implicated as the primary predator of murrelets.

Recent studies have shown that most active nests have failed (>50%) it and the majority of nest failures are due to predation (78%). Murrelets tend to nest along natural edges, but the relationship between nest success and distance to edge is equivocal. However, the abundance and predation rates of avian predators, especially some corvid species, can be affected by edge and areas of low forest cover, as well as by landscape features such as type of matrix habitat and the proximity to human disturbance (Raphael et al. 2002b).

The highest risk of predation has been documented in areas close to humans (<1 km); the food sources in these areas attract predators, particularly corvids.

The most important factors in the risk of predation of murrelet nests seem to be landscape context or composition (including proximity to human activities) and its effect on the type of predators present, their abundance, and foraging efficiency. In conclusion, it appears that:

- In fragmented landscapes, murrelet nesting stands may be more productive if surrounded by simple structured forests and by minimizing the effects of human recreation and settlement; and

- In extensive mature forest landscapes, murrelet productivity will best be enhanced by maintaining large, complex-structured forest far from human activity.

5.0 MARINE HABITAT

5.0 MARINE HABITAT

The breeding and winter range of the marbled murrelet is associated with the subarctic/temperate waters that characterize portions of the eastern North Pacific Ocean. Along the North American coast, the marbled murrelet range stretches from the western Aleutian Islands southeastward to southeast Alaska and British Columbia. Subarctic waters (and the murrelet breeding and wintering range) also juts farther southward in a narrow band along the North American coast to central California (Nelson 1997). The range, thus, overlaps 2 major oceanographic systems — the Alaska Current and the California Current — and a third smaller one, the Aleutian North Slope Current. The former 2 currents originate from the Kuroshio Current or west wind drift, which is part of the North Pacific Gyre to the south and the Subarctic Gyre to the north; when this flow reaches the North American continent, it bifurcates into the northward flowing Alaska and southward flowing California currents. The Aleutian North Slope Current originates in the Bering Sea and flows eastward along the northern side of the Aleutian Islands.

5.1 Marine Habitat Characteristics

There is no new information on the marine habitat as it directly applies to marbled murrelets, but there is a growing evolution of knowledge in regard to decadal scale changes in the marine climate of the North Pacific Ocean. The discussion below on the oceanography of the eastern North Pacific and Aleutian Islands is based on the following sources: Favorite (1974), Ainley and Boekelheid (1990), Hunt (1995), Hickey (1998), Royer (1998), Reed and Stabeno (1999), Stabeno et al. (2002), Jahncke et al. (in prep.), and Ladd et al (in prep.); these references should be consulted for more detail.

5.1.1 Large-Scale Geographic and Temporal Variability in Marine Habitat

Marine habitat characteristics vary greatly over the range of the marbled murrelet. Murrelets occur in marine habitats whenever they are not involved in breeding activities in the terrestrial environment. The effects of geographic and temporal variability in physiography, ocean currents and conditions, and weather are discussed below for the northern (i.e., Aleutian Islands), central (i.e., Alaska Peninsula to northern Washington), and southern (i.e., Western Washington to California) portions of the marbled murrelet's range.

5.1.1.1 Physiography

Coastal waters in the Aleutian Islands used by the northern murrelet population are limited to narrow, shallow areas around islands, including more stratified waters in bays and inlets and strong currents between islands. On occasion, murrelets are also observed more than 62 miles (100 km) from shore (Piatt and Naslund 1995). Little or no estuarine habitats exist. High marine productivity occurs regularly near the Aleutian Islands due to upwelling, water mixing, and tidal fronts (Shuntov 1993). In winter, much of the northern and eastern parts of the Bering Sea become covered in sea ice, but the Aleutian Islands remain ice free. The main features important to small-scale current flow in the Aleutians are complex tidal currents, weather fronts, tide rips, and convergence areas they create when flowing through passes (primarily from the North Pacific to Bering Sea) separating islands.

From the Alaska Peninsula to northern Washington, coastal waters used by the central murrelet population are extensive and have a complex marine habitat structure, consisting of a broad continental shelf, numerous forested islands, and an abundance of protected waters. Off the Alaska Peninsula to Yakutat Bay, murrelets also occur farther from shore (i.e., >62 miles [100 km] on occasion (Piatt and Naslund 1995). Nearshore habitats used by murrelets are dominated by productive estuarine waters in fiords and inlets, with a variety of small-scale currents and tidal actions that can concentrate prey. Offshore waters are dominated by the Alaska Current, which flows north in this region (Thomson 1981) and leads to highly productive marine conditions for much of the year due to local upwellings, divergences, convergences, and nutrient input from fiords and coastal rivers. At different times of the year, offshore waters can project into nearshore habitats (late summer and fall), and nearshore waters can project into offshore habitats (winter to early summer).

In stark contrast, the coast of North America south of Cape Flattery (Washington) is relatively simple, having mostly a very narrow shelf (3-13 miles [5-20 km] wide) and no major islands. Coastal rocks and sea stacks occur infrequently and, in general, are sparsely vegetated or covered with herbaceous plants. Coastal waters are limited and directly affected by highly productive offshore waters of the southward-flowing California Current, as well as estuarine conditions near large bays and the mouths of major rivers. Nearshore habitats used by murrelets are dominated by productive estuarine waters in fiords and inlets, with a variety of small-scale currents and tidal actions that can concentrate prey. The primary features influencing small-scale California Current flows are several capes that deflect the southward flow of the current in meso-scale eddies (e.g., Cape Blanco, Cape

Mendocino, Point Arena, Point Reyes, Point Sur, Point Conception). The only major banks or extensive nearshore shallows are Heceta Bank off central Oregon, the Gulf of the Farallones, and northern Monterey Bay. Because of meteorological conditions, the width of the marbled murrelet breeding habitat decreases rapidly with decreasing latitude. A short way inland in Oregon, and especially California, the climate becomes near xeric in character because the coastal mountain ranges trap moisture from storms. The murrelet population along this coast is small and patchy (Nelson 1997).

5.1.1.2 Ocean Currents

The regions influenced by the Alaska and California currents contrast in important ways. Both major currents are the result of average wind patterns. Because of winds around the approximately stationary Aleutian Low Pressure system in the Gulf of Alaska (i.e., the so-called Subarctic Gyre), the basin-scale current flow is counter-clockwise. Due to the effect of the Earth's rotation (Coriolis Force), whereby moving water particles in the Northern Hemisphere are deflected to their right, the large-scale flow along the North American coast is, in part, onshore. The onshore flow results in down-welling. Very nearshore and over the continental shelf, the Alaska Current, already of low salinity, is diluted even more by the tremendous runoff of numerous coastal rivers. This runoff is the result of numerous storm centers that move across the Gulf toward the mainland, and which then dump their moisture upon reaching the coastal mountains. These inshore waters are known as the Alaska Coastal Current, which like the Alaska Current flows north and then west along the southern border of the Aleutians. These 2 currents are highly productive for much of the year owing to local upwellings, divergences, and convergences, as well as the input of nutrients from the many coastal rivers. However, differences in biological and physical properties of waters surrounding the Aleutians indicates 2 distinct marine environments in the Aleutians that are divided at approximately Samalga Pass (at 169°W in the eastcentral Aleutians). Waters to the west of Samalga, influenced by the Alaska Current and deep Bering Sea, are considered as oceanic, whereas those to the east of Samalga, influenced by the Alaska Coastal Current, are considered coastal.

In contrast to the Gulf of Alaska and Aleutians, airflow around the North Pacific High Pressure System, located south of the Gulf, is clockwise and, thus, so is current flow around the North Pacific Gyre. Because of the general northwest-southeast orientation of the coast from Cape Flattery south, the resulting northwest winds are parallel to the coast and, owing to the Coriolis Force, water is diverted offshore. To replace these surface waters, cold water is upwelled from depth, especially true during the spring

and summer when the High Pressure system is at its northernmost position of the year. Winds during those seasons are persistent from the northwest. The High Pressure system also shifts the jetstream to the north. As a result, the Pacific storms traveling along it pass to the east through British Columbia and southeast Alaska. During fall and winter, this pressure system, and the jetstream, migrates south. The North Pacific storms then move onto the Oregon and northern California coasts. These winds, associated with low pressure, blow from the south and lead to northward water flow and downwelling of coastal waters as well as to the surface expression of the Davidson Countercurrent, which flows north along this coast. During summer, the Davidson Countercurrent is subsurface. Were it not for the upwelling of cold waters along this coast during summer, the marine climate would be subtropical and marbled murrelets likely would not exist south of Washington. Moreover, production of phytoplankton is largely confined to the upwelling period of spring and early summer.

5.1.1.3 Variability in Ocean Conditions and Weather

The weather is very wet, more or less year round, along the North American and Aleutian coast of the Gulf of Alaska. In contrast, rainfall becomes increasingly seasonal with decreasing latitude from Cape Flattery south. In Washington, storms are infrequent only from midsummer through fall, but in central California Pacific storms are rare for most of the year, from spring to fall. This pattern is the result of the seasonal migration of the eastern Pacific high pressure system and its effect on the jetstream as noted above.

Because of its intense wind-driven upwelling, the California Current is one of the most productive ocean stretches in the world. However, this production besides being seasonal, is also affected dramatically on an interannual basis by remote factors. Among these is the El Niño-Southern Oscillation (El Niño). El Niño warm events occur every 4-7 years and last for a few to several months. These events result from internal ocean waves generated in the tropics, with corresponding changes in the mixed layer depth and local weather as the result of an infusion of warm water along the western North American coast. Productivity and the food web are negatively affected. In strong El Niño events, seabirds forgo breeding entirely, and in weaker ones seabird reproduction is reduced significantly (fewer birds breed, and greater numbers fail in their breeding attempts; Ainley et al. 1995b). The opposite occurs during La Niña events, when the infusion of nutrients is enhanced. During La Niña, west coast seabirds are by far their most productive.

With increasing latitude along the North American coast, the effect of El Niño decreases. In an analysis of commercial catches of fish, Hallowed et al.

(2001) found that of 16 El Niño warm events identified in the tropical Pacific between 1950 and 1998, fish production in the California Current (Washington to California) was affected in the 13 strongest events. In the eastern Gulf of Alaska (British Columbia, southeast Alaska) reduced production was evident in correspondence to only 6 of these events. In the western Gulf of Alaska, production was affected in 9 events. These results indicate the much greater instability in the ocean processes of the California Current compared to those of the Alaska Current.

Another factor related to production in the northern North Pacific region is the Pacific Decadal Oscillation (PDO) (Hare and Mantua 2000, Mantua and Hare 2000). The PDO oscillates on a time scale of 15-20 years. The ocean processes involved are not fully understood, but a change in the depth of the mixed layer seems to be a pivotal factor. What is well known is that fish populations, especially salmon, do well in the Alaska Current during "warm" phases of the PDO (mixed layer deep) but do poorly during "cold" phases (mixed layer shallow); just the opposite is true for fish populations in the California Current (Hallowed et al. 2001). Zooplankton are affected in a corresponding fashion (Batchelder et al. 2002, Batchelder and Powell 2002). In other words, periods of highest production are out of phase between the 2 ocean systems.

Not much is known about how marine bird populations respond to the PDO in Alaskan and British Columbian waters, but in the California Current, seabirds have done very poorly during the most recent warm phase of the PDO (Veit et al. 1997, Ainley and Divoky 2001). The response of the marbled murrelet to the PDO is unknown, but presumably variation in its breeding success is consistent with other species in the 2 respective climate-ocean systems. There is evidence that murrelets respond to ocean conditions similarly to other seabirds, at least on the El Niño-scale of temporal variation, with fewer individuals seen during El Niño warm events (Ainley et al. 1995a; Strong et al. 1995; Burger 1995, 2000). Thus, marbled murrelets in the California Current may have been similarly negatively affected by the recent warm phase of the PDO. Further discussion of changes in prey availability concurrent with the PDO can be found in Section 5.3.3.

5.1.2 Small-Scale Geographic and Temporal Variability in Marine Habitat

The findings of Miller et al. (2002) are of considerable relevance when evaluating small-scale variability of the marine habitat used by foraging marbled murrelets. Especially during the breeding season, marbled murrelets were most abundant offshore of large, clustered, and unfragmented old-

growth forests adjacent to abundant second-growth conifer forests, regardless of the characteristics of the adjacent marine habitat. Similar findings were recorded by Ainley et al. (1995a) farther south, offshore of Año Nuevo and the Big Basin Redwoods California State Park.

In all parts of its range and in all seasons, this species is found mostly in shallow (<197 feet [60 m]), usually sheltered, nearshore waters (Nelson 1997), although off southeast and western Alaska, the species is occasionally recorded >62 miles (100 km) from shore (Piatt and Naslund 1995). However, the continental shelf in that area is very wide. In Alaska, murrelets are most abundant within or near bays, fjords, and island passes (Piatt and Naslund 1995). Alaskan waters subject to strong tidal mixing provide poor habitat compared to stratified coastal waters (Piatt 1993). Murrelet densities off British Columbia are highest over sandy substrate, near estuaries, and where waters are coolest, but they avoid waters close to glaciers and near herring spawn areas (Yen et al. in prep.; Burger 2002). Murrelets on the outer coast of Vancouver Island and the Queen Charlotte Islands disperse during the post-breeding period to more sheltered waters, including shallower waters in many of the fjords along the mainland (Burger 1995).

Off California and Oregon, murrelets are most abundant near ocean bays, river mouths, sandy shores, and submarine canyons (Meyer et al. 2002). Becker and Beissinger (2003) found that marbled murrelets foraging off Santa Cruz County preferred cooler waters associated with areas of recent upwelling. However, such waters also correspond to those closest to breeding sites in that region (i.e., upwelling plume emanating from Point Año Nuevo).

In general, the selection of marine habitat, on the small scale, appears to be fairly uniform across the range of this species, barring the fact that fjords and island passes are absent along the North American coast south of British Columbia and Puget Sound of Washington.

5.2 Prey Abundance and Distribution

This section describes the abundance and distribution of prey over the range of the marbled murrelet. Virtually all information on food habits of the marbled murrelet has been summarized by Burkett (1995), and almost all of the available information is from areas north of Washington. Primary prey species range-wide include (in approximate order of importance) sand lance (*Ammodytes hexapterus*), Pacific herring, smelts (Osmeridae; especially capelin in the north and night smelt [*Spirinchus starskii*] in the south), northern anchovy, and crustaceans (mysids, euphausiids). Except for the crustaceans,

larvae and subadults of these fish species would be of a suitable size and shape for marbled murrelets.

5.2.1 Geographic Variation in Murrelet Diet

Over its entire range, prey for the marbled murrelet includes at least 27 taxa; at the small (local) scale, however, diet is much less diverse. Since the summary by Burkett appeared, there have been few new data presented on marbled murrelet diet. Kuletz et al. (1997) presented unpublished historic data from 129 marbled murrelets collected in Alaska, showing that sand lance, smelts, and herring were primary prey species there. Kuletz et al. (1997) also found that gadids (Gadidae; especially walleye pollock [*Theragra chalcogramma*]) were an important prey species, particularly after 1979. Grettenberger et al. (in prep.) analyzed the stomach contents of 7 marbled murrelets salvaged from gill-nets in Washington (mainly Strait of Juan de Fuca) in the 1990s, and also found that sand lance (85.7% frequency of occurrence) and Pacific herring (14.3%) predominated.

With the exception of birds salvaged from gill-nets, analysis of stomach contents requires collection of live birds. Because of the difficulty of studying diet of a threatened seabird, other recent (post-1995) studies involving foraging ecology have relied on at-sea observations of birds holding fish (Kuletz et al. 1995b, Day and Nigro 2000, Speckman et al. 2003), sampling of potential prey in-situ, where marbled murrelets forage (Derocher et al. 1996, Ostrand et al. 1998, Henkel and Harvey 2002, Becker and Beissinger 2003), and use of stable isotopes (Becker 2001). Observations of birds holding fish confirmed that marbled murrelets in Alaska forage principally on sand lance, herring, and capelin during summer. A study conducted in Prince William Sound, Alaska (Ostrand et al. 1998), found that potential prey in the water column where marbled murrelets foraged included walleye pollock, sand lance, capelin, and herring. In Desolation Sound (British Columbia), prey sampled near foraging marbled murrelets included sand lance and herring (Derocher et al. 1996).

South of Alaska and British Columbia, information on murrelet diet is meager, at best. Very little is known about diet on the outer coast of Washington and off Oregon. In central California, 2 studies (Henkel and Harvey 2002, Becker and Beissinger 2003) found that available prey where murrelets foraged included night smelt, white croaker (*Genyonomus lineatus*), northern anchovy, market squid (*Loligo opalescens*), Pacific sardine, Pacific herring, and Pacific sanddabs (*Citharichthys sordidus*). White croaker and night smelt have not been confirmed in the diet of marbled murrelets. Unidentified osmerids, recorded in diet studies in Oregon and California

(Burkett 1995), may well be night smelt. Most of the sardines found where marbled murrelets forage are adults, which are far too large to be swallowed by this species.

Becker (2001) used stable isotopes to study diet off Año Nuevo, central California. This method cannot resolve actual prey taken, but analysis of feather samples can indicate the trophic level at which the birds are feeding when the feathers are growing during pre-alternate (spring) and pre-basic (fall) molt. Becker (2001) found that prior to pre-alternate molt during 1999 and 2000, marbled murrelets fed at a low trophic level, presumably on euphausiids (or mysids). This pattern mirrors previous studies showing a higher proportion of crustaceans in the diet of marbled murrelets during winter (Burkett 1995). It also mirrors the diet of pelagic cormorants (*Phalcrocorax pelagicus*), which also forage near to shore like the murrelets, and whose diet in such habitat in central California contains a predominance of mysid shrimp (Ainley et al. 1981).

Analyses of geographic variation in diet are limited in that few studies have been conducted on diet south of Canada. However, Burkett's (1995) summary, which includes much anecdotal information from the southern portion of the marbled murrelet range, in conjunction with more recent information, reveals patterns in diet associated with latitude. Marbled murrelets likely prey on the most abundant, suitable prey (Burkett 1995, Nelson 1997; see Section 5.2.2 below). Sand lance and smelts are taken by murrelets throughout their range, with the former more predominant in the north and the latter more so in the south; herring and crustaceans may also be ubiquitous diet items (Table 5.2-1).

Sand lance abundance diminishes significantly with decreasing latitude south of Cape Flattery, and in central California it is unlikely that sand lance are a regular prey species. Smelts are likely important throughout the range, but dominant smelt species also change with latitude. For instance, capelin is an important prey item in Alaska, whereas surf smelt (*Hypomesus pretiosus*) and

Table 5.2-1. Distribution of major prey taxa in diet of marbled murrelets.

Prey Species	Geographic Area			
	Alaska	British Columbia	Washington & Oregon	California
Sand lance	X	X	X	X
Smelts	X	X	X	X
Crustaceans	X	X	P	P
Pacific herring	X[a]	X	X	P
Market squid		X	P	P
Rockfish		X	P	P
Northern anchovy		X	X	X

X=Documented prey item; P = Probable prey item, based on prey distribution.
[a] *Data from Kuletz et al. (1997); all other data from Burkett (1995).*

night smelt are likely important osmerid prey in Oregon and California. Euphausiids and mysids have been directly documented as prey only in Alaska and British Columbia, but more in-depth diet studies in the lower 48 states would likely reveal them to be consumed there as well. As noted above, Becker (2001) found that marbled murrelets in central California fed at a low trophic level during spring in some years, presumably on crustaceans. Similarly, Pacific herring, market squid, and rockfish (Scorpaenidae) have not been documented as prey items in California, but this may be due to the lack of direct observations. Juvenile rockfish are the prime prey species during spring of almost every seabird species so far studied in central California (Ainley and Boekelheide 1990), and would be of suitable size and shape to be consumed by murrelets.

In summary, diet north of Washington is dominated by sand lance, herring, and capelin, and south of Canada is probably dominated by northern anchovy, surf/night smelt, and herring. Mysids and euphausiids likely are consumed to a similar degree in both regions.

5.2.2 Geographic Variation in Prey Distribution

Marbled murrelets likely prey on the most available, suitable fish and mid-water crustaceans (Nelson 1997); thus, geographical variation in diet (Section 5.2.1, above) probably mirrors the small- and large-scale geographic variation in the availability of prey. Species composition of available prey changes to some degree between the Alaska and California currents, which diverge at Cape Flattery (see Section 5.1.1). North of Cape Flattery, sand lance, capelin, and herring are abundant. South of Cape Flattery, capelin do not occur, and sand lance become increasing scarce with decreasing latitude. South of Cape Flattery, northern anchovy and, in spring, juvenile rockfish are the dominant small, fusiliform (long, thin shaped) schooling fish found in inshore waters.

5.3 Variation in Prey Availability Due to Ocean Cycles

Environmental conditions, particularly El Niño events, have been shown to have dramatic effects on seabird productivity in California as a result of reduced prey availability (Ainley and Boekelheide 1990, Ainley et al. 1995b). Prey availability can vary at a variety of temporal scales, from hours to centuries. The following narrative addresses 3 primary scales of variation: seasonal, annual, and decadal.

5.3.1 Seasonal Variation

Spring and summer are the most productive oceanographic seasons throughout the marbled murrelet's range, though conditions vary north and south of Cape Flattery. To the north, longer summer days lead to strong primary production (growth of phytoplankton), fueled by nutrient input from coastal rivers and localized upwellings across the broad continental shelf. During winter, primary production is greatly reduced.

South of Cape Flattery, strong northwest winds during spring and summer drive extensive coastal upwelling, resulting in a productive marine environment over the continental shelf. This upwelling leads to abundant forage fish during summer and fall months, but prey availability is reduced during winter. This pattern is partially inferred from what is known about other seabirds along this coast (e.g., the common murre [*Uria aalge*]) (Ainley and Boekelheide 1990, Ainley et al. 1996), which is also abundant in the coastal waters frequented by murrelets. Throughout their range, marbled murrelets apparently consume more crustaceans during winter, presumably in response to decreases in forage fish availability.

While upwelling off Oregon and California fuels a productive marine environment, effects of upwelling early in the year on subsequent availability of larval and juvenile fish (the size-classes appropriate for murrelets) tend to be negative. Strong upwelling (or strong downwelling) in January and February (the peak parturition time of rockfish) can lead to decreased abundance of larval rockfish later on (Ainley et al. 1993). Similarly, Bailey and Francis (1985) found that recruitment of Pacific hake (*Merluccius productus*) negatively correlated with winter upwelling. Petermen and Bradford (1987) found that greater wind stress (which leads to upwelling) results in decreased survival of larval anchovies. These studies all concern abnormally strong and persistent (rather than pulsed) upwelling during the early spawning season. Normal upwelling (April to July) may transport juvenile fish offshore, which may lead to decreased survival (Ainley et al. 1993) or increased survival due to decreased predation (Yoklavich et al. 1996). Pulsed upwelling, with periods of wind relaxation and onshore transport, provides optimal conditions for larval fish survival (Ainley et al. 1993). Offshore advection of larval fish may reduce available prey for marbled murrelets, which typically forage very close to shore (<1.2 miles [2 km] from shore) south of Cape Flattery.

5.3.2 Annual Variation and El Niño

Variation in certain climatic factors, including the strength and timing of upwelling, contributes to annual variability in prey availability. Reduced upwelling and strong stratification of the water column during El Niño events lead to reduced prey for seabirds (Ainley et al. 1995b). Negative effects increase with decreasing latitude: effects are pronounced in California, whereas El Niño effects are rarely reported north of Washington (see Section 5.1.1, above).

Because so little is known about marbled murrelet diet or foraging behavior, especially during El Niño events, it is difficult to comment on the importance of oceanographic changes on marbled murrelets. However, changes in at-sea distribution of marbled murrelets in many parts of their range have been noted during El Niño events (see Section 5.1.1, above).

5.3.3 Decadal and Longer-Term Variation

The PDO (see Section 5.1.1) affects prey availability on a scale of decades. A major PDO-related regime shift in 1977 has been linked to concurrent changes in populations of fish and crustaceans north of Cape Flattery (Anderson and Piatt 1999, McFarlane et al. 2001, Hallowed et al. 2001). Crustacean populations in the Gulf of Alaska crashed, pelagic schooling fishes declined, and larger fish, including pollock, increased dramatically after 1977 (Anderson and Piatt 1999). Because juvenile pollock have lower energy density than pelagic schooling fishes, like capelin and sand lance (Anthony and Roby 1997, Van Pelt et al. 1997), this increase in pollock has apparently led to poor reproductive success of seabirds, and potential long-term population declines (Hayes and Kuletz 1997, Agler et al. 1999). It is not known whether or not marbled murrelets were also affected.

South of Cape Flattery, fish populations and zooplankton abundance have declined during the warm phase of the PDO, which was in place after 1977 (Batchelder et al. 2002, Hallowed et al. 2001), with concurrent negative effects on seabirds (Roemmich and McGowan 1995, Veit et al. 1997, Ainley and Divoky 2001). Perhaps related to the PDO is the crash of the Pacific sardine fishery in the California Current ca. 1950, followed by an increase in northern anchovy landings (Schwartzlose et al. 1999). Although the sardine crash likely was exacerbated by intense commercial fishing pressure, the timing of this decline indicates a link as well to atmospheric and oceanographic conditions (Klyashtorin 1997, Chavez et al. 2003). In the 1990s, there seems to have begun a shift from anchovies back to sardines (Chavez et al. 2003). Sardines are reportedly part of the murrelet diet, but

mostly the very large adults are found nearshore where the murrelets forage. Thus, it is not known if the recent resurgence in sardine stocks has had any measurable effect on murrelets. In addition to decadal-scale changes in marine food webs, there may be longer term changes in prey abundance. Emmett et al. (1997) suggested that anchovy abundance off Oregon declined between the 1970s and 1995. These long-term changes could have potential negative repercussions on marbled murrelets, if anchovies and not smelts were the predominant prey.

It should be noted that analyses of oceanographic effects on forage fish abundance can be confounded by effects of commercial fishing pressure for some species (e.g., Pacific sardine, Pacific herring, rockfish). The effects of commercial fisheries on prey resources for marbled murrelets are addressed in Section 5.4.1.

5.4 Marine Threats

The primary threats to marbled murrelets in the marine environment include:

- Reduced prey availability from overfishing;
- Trends in prey availability from oceanographic variability;
- Oil spills;
- Gill-nets;
- Other marine contaminants; and
- Disturbance from recreational boating and research and monitoring efforts.

5.4.1 Effects of Reduced Prey Availability from Overfishing

In Washington, Oregon, and California, many fish populations have been depleted due to: (1) overfishing from coastal fisheries; (2) reduction in the amount or quality of spawning habitat; and (3) pollution of coastal waters. Natural variation due to ocean cycles also affects the amount and distribution of fish populations. Often, the effect of potential overfishing is difficult to separate from other factors. The loss of commercial sardine fisheries in Washington, Oregon, and California in the early 20th century likely resulted partly or largely from overfishing. Over the last decade, in the absence of fishing, sardines have been staging a comeback and populations have been

increasing in California, Oregon, Washington, and southern British Columbia, although large-scale fisheries have not yet redeveloped.

Seabirds relying heavily on harvested fish species presumably could be affected by reduced fish populations, but impacts to murrelets may be greatly reduced or completely mitigated if they forage opportunistically on a wide range of available prey (including many non-harvested species), large potential foraging areas, and local shifts in foraging distribution (Service 1997). Primary prey species of the marbled murrelet (e.g., anchovy, herring, smelt, sand lance, tomcod, euphausiids, mysids) are of slight commercial fishery value in the lower United States, and there is very little geographic overlap between murrelet distribution and areas of commercial harvest (e.g., anchovy, herring). However, fisheries usually target larger adult fish while murrelets often eat smaller subadult and juvenile fish. Temporal and spatial lags may occur in impacts from overfishing, which are difficult to detect. Overall, it appears unlikely that marbled murrelets are affected to a significant degree by current levels of overfishing, but local impacts may occur in certain areas which could affect murrelet distribution at sea.

5.4.2 Effects of Prey Availability from Oceanographic Variation

Oceanographic variability can occur at a number of temporal scales, from within-day to decadal (see Section 5.3). However, marbled murrelet populations are most likely affected by fluctuations in prey availability resulting from El Niño events and longer-term changes, such as the PDO. Effects of the El Niño on marbled murrelets have not been well documented, and little is known regarding the influence of El Niño events on nearshore fish communities. In addition, because there is little information on the diet of marbled murrelets south of Canada, it is difficult to know how El Niño may affect marbled murrelets. However, effects on other piscivorous seabirds have been well documented (Ainley et al. 1995b), and several studies have found that El Niño events can influence the behavior of marbled murrelets. Burger (2000) found differences in the number of marbled murrelets detected with radar entering drainages in British Columbia in El Niño versus non-El Niño years. He thought that this indicated a decrease in breeding activity due to reduced prey availability. Early movement of alcids into Puget Sound in El Niño years may suggest that prey were more available in inner waters than in outer coastal waters, but abandonment of nesting colonies also may simply cause early dispersal. While El Niño events are thought to reduce overall seabird prey availability (and less so for marbled murrelets), inner coastal waters in Puget Sound and Strait of Juan de Fuca, as well as estuarine areas along the outer coast, are less

influenced by El Niño conditions due to mixing and nutrients from other sources than outer coastal waters (Service 1997).

Becker (2001) found changes in the at-sea distribution of murrelets and a shift to higher trophic-level feeding during spring (determined using stable isotope analysis) in El Niño years. The reproductive success of murrelets in central California was also found to be lower during years of decreased prey availability. Results of another study in central California documented a greater proportion of breeding birds, and shorter foraging times (time spent diving) in 2001 compared to 2000 (Peery et al. in prep.). Although there are no data on the relative prey abundance during these 2 years (neither year was considered an El Niño year), Peery et al. (in prep.) propose that these changes are the result of differences in prey availability.

Decadal-scale variation in oceanographic climate also may affect prey availability for marbled murrelets. Dramatic changes in the Gulf of Alaska marine community coincident with a climatic shift in the late 1970s may have had detrimental effects on marbled murrelets and other seabirds (see Section 5.3). In the California Current, seabirds have done very poorly during this most recent warm phase of the PDO (Veit et al. 1997, Ainley and Divoky 2001). These variations in prey availability can be viewed as natural and cyclic (although some climatic changes may be linked to anthropogenic causes), but may exacerbate other threats to murrelets in the marine environment.

5.4.3 Effects of Oil Spills

Marbled murrelet and seabird mortality from oil pollution is a significant conservation issue in California, Oregon, and Washington (Ohlendorf et al. 1978, Burger and Fry 1993, Carter and Kuletz 1995, Service 1997). When murrelets contact floating oil, feathers and skin become coated in varying amounts of oil, oil can be ingested (often as a result of preening), and fumes can be inhaled. Oiling probably results in death within days because oiled murrelets are recovered dead on beaches, and the few recovered alive usually die in captivity. Dead oiled birds can float at sea for several days or weeks, allowing them to be carried to shore by winds and currents. Oil spills in California, Oregon, and Washington typically occur close to shore. Given their nearshore distribution and dominant westerly and northwesterly winds in this area, most dead oiled murrelets are expected to reach shore, although some may be taken by scavengers at sea or become waterlogged and sink before reaching shore. Although large sections of accessible coastlines are routinely surveyed for beached birds, the small body size of murrelets makes them difficult to recover on shore. Carcasses are often removed by

scavengers, hidden in debris, or buried in sand. Small-bodied seabird carcasses are typically undercounted on beaches to a greater degree than large-bodied seabird carcasses, disappearing at rates 2-4 times more rapidly on the open coast (Ford et al. 2002). In some areas, lack of access also leads to very incomplete coverage of shorelines by beached bird surveys.

The actual number of marbled murrelets killed by oiling can be estimated through various modeling techniques that correct for undercounting (Ford et al. 2002). The chief long-term impact to murrelet populations from oiling is reduction in population size, but decreased breeding success and loss of certain nesting areas also may result when relatively large numbers are killed , as found in other alcids (Carter et al. 2003). The significance of oiling impacts on murrelets depends on the degree to which oil-related mortality increases the rate of decline and speeds the time to extinction for affected populations. Regardless of the level of oil impacts, they are additive to other factors negatively affecting murrelet populations (Service 1997).

5.4.3.1 Oil Pollution Sources

On the west coast, most oil spills and chronic oil pollution that can affect marbled murrelets occur in areas of high shipping traffic, particularly near the large tanker and shipping ports at Los Angeles and San Francisco, and in the Strait of Juan de Fuca/Puget Sound. Lower levels occur near smaller ports in Grays Harbor, Humboldt Bay, and the Columbia River (Neel et al. 1997, Service 1997, Carter 2003). During the 1990s, oil tanker and shipping traffic into west coast ports grew, increasing the amount of oil that could be spilled (Hampton et al. 2003a). In 1992, there were about 1,900 tanker trips into San Francisco Bay, 1,500 into Los Angeles (or Long Beach), and about 1,400 trips into the Strait of Juan de Fuca (Service 1997). Between 723 and 907 tankers entered Washington waters per year in 1996-98 (Neel et al. 1997). Canadian shipping (including 20% oil tankers) increased 6-11% between 1989-91 and 1999 through the Strait of Juan de Fuca (Hull 1999, Burger 2002). Small numbers of marbled murrelets in southern California also can be affected by oil spills from both ships (including tankers) and offshore oil platforms. Except for the 1969 Santa Barbara oil spill, most spills from platforms have been relatively small (Carter 2003, McCrary et al. 2003).

Since the U.S. Oil Pollution Act (1990) was instated, increased government regulations and industry efforts have led to lower numbers of oil spills from ships and tankers (Hampton et al. 2003a). In addition, offshore oil development in California (northern/central), Oregon, and Washington has not occurred; the 1992-2002 moratorium has been extended to 2012; and 5 national marine sanctuaries that prohibit offshore oil development have been

established (3 in central California, 1 in southern California, and 1 on the west coast of Washington). Overall, the threat of mortality of marbled murrelets from oil pollution appears to have been reduced by some unknown degree since the early 1990s, but further work is needed to summarize and examine tanker and other shipping traffic and information on oil spills. In addition, offshore oil development is currently under consideration in British Columbia, which could cause oil spills that enter Washington waters (K. Morgan, pers. comm.; D. Bertram, pers. comm.). No large spills have occurred in Alaska and British Columbia since 1989, the year of the infamous *Exxon Valdez* oil spill that killed over 8,000 marbled murrelets (K. Kuletz, pers. comm.; D. Bertram, pers. comm.). However, 6 marbled murrelets were recovered from a skimming boom after a diesel spill in Prince William Sound in 2001 (K. Kuletz, pers. comm.).

Despite some apparent reduction in oil pollution, oil spills continue to occur and kill relatively large numbers of seabirds, including murrelets. Federal and State agencies continue to respond to oil pollution by documenting seabird mortality, assessing seabird population impacts, rehabilitating some live oiled seabirds, and partly restoring seabird populations. Through litigation, Federal and State trustee agencies attempt to recover financial damages for the injury of seabirds from responsible parties. To date, settlement funds have been used in Conservation Zones 2 and 6 to partly mitigate impacts to marbled murrelets through purchase of private properties with old-growth forest nesting habitat, purchase of timber rights on tribal properties, and attempts to reduce gill-net mortality (Carter and Kuletz 1995, Melvin et al. 1999, Momot 1995, TMOSNRT 2000, Carter et al. 2003). However, these projects have provided only limited partial mitigation for spill impacts. In 2002, the source of several large mystery spills since 1990 in central California was determined to be the tanker *Jacob Luckenbach,* which sank in 1953. This discovery recently has increased concerns about oil pollution from other aging sunken vessels without responsible parties. Marbled murrelet mitigation through measures that reduce nest predation are being considered with funds sought from the Oil Spill Liability Trust Fund (Hampton et al. 2003b, COSNRTC 2003). If much higher breeding success can be restored, this project would provide more complete mitigation for oil spill impacts over a period of time and may help prevent extirpation of the murrelet population in Conservation Zone 6.

5.4.3.2 Oil Spill Frequency

Carter and Kuletz (1995) reviewed available information on numbers of marbled murrelets recovered and total mortality estimated from oil pollution prior to 1993-94. Seabird mortalities for many spills were reported after

about 1976, but most spills did not appear to be reported before 1977 (Carter 2003). In addition, 2 spills in Oregon and 3 spills in California were not included in the earlier summary because no marbled murrelets were known to be recovered, and there was little available information on seabird mortality from these spills (Watson 1979, Nehls 1980, Carter 2003). Using a more complete list of spills, including those with known murrelet mortality as well as spills with expected murrelet mortality based on documented seabird mortality where murrelets occur, 14 spills, or 0.88 spills per year, occurred in 1977-92 (Table 5.4-1). From 1993-2003, 9 spills, or 0.82 spills per year, occurred (Table 5.4-1). The overall rate of murrelet spills does not appear to have changed substantially between periods. Although also poorly documented, chronic oiling of murrelets also was likely similar between periods, given increased tanker and other shipping traffic but greater regulation. However, murrelet spills occur irregularly and the selection of specific years for period endpoints affects comparisons. In addition, some spills in 1977-92 may have been omitted.

To assess major changes in the frequency of spills potentially affecting murrelets within sections of the listed range, we examined the number of spills by zone between 1977-92 and 1993-2003. In 1977-92, spills occurred in Zone 1 (n=3), Zone 2 (n=3), Zone 3 (n=3), and Zone 6 (n=5; Table 5.4-1). In 1993-2003, spills occurred in Zone 3 (*New Carissa*; Ford et al. 2001), Zone 4 (*Kure*, *Stuyvesant*; Ford et al. 2002; D. Welsh, pers. comm.), and Zone 6 (*Cape Mohican*, Point Reyes Tarball Incidents, *Command,* San Mateo Mystery/*Jacob Luckenbach #1*, *Jacob Luckenbach #2*; Ford 2002, Carter 2003, Himes Boor et al. 2003; S. Hampton, pers. comm.). The number of spills appeared to decrease in Zones 1-2, increase in Zone 4, and remained unchanged in Zones 3, 5, and 6. However, the sporadic nature of larger oil spills with reported murrelet mortality makes the estimation of spill rates difficult at time scales of decades.

5.4.3.3 Oil Pollution Mortality

To assess major changes in yearly average mortality, we estimated mortality of marbled murrelets from oil pollution in each zone for all spills and chronic oiling from 1977-92 and 1993-2003 (Tables 5.4-1 and 5.4-2). Where estimates comparable to recent studies were not available, we determined approximate mortality by a rough estimation technique that used correction factors developed by Ford et al. (2002) (Table 5.4-1). Mortality from chronic oiling of marbled murrelets also was taken into account by assuming conservative annual levels of mortality per zone because oiled murrelets are rarely found on beaches except after spills (Carter and Kuletz 1995; S.

Table 5.4-1. Summary of oil spill mortality of marbled murrelets in Conservation Zones 1-6, 1977-2003.

Oil Spill[10]					No. Murrelets	Estimated	
No.	Mo.	Year	Name	Zone	Recovered[3,4]	Mortality[1]	Sources[2]
1	Dec	1984	Whidbey Island	1	0	[10-20]	1
2	Dec	1985	ARCO *Anchorage*	1	1	[10-20]	1
3	Feb	1991	Texaco Anacortes	1	0	[10-20]	1
4	Mar	1984	Mobil oil	2	1	[10-20]	1
5	Dec	1988	*Nestucca*	2	2-13[5]	8 [20-260][6]	1,2
6	Jul	1991	*Tenyo Maru*	2	[40]	[175-350][7]	1,3
7	Feb	1979	Lincoln Co. Coast	3	[1-10]	[10-200][8]	4
8	Mar	1980	Lincoln Co. Coast	3	[1-10]	[10-200][8]	5
9	Nov	1983	*Blue Magpie*	3	2-4	[20-80]	1
6	Jul	1991	*Tenyo Maru*	3	[5]	[25-50][7]	
10	Feb-Mar	1999	*New Carissa*	3	26	262	6
11	Nov	1997	*Kure*/Humboldt Bay	4	10	151	7
12	Sep	1999	*Stuyvesant*	4	24	135	8
13	Nov	1984	*Puerto Rican*	6	1-3	[10-60]	1, 9
14	Feb	1986	*Apex Houston*	6	5	12 [50-100][6]	1, 10
15	Dec-Feb	1989-1990	San Mateo Co. Coast	6	NA	[10-50]	11
16	Dec-Mar	1990-1991	San Mateo Co. Coast	6	NA	[10-50]	11
17	Dec-Jan	1992-1993	San Mateo Co. Coast	6	NA	[10-50]	11
18	Nov	1996	*Cape Mohican*	6	0	[6-12]	11
19	Nov-Mar	1997-1998	Point Reyes Tarball Incidents	6	3	122	12
20	Aug	1998	*Command*	6	0	6-12	13,14
21	Nov-Mar	2001-2002	San Mateo Mystery/ *Jacob Luckenbach #1*	6	0	[6-12]	15
22	Winter	2002-2003	*Jacob Luckenbach #2*	6	0	[6-12]	15

[1] Numbers in square brackets were estimated roughly during this review. In Zones 1-4, rough estimates were made using a correction factor of 10-20 times those recovered (Ford et al. 2002). In Zone 1, minimum recovery of 1 murrelet was assumed (due to the high likelihood of missing small numbers of oiled murrelets) without carcass recovery for certain spills. In Zone 6, Ford (2002) estimated 6-12 oiled murrelets in the 1998 Command *oil spill without carcass recovery, which was applied to other spills without carcass recovery in Zone 6.*

[2] Sources: 1 (Carter and Kuletz 1995); 2 (Momot 1995); 3 (TMOSNRT 2000); 4 (Watson 1979); 5 (Nehls 1980); 6 (Ford et al. 2001); 7 (Ford et al. 2002); 8 (D. Welsh, pers. comm.); 9 (PRBO 1985); 10 (Carter et al. 2003); 11 (Carter 2003); 12 (Himes Boor et al. 2003); 13 (Ford 2002); 14 (COSNRTC 2003); 15 (S. Hampton, pers. comm.)

[3] For the 1977-2001 period, we assumed minimal rates of 1-2/year in Zone 1, 1/year in Zones 2-4, none in Zone 5, and 1-3/year in Zone 6.

[4] NA, not applicable.

[5] Low end of range is number of marbled murrelets recovered; high end of the range includes unidentified alcids.

[6] Estimated mortality of 8 murrelets during the Nestucca oil spill and 12 murrelets during the Apex Houston oil spill probably were heavily underestimated. Numbers in square brackets were used in this review.

[7] This spill occurred mainly in Zone 2 but also in northern Zone 3. Since the majority of this spill occurred in Washington in the breeding season (70% juveniles killed; Warheit 1996), we assumed that 40 of 45 murrelets recovered were from Zone 2 and 5 of 45 were from Zone 3.

[8] Recovery of 1-10 murrelets was assumed, based on location and available spill information (Ford et al. 2001).

[9] Some oil mortality has occurred at the lower end of Zone 5 but we have assumed that these birds belong to the Zone 6 breeding population.

[10] Does not include the Chevron Texaco spill on December 30, 2003 in Zone 1.

Table 5.4-2. Summary of estimated oiling mortality of marbled murrelets by Conservation Zone, 1977-2002.

Zone	Period	Reported Spills[1]	Chronic[2]	Annual Mortality
1	1977-1992	30-60	16-32	2.9-5.8
	1993-2003	0	11-22	1.0-2.0
2	1977-1992	205-630	16	13.8-40.4
	1993-2003	0	11	1.0
3	1977-1992	65-530	16	5.1-34.1
	1993-2003	262	11	24.8
4	1977-2003	0	16	1.0
	1993-2003	286	11	27.0
5	1977-1992	0	0	0.0
	1993-2003	0	0	0.0
6	1977-1992	80-260	16-48	6.0-19.3
	1993-2003	156-220	11-33	15.2-23.0
Total	1977-1992	380-1,480	80-128	28.8-100.5
	1993-2003	704-768	55-88	69.0-77.8

[1] *See Table 5.4-1 for estimates per reported spill; does not include the Chevron Texaco spill on December 30, 2003 in Zone 1.*

[2] *Conservative annual chronic oiling mortality rates were assumed (Zone 1 = 1-3; Zone 2-4 = 1; Zone 5 = 0; Zone 6 = 1-3).*

Hampton, pers. comm.). However, mortality from chronic oiling may be higher (e.g., 10-20/year) without carcass recovery. Based on tanker and shipping traffic and available information on seabird oiling, chronic oiling was considered greater in Zones 1 (1-2/year) and 6 (1-3/year), lower in Zones 2, 3, and 4 (1/year/zone), and non-existent in Zone 5. Between the 2 periods, yearly average mortality appeared to be reduced in Zones 1 and 2, increased in Zone 4, and was similar in Zones 3, 5, and 6. Adding spills and chronic oiling estimates, we found similar overall numbers of murrelets killed by oil in the listed range between periods (1977-92: 29-101/year; 1993-2003: 69-78/year). However, population sizes within each zone probably were much lower in 1993-2003 than in 1977-92. Thus, even though similar numbers of murrelets were killed between periods, recent oil impacts likely constitute a greater proportion of populations than previously.

To assess the general degree of potential impacts of oil spill mortality for each Conservation Zone, we examined the number of murrelets killed in relation to zone population size and available information on source populations within each zone during winter when most birds are killed by oil spills (see Tables 5.4-1 and 5.4-2):

- Conservation Zone 1: Three small-moderate spills occurred between 1977-92, all during winter months. Many oil-killed murrelets in Zone 1 waters probably are resident members of the Zone 1 population, but 17-37% of winter numbers in northern Washington are considered to belong to Zone 2 and British Columbia populations (see Section 5.4.4.4). Using a series of assumptions, a rough estimate of winter population of 5,400-12,571 was derived for the San Juan Islands/Northern Puget Sound area, including about 900-4,700 (~17-37%) from outside this zone. Annual oiling rates of Zone 1 birds within Zone 1 waters in 1977-92 (~1.8-4.8/year) and 1993-2003 (1-2/year) are well below 1% of 4,500-7,900 Zone 1 birds, as estimated in 2000-02. Beissinger (1995a) noted that declining population projections begin to differ greatly when human-caused mortality exceeds 1% of the population.

  There were no oil spills recorded in Zone 1 from 1993 until December 30, 2003, when about 4,800 gallons of oil overflowed during refueling operations at Chevron's Texaco terminal at Point Wells in northern Puget Sound. No murrelets were recovered, but some may have been killed and not recovered.

- Conservation Zone 2: Two large oil spills (*Nestucca* and *Tenyo Maru*) occurred in Zone 2 in the 1977-92 period (Table 5.4-1). The *Nestucca* spill occurred in December when some Zone 2 murrelets appear to be resident while others appear to disperse, possibly into Zone 1 (Speich et al. 1992). There is evidence of some dispersal of birds into Zone 2, probably from Zone 3 (Speich and Wahl 1995; Thompson 1997, 1999). Higher numbers of murrelets are seen in winter than in summer in Zone 2, especially in southern areas. Given that dispersal of Zone 3 birds may occur only into southern Zone 2 waters and may hot have occurred at the time of the spills, we assumed that all birds killed by the *Nestucca* oil spill belonged to the Zone 2 sub-population. However, it is possible that a few birds from Zone 3 were killed.

  The *Tenyo Maru* oil spill occurred in July, and many juveniles were reported killed (22 of 45 recovered; Warheit 1996). Most of this spill occurred in Zone 2, but some oiling also occurred in Zone 3. We considered that about 40 of 45 birds were probably recovered in Washington and belonged to the Zone 2 population. Current estimated mortality of 1 murrelet/yr in Zone 2 is below 1% of the estimated sub-population size in 2000-02 (800-2,600) (Huff et al. 2003) but past oiling levels (13.8-40.4/year) may have reached 1-5%, potentially causing significant additional population decline and shortening the time to extinction (Beissinger 1995a). Large numbers of murrelets killed during

the *Tenyo Maru* oil spill may have had additional impacts, including reduced breeding success or loss of nesting in certain forested areas with few nesting birds.

- Conservation Zone 3: Two poorly described oil spills occurred in Zone 3 between 1977-92 that likely killed murrelets, perhaps many, but seabird mortality was not properly assessed. In 1999, the *New Carissa* oil spill killed 262 murrelets, the largest number estimated for any spill in the 1993-2003 period. This large mortality may have had additional impacts beyond reduction in population size, including reduced breeding success or loss of nesting areas used by small numbers of birds. Murrelets wintering in Zone 3 appear to belong largely to the resident Zone 3 population but little information is available on dispersal movements. Post-breeding dispersal of small numbers of Zone 4 birds into Zone 3 has been demonstrated with radio telemetry (Hebert and Golightly 2003). We assumed that most or all murrelets killed in Zone 3 belonged to the Zone 3 population. Current (24.8/yr) and past mortality levels (5.1-34.1/year) were below 1% of the estimated population size in 2000-02 (5,500-6,700; Huff et al. 2003).

- Conservation Zone 4: Two large oil spills occurred in Zone 4 between 1997 and 1999 but none were reported between 1977 and 1992. Murrelets in Zone 4 in winter appear to belong largely to the Zone 4 population. Post-breeding dispersal is thought to occur largely within the zone, although a few birds were found to enter Zone 3 waters (Hebert and Golightly 2003). Little information is available on movements later in fall or winter. Many Zone 4 birds are resident based on evidence of extensive winter attendance of nesting areas in this zone (Carter and Erickson 1988, 1992, O'Donnell et al. 1995). Thus, most birds killed in this area in winter probably belong to the Zone 4 population. Current mortality levels (27.0/year) are below 1% of the Zone 4 population as estimated in 2000-02 (3,900-4,900) (Huff et al. 2003). The 2 larger spills may have had greater impact on the Southern Humboldt region than on the Del Norte-Northern Humboldt region because both spills occurred just outside of Humboldt Bay and killed murrelets mainly between the mouth and Big Lagoon, which is about 22 miles [35] km to the north. This area is closest to nesting habitat in the Southern Humboldt region and is more likely to be used by murrelets from this region, especially when attending nesting areas in summer and winter. The *Stuyvesant* oil spill occurred during the pre-basic molt period (September) when murrelets do not attend nesting areas and may have included birds from both nesting regions within Zone 4.

- Conservation Zone 5: No oil spills with seabird mortality or chronic oiling have been noted in most of Zone 5 (i.e., north of the Bodega Bay) in 1977-2003. However, oil from some spills (e.g., *Puerto Rican*, *Apex Houston*, Point Reyes Tarball Incidents, *Command*) off the Golden Gate and areas farther south in Zone 6 moved north into Zone 5, and dead seabirds have been found over a wide area from Bodega Bay to Monterey Bay. Single murrelets were recovered in Zone 5 during the *Puerto Rican* and Point Reyes Tarball Incidents, but the associated assessments of mortality did not make separate estimates for murrelets killed per zone. All murrelets estimated killed in the *Apex Houston* and *Command* spills were assigned to Zone 6. Murrelets are not known to breed in the small remnant old-growth forest patches in Marin and Sonoma counties, and few are seen at sea during the breeding season. Almost all of the 100-300 murrelets estimated in 2000-02 (Huff et al. 2003) were found in the northern part of Zone 5. A few radio-marked murrelets from Zone 6 breeding areas have dispersed as far north as central Zone 5 but no further (Peery et al. in prep.; Burkett, unpubl. data). We consider that the small numbers of murrelets found in winter in southern Zone 5 likely are mainly birds dispersed from Zone 6. Small numbers of Zone 4 birds may disperse, mainly into the northern part of Zone 5.

- Conservation Zone 6: Several spills with large estimated mortality occurred in Zone 6 during both the 1977-92 and 1993-2003 periods. Murrelets in Zone 6 appear to be largely resident, and extensive winter attendance of nesting areas has been noted (Carter and Erickson 1988, 1992; Naslund 1993; Carter et al. 2003). A few radio-marked birds have been noted moving as far north as central Zone 5 and as far south as San Luis Obispo County (Peery et al. in prep.; Burkett, unpubl. data), beyond the current southern boundary of Zone 6 (Service 1997). There is no evidence of fall or winter dispersal of murrelets into Zone 6 from other zones, but small numbers from Zone 5 or 4 could move into this area without detection (Carter and Ercikson 1992, Carter et al. 2003). In this review, we assumed that all murrelets killed in Zone 6 belonged to the Zone 6 population. Current (15.2-23.0/yr) and past (6.0-19.3/year) levels of oiling mortality were between 1-5% of the Zone 6 population as estimated in 2000-02 (487-637) or using likely higher earlier numbers (e.g., 1,000-2,000). Thus, oiling mortality is thought to have significant additive effects to this small, declining population, which may reduce the time to extinction and possibly prevent recovery (Beissinger 1995a, Carter et al. 2003). Large mortality from the Point Reyes Tarball Incidents (122 murrelets) resulted from prolonged oiling over several months related to leakage from a sunken vessel. This mortality may have

had additional impacts beyond population reduction, including reduced breeding success and loss of use of nesting areas used by few birds.

Overall, the threat of oil mortality has been reduced to some degree since the late 1990s by increased regulation. However, increases in shipping traffic (including oil tankers and other ships) and continued oil pollution will likely outweigh the short-term benefits of increased regulation in the future. Aging sunken vessels also may contribute to additional oil spillage. If offshore oil development occurs, the risk of oil pollution could increase substantially although improvements in safety should greatly reduce the risk of a very large spill. A very large or "catastrophic" oil spill (e.g., the general magnitude of the *Exxon Valdez* oil spill) has not yet occurred in northern or central California, Oregon, or Washington (Carter and Kuletz 1995, Service 1997, Carter 2003). While the threat of a very large spill is very low, such spills have the potential to kill most of a zone population, with complete extirpation of murrelets from portions of zones (or entire Zones 2, 5, and 6) where populations are already small. Much additional work is needed to better assess the risk of different sizes, locations, and timing of oil spills to marbled murrelets, including a systematic investigation of all sources of oil pollution. Effective mitigation projects to restore murrelet populations and efforts to reduce future oil pollution are needed.

5.4.4 Effects of By-Catch from Gill-Nets and Other Fisheries

Marbled murrelet and seabird bycatch mortality from coastal gill-net fishing has been considered a significant conservation issue in central California (Zone 6), northern Washington (Zones 1 and 2), British Columbia, and southern Alaska (Carter and Sealy 1984, Takekawa et al. 1990, Wilson 1991, Carter et al. 1995, 2001, Piatt and Naslund 1995, Service 1997, Melvin et al. 1999). Gill-net fishing has been prohibited and bycatch has not been a significant factor for murrelet populations in Oregon and northern California (Zones 3-5). Murrelets also can be killed by hooking with fishing lures and entanglement with fishing lines (Carter et al. 1995). However, these forms of mortality appear to occur sporadically in localized areas.

5.4.4.1 Mortality from Gill-Nets

When murrelets dive underwater for foraging or escape purposes, they can be entangled and drown in monofilament nets. In British Columbia and Alaska, most murrelets are killed in near-surface salmon nets left out for several hours at night, at shallow depths (<33 feet [10 m]), and in small numbers per net (Carter and Sealy 1984). In central California, murrelets also have been killed in halibut gill-nets set at deeper depths (often on the

bottom), with mortality at recorded depths of 62 and 89 feet (19 and 27 m) (Carter and Erickson 1992). Dead murrelets can be retrieved by fishermen and kept on boats (i.e., for lawful provision to authorities or unlawful disposal on land) or discarded overboard. Dead birds can float at sea for several days, allowing them to be carried to shore by winds and currents. Few birds are recovered on beaches, presumably because they are removed rapidly by scavengers, and few carcass counts are conducted on a regular basis in many coastal areas where such mortality might be detected. Estimates of gill-net mortality can be determined using fishery observer data on seabird bycatch and detailed information on fishing effort in different areas. Such fishery observer programs are expensive, difficult to coordinate, inconvenient for fishermen, sample a small proportion of nets fished, and have not been implemented in many areas or sustained over time. On the other hand, fishing effort data are regularly collected for fishery management purposes.

The chief long-term murrelet population impact from gill-net mortality is reduction in population size. However, reduced breeding success and loss of use of certain nesting areas also may occur in the short or long term. When only small numbers of birds are nesting in certain areas, loss of a few birds can lead to loss of use of those nesting areas, if other birds do not recolonize these areas. The significance of gill-net impacts to murrelet populations depends on whether or not gill-net mortality increases the rate of decline and speeds the time to extinction for affected populations. Regardless of the level of oil impacts, they are additive to other factors negatively affecting murrelet populations. The following sections describe the effects of fisheries on marbled murrelets in northern Washington (Conservation Zone 1), western Washington (Zone 2), central California (Zone 6), and British Columbia/Alaska. Little to no gill-net fishing occurs off the coasts of Oregon and northern California (Zones 3-5), so there are no related effects on murrelets in these areas.

Northern Washington (Conservation Zone 1)

Little information was available on murrelet mortality from all net fishing in Washington prior to the 1990s although it was known to occur (Carter et al. 1995). Potential for significant entanglement was noted by Troutman et al. (1991). After the species was listed in 1992, several tribal and non-treaty studies were conducted in 1993-96 and low entanglement rates were found. A series of fisheries restrictions and changes followed these studies to address mortality of all species of seabirds, before fishing effort decline due to reduced salmon catches. Available information on the series of events

that led to lower mortality of marbled murrelets in gill-nets in Washington since listing is summarized below.

- 1993-94: In August 1993, murrelets in northern Washington were observed mainly in the San Juan Islands, western Fidalgo Island, Cypress Island, and northern Hood Canal. Reports of gill-net related murrelet mortality in 1993 include the following:

    - The Service (1996) reported an estimated 12 murrelets entangled in the Makah set gill-net fishery near Cape Flattery in 1993. Grettenberger et al. (in prep.) examined 7 murrelets killed in Washington gill-nets in 1993, but only 1 was noted as killed in northern Washington fisheries. The other 6 birds were from tribal fisheries near Cape Flattery (J. Grettenberger, pers. comm.).

    - Beattie and Lutz (1994) noted no marbled murrelets killed in a 1993 tribal fisheries low-effort observer program in northern Washington waters (San Juan Islands, southern Straits of Georgia, Strait of Juan de Fuca, and northern Puget Sound) mainly from August-November.

    - Similarly, Pierce et al. (1994) reported that no marbled murrelets were entangled in monitored nets in the August-November 1993 non-treaty fishery low-effort observer program in northern Puget Sound and Hood Canal.

    - Craig and Cave (1994) documented 1 murrelet killed in 1993 and 1 in 1990 in a low-effort test fishery at the south entrance to the San Juan Islands, although most fishing occurred too far from shore to catch murrelets.

  In August and September 1994, marbled murrelets were mainly noted off Orcas Island, Lopez Island, and Fidalgo Island. Reports of gill-net related murrelet mortality in 1994include the following:

    - Lummi Nation (1994) reported no marbled murrelets killed in a tribal fishery observer program in the San Juan Islands and southern Strait of Georgia in 1994.

    - Pierce et al. (1996) reported on the 1994 tribal and non-treaty observer programs in the San Juan Islands and southern Straits of Georgia. One marbled murrelet was entangled (and released alive) in the San Juan Islands; the estimated entanglement rate was 0.00045 birds/gill-net set (1/2,222 sets), and 15 murrelets

(confidence interval=2-59) were estimated to have been entangled in 1994.

- Erstad et al. (1996) reported no marbled murrelets killed during the fishery observer program in October-November 1994 in Hood Canal and northern Puget Sound, but murrelets were observed only in northern Hood Canal. Observer coverage was calculated to be 10% of 500 fishing boats.

- 1995-98: In 1995, the Service authorized take of murrelets in tribal and all-citizens fisheries but required certain measures to reduce mortality (Wolf et al. 1996). In 1995, the WDFW voluntarily implemented closures of non-tribal fisheries in certain portions of fishing areas in the San Juan Islands, east mainland areas, and Hood Canal to protect murrelets. Later, reduced gill-net openings and fishing hours, restricted fishing to daylight hours, and required investigation of alternative gear also were implemented voluntarily by WDFW for non-tribal fisheries. Area and night closures to reduce murrelet and other seabird mortality had been suggested in Barkley Sound, British Columbia, and had been used in central California (Carter and Sealy 1984, Takekawa et al. 1990, Wild 1990).

  The WDFW did not close important fishing areas on the south side of the San Juan Islands in 1995. With restrictions, however, mortality in the non-treaty gill-net fishery should be in the lower end of the confidence interval range of 2-59 birds killed in the San Juans since 1994. These closures and restrictions have remained in place since 1995 (J. Grettenberger, pers. comm.).

  Alternative fishing gear and factors affecting seabird entanglement (e.g., time of day) were investigated in 1994-96 (Melvin et al. 1997, 1999). Three basic approaches to reduce seabird mortality without reducing fish catch in Puget Sound were recommended: (1) abundance-based or ecosystem management; (2) alternative gear; and (3) time of day. Alternative gear included traditional monofilament nets modified with visual alerts (i.e., panels of multifilament net in upper 7 or 15 feet [2 or 5 m]) or acoustic alerts (i.e., pingers attached to surface floats). Alcid (common murre and rhinoceros auklet) entanglements were reduced with alternative gear. However, only 1 marbled murrelet was killed in monitored nets in 1996, and no conclusion could be drawn regarding the efficacy of the modified nets in reducing murrelet entanglement. Marbled murrelets averaged 0.03 sightings/set and 0.01 encounters per set. The study did not occur in areas closed to protect murrelets and may

not have included many areas close to shore where murrelet abundance is higher. Merizon et al. (1997) noted concentrations of murrelets in Hood Canal and Burrows Bay in fall 1995 and 1996 which might be susceptible to tribal gill-net mortality. Elsewhere in northern Puget Sound, marbled murrelets occurred in relatively low numbers or were not present. In 1997, the Washington Fish and Game Commission required alternative gear (i.e., visual alert within the upper 7 feet [2 m] of multifilament net) and stopped nocturnal and dawn fishing for all-citizens (non-treaty) fisheries. Alternative gear may not have greatly reduced mortality of marbled murrelets, but closures and stopping nocturnal and dawn fishing may have (Carter and Sealy 1984). Additional study of alternative gear is needed to determine its effectiveness for marbled murrelets. Such studies may need to be conducted in British Columbia or Alaska where capture rates are higher and murrelet-specific closures do not currently exist.

Reported and estimated gill-net related murrelet mortality in 1995-1998 included the following:

- No murrelets were observed entangled in the northern Washington Makah fishery in 1995-96, but 3-6 murrelets were assumed to have been killed in 1996 (Service 1996).

- 6-9 murrelets were anticipated to be killed in fishing areas west of Port Angeles to Cape Flattery in 1996-98 (Service 1996).

- In northern Washington, a total of 34 murrelets were anticipated to be entangled annually from 1996-98 with projected fishing effort, but actual mortality was likely less due to lower fishing effort (Service 1999).

- 1999-2003: Tribal fisheries were not affected by 1995 regulations although some fishermen likely abided by some of these restrictions. A total of 16 murrelets (range 13-28) were estimated to be killed in 1999 in all treaty fisheries in northern Washington (Service 1999). This level of mortality was considered to have a significant adverse impact on a declining population. An incidental take permit was issued with conditions to: (1) develop a tribal plan in 2000 to reduce the potential for bycatch in 2000-03; and (2) provide the Service with fishing effort and murrelet bycatch information each year.

The Service issued consultations that anticipated the harm of 570 murrelets, primarily as a result of entanglement in gill-net fisheries in the Conservation Zone 1 (Puget Sound) between 1992 and 2003. The anticipated harm from

gill-nets is based upon the maximum allowable fisheries. Therefore, the anticipated harm of 570 murrelets is probably an overestimate because the maximum fishing effort did not occur due to the declining salmon populations.

Western Washington (Conservation Zone 2)

Murrelet entanglements occurred in the Makah gill-net fishery just north of Cape Flattery, but none were noted south of Cape Flattery (Service 1996). Some murrelets may be killed in these areas. Jefferies and Brown (1993) reported on seabird entanglement in the Columbia River, Willapa Bay, and Grays Harbor in 1991-93. No marbled murrelets were recovered but unidentified birds may have included murrelets.

Central California (Conservation Zone 6)

Extensive murrelet mortality from gill-net fishing was noted in central California between 1979-87, with an estimated 175-300 birds killed over this period (Carter and Erickson 1992). In 1987 and 1990, gill-net fishing was greatly reduced by regulations to reduce mortality of seabirds and marine mammals and murrelet mortality may have ended at this time (Takekawa et al. 1990, Wild 1990). The fishery continued after 1990 in portions of Monterey Bay and farther south off Morro Bay, but no murrelets were reported killed, probably because few if any murrelets occurred in permitted fishing areas (Julian and Beeson 1998, Forney et al. 2001). In 2000, the fishery was closed through emergency action. In 2002, this fishery was further prohibited, and gill-net fishing in waters less than 60 fathoms (i.e., 100 m within the depth used by murrelets) from Point Reyes to Point Arguello has ended (CDFG 2002).

British Columbia and Alaska

Little new information is available on levels of gill-net fishing mortality of marbled murrelets in Alaska and British Columbia since Carter et al. (1995) and Piatt and Naslund (1995) (K. Kuletz, pers. comm.; D. Bertram, pers. comm.). Large gill-net fisheries still occur in British Columbia, which have potential for killing large numbers of murrelets marbled murrelets (Hull 1999, Burger 2002). In a test chum salmon fishery off Nitinat (southwest Vancouver Island), 4 murrelets were killed in 1997 and none in 1998. Kuletz et al. (2003) reported 4 murrelets killed in set gill-nets between June and August 2002 near Kodiak, Alaska, and a total of 56 $\pm$ 27 (SD) murrelets were estimated killed (K. Kuletz, pers. comm.). None were previously known to be killed in gill-nets near Kodiak Island (Carter et al. 1995). Gill-net

mortality continues in southeast Alaska and Prince William Sound (K. Kuletz, pers. comm.).

Previous summaries (Carter et al. 1995, Piatt and Naslund 1995) reported mortality estimates but did not include raw numbers of murrelets recovered in Alaska fishery observer programs in 1990 and 1991, which would be useful for general comparison to Washington programs. In 1990, Wynne et al. (1991) reported 31 marbled murrelets, 2 Kittlitz's murrelets, and 3 unidentified murrelets recovered in monitored gill-nets in the May-September Prince William Sound driftnet fishery, and 1 marbled murrelet was recovered in the June-July South Unimak driftnet fishery. In 1991, Wynne et al. (1992) reported 16 marbled murrelets, 7 Kittlitz's murrelets, and 1 unidentified murrelet recovered in monitored gill-nets in the May-August Prince William Sound driftnet fishery, and a total of 260 marbled murrelets were estimated to have been killed in this area in 1991. Piatt and Naslund (1995) reanalyzed 1990 and 1991 data and estimated 775 and 600 marbled murrelets killed, respectively, in these years in Alaska.

Given the large numbers of murrelets likely killed in British Columbia and Alaska, extensive programs are needed to determine mortality levels. Such programs have not been implemented due to the cost of fishery observer programs; the lack of ESA requirement to assess and reduce fisheries mortality; potential impacts to fishermen from fisheries restrictions; and competing needs for funding for assessing forest-related impacts to the marbled murrelet. The ESA listing of the marbled murrelet in California, Oregon, and Washington stimulated the need for estimation of mortality levels in Washington, but the complex nature of non-treaty and treaty fisheries, various fishery restrictions (needed for more immediate partial solutions), and cost of fishery observer and gill-net assessment programs have made it very difficult to estimate mortality each year and over time. The difficult Washington experience may have impeded development of similar mortality assessment programs in British Columbia and Alaska, even though mortality is likely much greater north of Washington (Carter et al. 1995).

5.4.4.2 Gill-Net Fishing Effort

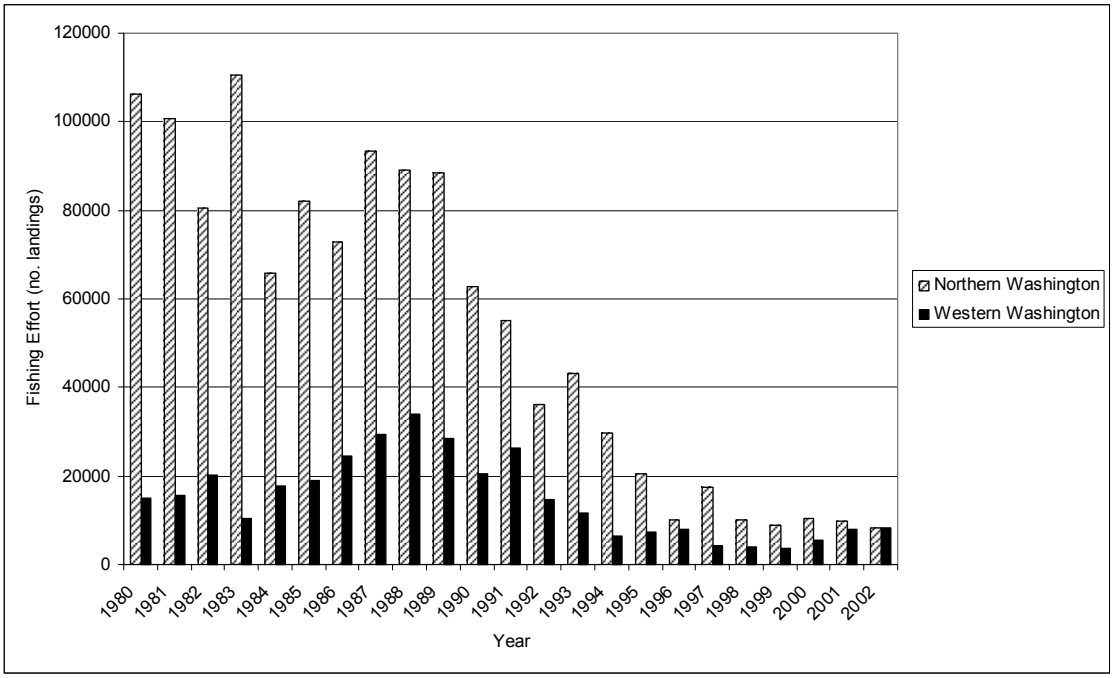
Information on gill-net fishing effort for treaty and non-treaty fisheries in Washington from 1980 to 2002 was provided by WDFW to the Service for this assessment (Figure 5.4-1). Effort data can be used to estimate potential levels of gill-net mortality with the use of appropriate entanglement rates for different fishing areas. Specific potential levels of mortality were not calculated because: (1) entanglement rates have been estimated only in certain years, fisheries, areas, and times; (2) much of the available information is

from before fishing restrictions and use of alternative fishing gear; and (3) data on murrelet abundance and distribution indicate a patchy, variable, and declining population that likely leads to much variation in the potential for entanglement in all areas. A detailed analysis of fishing effort, known entanglement data, and murrelet distribution and abundance over the past 25 years is needed to estimate past and current mortality in all fisheries but such an analysis is beyond the scope of this review.

For a broad assessment of the current risk of gill-net mortality versus pre-listing risk, we summarized effort alone since it is likely to be positively correlated with overall mortality (Carter and Sealy 1984). Between 1980 and 1991, gill-net fishing effort in northern Washington (catch areas 5-13) was high and varied between 55,000-108,000 landings per year (note: landings correspond roughly with number of days fishing with 1 landing/day on average) (J. Grettenberger, pers. comm.). From 1992-97, fishing effort decreased and ranged between 10,000-43,000 landings per year. In 1998-2002, effort was low and ranged between 8,000-10,000 per year. In western Washington (catch areas 1-4), effort also was high between 1980 and 1991 (13,000-35,000 per year), moderate from 1992-96 (6,000-15,000), low from 1997-2000 (4,000-6,000), and moderate in 2001-02 (~8,000/year). In general, fishing effort decreased by a factor of 5-10 fold between the 1980s and the late 1990s due to lower catches, fewer fishing vessels, and greater restrictions (e.g., shorter fishing openings) (Figure 5.4-1). Beattie and Seiders (2003) also noted that total gill-net effort for tribal fisheries in northern Washington waters (including the Strait of Juan de Fuca, San Juan Islands, southern Straits of Georgia, and northern Puget Sound) declined by 57% (31-82% between regions) between 1993-97 and 1998-2002. A similar reduction appears to have occurred in the non-treaty fishery during this period, although some increase was noted in 2002-02, which may signal a regrowth in gill-net fishing, if salmon stocks increase.

5.4.4.3 Mortality From Purse Seines

Three murrelets were observed encircled in the non-treaty purse seine fishery observer program in 1993-94, but all were released unharmed (NRC 1995, Carter et al. 1995). This fishery operates in the San Juan Islands, Puget Sound, and Hood Canal. No further studies were conducted. Purse-seine encircled murrelets also have been observed in Alaska, but no murrelet mortalities have been noted (Carter et al. 1995). Openings in the cork-lined tops of nets are now required in Washington to minimize potential mortality, as suggested in Carter et al. (1995).



Source: A. Hagen-Breaux, pers. comm.

Figure 5.4-1. Gill-net fishing effort (in landings) in 1980-2002 for non-treaty and treaty fisheries in: (1) northern Washington (Conservation Zone 1), and (2) western Washington (Conservation Zone 2).

5.4.4.4 Effects of Gill-Net Mortality

In Washington, there is sufficient information to indicate that the number of murrelets killed in gill-nets for tribal and non-treaty fisheries has probably reduced since the 1980s due to lower fishing effort and fishery restrictions. Little solid evidence is available to estimate levels of mortality in any year due to the complicated nature of fisheries, difficulty of obtaining suitable observer data, and clumped and variable distribution and abundance of marbled murrelets in northern Washington waters during the fall and winter.

To provide a general assessment of the potential significance of gill-net mortality versus other factors affecting the species in this review, we estimated that about 30 murrelets were killed per year from 1993-2003, using the 1999 estimate of 16 murrelets/year in tribal fisheries and the 1994 estimate of 15 birds/year killed in tribal non-treaty fisheries. This approach likely underestimated the total number of murrelets killed and, given the many difficulties of determining mortality levels (e.g., low-effort observer programs), we felt that this level of mortality was a minimum. For example, while 16 murrelets were estimated killed in 1999 in tribal fisheries, up to 28 could have been killed based on the upper end of the confidence interval of the estimate. Similarly, while 15 murrelets were estimated killed in 1994, up to 59 birds could have been killed based on the upper end of the confidence

interval of the estimate. While the 1999 tribal estimate may be low compared to mortality in the early 1990s, the 1994 estimate also may be high for the late 1990s. This estimate also excludes some areas where take has not been calculated by the Service. Some overlap existed in combining 1994 and 1999 estimates for extension over the 1993-2003 period, but we felt that overall biases led to 30 birds, indicating underestimation rather than overestimation of numbers of murrelets killed. We chose not to use the lower estimate of 15 birds from 1994 because later evidence indicated that this estimate was too low.

For the period from 1980-1992, we estimated an average annual mortality of at least 120 murrelets per year. This estimate was based on 2 factors: (1) the fishing effort in the years between 1980 and 1991 was at least 4 times higher than the annual effort after 1991; and (2) during high fishing effort, more boats may have fished farther from shore or in areas where few murrelets occur. Using this rough estimation technique, 1,440 and 360 murrelets may have been killed in 1980-91 and 1992-2003 periods, respectively. The ESA listing of the marbled murrelet, related fishing restrictions, and changes in fisheries have acted to reduce murrelet and other seabird mortality in gill-nets in Washington.

Many gill-net killed murrelets in northern Washington waters probably are resident members of the Zone 1 population, and this mortality probably has significantly contributed to population declines prior to and after listing. However, some gill-net killed murrelets in Zone 1 likely were from British Columbia or Zone 2. Post-breeding birds appear to move from the exposed coastlines along the outer portions of Juan de Fuca Strait and British Columbia into more sheltered waters in Puget Sound, San Juan Islands, and Straits of Georgia, causing larger numbers in protected inner waters in fall and winter than during the breeding season (Campbell et al. 1990, Rodway et al. 1992, Speich et al. 1992, Burger 1995). Some dispersal to sheltered waters occurs in July and August (prior to pre-basic molt), and some occurs in September to December (after pre-basic molt). Gill-net fishing occurs mainly between August and November, such that both resident and immigrant birds are exposed to fishing-related mortality (but resident birds more so). In the San Juan Islands/Northern Puget Sound region, Speich et al. (1992) found that numbers were about 1.2-1.6 times higher in winter (3,400 birds; November-March) than in summer (2,100-2,900 birds; April-July and June) in 1978-79. An increase was not noted in southern Puget Sound (i.e., south of Seattle) and seasonal changes on the outer coast could not be determined. Most dispersal movements from Barkley Sound, British Columbia (on the exposed outer coast of Vancouver Island), occurred in late July and August, although small numbers molted and may have remained

through the winter or dispersed after molt (Carter and Stein 1995). One murrelet banded in June in Desolation Sound, British Columbia, was recaptured off the San Juan Islands in August (Beauchamp et al. 1999).

To provide a general assessment of the potential impacts of gill-net mortality on Washington populations for this review, we assumed that:

- Birds from different populations are killed in proportion to their likely occurrence in fishing areas in fall;
- Zone 1 murrelets are resident year round;
- The ratio of birds in winter versus summer in 1978-79 (1.2-1.6:1) is generally representative throughout the 1977-2003 period;
- The ratio of birds in northern and southern Puget Sound in April-July 1978-79 (i.e., 4.4:1 or 81% northern) is generally representative; and
- Zone 1 population estimates in 2000-02 (5,600-9,700 murrelets; Huff et al. 2003) reflect general population sizes during the 1993-2003 period.

With these rough assumptions, a winter population of 5,400-12,571 can be roughly derived for the San Juan Islands/Northern Puget Sound area, including about 900-4,700 (~17-37%) from non-Zone 1 areas as follows:

- 5,600 (.81) = 4,536 murrelets in main fishing areas in San Juan Islands/Northern Puget Sound (minimum estimate; rounded to 4,500).
- 9,700 (.81) = 7,857 murrelets in main fishing areas in San Juan Islands/Northern Puget Sound (maximum estimate; rounded to 7,900)
- 4,500 (1.2) = 5,443 murrelets in winter in main fishing areas (minimum estimate; rounded to 5,400).
- 7,900 (1.6) = 12,571 murrelets in winter in main fishing areas (maximum estimate; rounded to 12,600).
- 5,400 – 4,500 = 900 murrelets in main fishing areas from non-Zone 1 populations (minimum estimate).
- 12,600 – 7,900 = 4,700 murrelets in main fishing areas from non-Zone 1 populations (maximum estimate).
- 900/5,400 = 17% of winter murrelets in main fishing areas from non-Zone 1 populations (minimum estimate).

- 4,700/12,600 = 37% of winter murrelets in main fishing areas from non-Zone 1 populations (maximum estimate).

Thus, current gill-net mortality of about 30 birds per year may remove about 19-25 murrelets per year from the Zone 1 sub-population, corresponding to less than 1% of the current estimated size of the sub-population, calculated as follows:

- 30 (.17) = 5.1 murrelets from non-Zone 1 populations (minimum estimate; rounded to 5 birds).

- 30 (.37) = 11.1 murrelets from non-Zone 1 populations (maximum estimate; rounded to 11 birds).

- 5/9,700 = 0.05% of Zone 1 populations are killed (minimum estimate).

- 11/9,700 = 0.11% of Zone 1 populations are killed (maximum estimate).

Some murrelets likely originated from the Zone 2 population (especially those killed near Cape Flattery in Zone 1) and some likely originated from British Columbia populations (especially those killed in southern Straits of Georgia and San Juan Islands). We assumed that about half of 5-11 murrelets (i.e., 2.5-5.5; approximately 4 murrelets) killed were from Zone 2 and the other half (i.e., 2.5-5.5; approximately 4 murrelets) were from British Columbia populations.

Beissinger (1995b) noted that declining population projections begin to differ greatly when bycatch exceeds 1% of the population. Thus, current gill-net mortality in Washington does not appear to be affecting apparent population declines to a great degree. In 1980-92 in Zone 1, annual mortality averaging 76-100 murrelets per year may have amounted to 1-2% per year of the population if its size was similar to that of 2000-02. However, population size probably was substantially higher in 1980-92 than in 2000-02, and that mortality likely remained less than or equal to 1% per year. Lower expected mortality of murrelets from Zone 2 and British Columbia populations appears to be well below 1% per year in both 1993-2003 and 1980-92.

While data are lacking on specific proportions of gill-net killed murrelets from Zone 1 and other areas (a proportion of which also likely varies between months, areas, and years), this rough assessment serves to identify the relative degree of potential impacts of gill-net mortality within Zone 1 on Zone 1, Zone 2, and British Columbia populations. Future work is needed to better determine actual mortality levels and population proportions.

In central California, 1987 restrictions kept gill-net fishing out of areas with large numbers of murrelets. No mortality is expected to have occurred since 1987, and there was very little gill-net fishing in this area prior to 1979. From 1979-1987, a total of about 150-300 murrelets were killed in gill-nets; highest mortality apparently occurred in the winter of 1980-81 when over 100 murrelets were recovered. Zone 6 murrelets are thought to be largely resident in central California during winter, and few if any murrelets from Zones 4 or 5 are thought to disperse south into Zone 6 (Carter et al. 2003). We consider that essentially all murrelets killed by gill-net fishing in Zone 6 were likely from the population in this zone. The Zone 6 population is quite small, and removing mortality from gill-net fishing would be expected to increase the time of population survival and allow greater time for recovery. Unfortunately, the only past population estimate for Zone 6 (i.e., 225 breeding birds in 1979-80; Carter and Erickson 1992) was probably inaccurate. Estimates of 487-615 birds were generated for 1999-2003 (Z. Peery, pers. comm.). We assume that this population has declined significantly since the late 1970s. If the Zone 6 population in 1979 had been 2-4 times higher than in 1999-2003 (i.e., about 1,000-2,000 birds), average annual mortality of 17-33 murrelets from 1979-87 would have exceeded 1% of the population per year. However, it is also possible that most mortality (~250-300 murrelets) occurred in 1 winter (1980-81), prior to the 1982 restrictions that prevented continued high mortality in northern Monterey Bay (Takekawa et al. 1990, Wild 1990).

Overall, the threat of gill-net fishing mortality of marbled murrelets has been reduced throughout the listed range since 1992. In Zones 1 and 2, impacts continue and are additive to other problems. We suggest that additional work is needed to better determine current bycatch levels and estimates of past mortality. Additional closures may be needed in nearshore fishing areas where relatively large numbers of murrelets are aggregated and mortality may be highest. While fishing effort is currently low, there is some indication that effort may increase in the near future if salmon stocks rebound.

5.4.5 Effects of Marine Contaminants

Fry (1995) indicated that the principal threat of marine contaminants to marbled murrelets is reproductive impairment from food web bioaccumulation of organochlorine pollutants (e.g., pesticides DDT/DDE, dieldrin, kepone, chlordane, methoxychlor, and dicofol; herbicides 2,4-D and 2,4,5-T; polychlorinated biphenyls [PCB]; polychlorinated dibenzo-dioxins; and polychlorinated dibenzo-furans) and heavy metals (e.g., cadmium, arsenic, mercury, lead, selenium, zinc). Such marine pollutants result primarily from pulp mill, industrial, and river discharges into marine waters

(and entrainment in sediments) where marbled murrelets feed and prey species concentrate. In the listed range, chief areas of contamination are the southern Strait of Georgia, Fraser River mouth, Puget Sound, Columbia River mouth, Humboldt Bay, and San Francisco Bay.

Little information is available about potential impacts on marbled murrelets because few studies have examined pollutant levels in body tissues, eggs, and prey. There is no information on the pollutant levels in prey or body tissues harmful to murrelets, but there are data on other avian species. Relatively low levels of DDE, PCB, cadmium, arsenic, mercury, lead, and zinc and relatively high levels of selenium have been found in small samples of murrelets from northern Washington and British Columbia, but high pollution areas near major sources were not examined (Noble and Elliot 1986, Grettenberger et al. in prep.). Relatively high mercury levels were found in marbled murrelets collected north of Vancouver, BC, in 1968-69 (Fimreite et al. 1971, Rodway et al. 1992). Monitoring of pollutants in pigeon guillemots in Washington has shown higher levels of industrial pollutants (e.g., heavy metals, petroleum hydrocarbons, PCB, PCDD, and PCDF) in the Seattle area than in the Strait of Juan de Fuca (Calambokidis et al. 1985, Fry 1995). However, pigeon guillemots forage largely on small benthic fish (with higher pollutant exposure), whereas marbled murrelets forage mainly on midwater fish (with lower pollutant exposure). Reproductive impairment from marine pollutants is not expected over wide areas because murrelets are distributed mainly in areas with lower pollutant threats, have wide foraging areas with seasonal dispersal, and feed extensively on transient juvenile and subadult midwater fish species expected to have low pollutant loads. Levels of PCBs, dioxins, and furans also appear to be declining in British Columbia (Burger 1995). However, murrelets that feed regularly in localized areas near major pollutant sources may be significantly affected. Specific studies are needed in high threat areas to better assess possible impacts.

5.4.6 Effects of Disturbance from Recreational Boating and Research and Monitoring Efforts.

In coastal and offshore marine environments, vehicular disturbance (e.g., boats, airplanes, personal watercraft, etc.) is known to elicit a behavioral response in murrelets of all age classes (Kuletz 1996, Speckman 1996, Nelson 1997). Aircraft flying at low altitudes and boating activity – in particular, motorized watercraft – are known to cause birds to dive and are thought to especially affect adults holding fish (Nelson 1997). Although such disturbance in marine environments is known to affect small-scale activity and distribution (Kuletz 1996, Speckman 1996), it is unclear how this may affect the distribution and movements of regional populations.

Hamer and Thompson (1997) conducted the only empirical study of marbled murrelet response to the presence of a motorboat during line-transect surveys at sea. Study methodology relied upon an "independent observer" documenting murrelet response while 2 "standard observers" surveyed for birds from the same research vessel. Of the murrelets for which paired observations were made, 68% (n=50) were found to move away from the approaching motor vessel. The majority of birds moved between 32-64 feet (10-20 m), although 11% of birds retreated greater than 96 feet (30 m).

No research on marbled murrelets has empirically correlated disturbance in marine environments with effects on either large-scale regional population distribution or reproductive success. The opportunities for research are limited by the inability to systematically track large numbers of breeding birds from marine environments to isolated nest sites. In addition, the unique breeding biology of the murrelet also makes it difficult to compare to other seabird species that have been studied to determine potential effects of long-term sublethal disturbance in marine environments. Disturbance from pleasure craft around breeding islands has been suggested to have caused the collapse of common murre and Atlantic puffin (*Fraterculus arctica*) breeding populations in Norway (Nettleship et al. 1985), but these birds nest in colonies adjacent to marine waters. Studies of common loons (*Gavia immer*), which are solitary nesters, have correlated decreased nesting and reproductive success, with increased development and recreational disturbance near lakeshore breeding grounds (Heimburger et al. 1983). However, this species nests on floating platforms and islands, which would be expected to be disturbed by boating and other water-based recreational activities. The murrelet nests high in trees, typically at some distance from the marine environment, and boating activities would not be expected to disturb nesting. Evidence suggesting adaptation and accommodation by individual marbled murrelets to recreation and other human disturbance (Golightly et al. 2002) does not account for or quantify the general large-scale effect of long-term sublethal disturbance on the fitness of regional populations.

5.5 Summary and Conclusions

This section summarizes the primary conclusions related to marine habitat for the marbled murrelet, as drawn from a review and evaluation of research and records from 1992-2003.

5.5.1 Marine Habitat Characteristics and Prey Availability

Breeding and corresponding at-sea populations of marbled murrelets occur from Attu Island in the Aleutians east and south to central California. The species' distribution appears to be almost continuous in suitable marine

habitat (protected, shallow waters) in the coastal waters of Alaska and British Columbia, but south of Vancouver its occurrence becomes sporadic, with 3 large gaps. Abundance decreases from an average of 272 murrelets per mile (169 murrelets per km) of coastline in Alaska, to 43 per mile (27 per km) in British Columbia, to about 18 per mile (11 per km) in the stretch from Washington to central California. Murrelets in Alaska and British Columbia are affected by conditions of the Alaska Current and Alaska Coastal Current, while those south of Vancouver Island are affected by the vagaries of the California Current.

Ocean climate is far more unstable (higher interannual frequency of anomalous conditions) in the California Current than in waters to the north. These different ocean conditions lead to geographic differences in diet, with sand lance and herring predominating in the Gulf of Alaska, and anchovy, smelt, and juvenile rockfish predominating to the south. The diet of marbled murrelets in the California Current is mostly inferred from indirect evidence. In both regions, crustaceans are prevalent in the diet during the winter. On a longer time scale related to the Pacific Decadal Oscillation, these 2 current systems exhibit contrary cycles in the prevalence of forage and other fishes, and presumably these conditions affect murrelets differentially as well.

5.5.2 Marine Threats

The primary quantifiable threats to marbled murrelets in the marine environment are oil spills and by-catch from gill-nets. Changes in food webs and prey availability from complex natural and human-related factors likely have profound effects on murrelet, but these factors are difficult to quantify.

5.5.2.1 Oil Spills

Overall, oil tanker and shipping traffic increased into west coast ports during the 1990s. However, the U.S. Oil Pollution Act, which was instated in 1990, has generally reduced the number of oil spills. Nonetheless, oil spills have occurred since 1992, with documented mortality of seabirds, including marbled murrelets. The number of spills and estimated annual mortality estimated for each Conservation Zone is shown in Table 5.5-1. The number of spills and their effects vary by zone, with at least some decrease in annual oil spill mortality rates shown for all zone since 1992 except for Zones 4 and 5, which remained the same and increased, respectively. For all zones except Zone 6, levels of current mortality from oil spills are less than 1% of the zone populations and are not thought to have had significant effects on populations. Although the threat of oil spills is thought to have been reduced since the marbled murrelet was listed in 1992, increases in shipping

Table 5.5-1. Annual mortality from oil spills before and after 1992.

Conservation Zone	Annual Mortality (Number of Birds)	
	1977-1992	1993-2003
1	2.5-6.5	1-2
2	14.8-41.4	1
3	6.5-35.1	24.8
4	1	27
5	None	None
6	6.6-22.4	14.3-18.5
Totals	40.4-107.6	68.1-73.3

traffic and/or off-shore drilling could increase the risk to this species in the future.

5.5.2.2 By-Catch from Gill-Nets

Marbled murrelet bycatch mortality from coastal gill-net fishing has been considered a significant conservation issue in central California (Zone 6), northern Washington (Zones 1 and 2) (Carter and Sealy 1984, Takekawa et al. 1990, Wilson 1991, Carter et al. 1995, 2001, Piatt and Naslund 1995, Service 1997, Melvin et al. 1999). Gill-net fishing has been prohibited in Oregon and northern California (Zones 3-5), and bycatch has not been a significant factor for murrelet populations in these areas.

In central California, gill-net fishing has been prohibited since 1987 from areas where large numbers of murrelets occur, so fisheries associated mortality since that time has been nonexistent. In Washington, the best estimate of annual loss from gill-net fishing is at 30 birds/year from 1993-2003; previously (1980-1992), up to 120 birds annual may have been killed from this fishing practice. Since 1995, WDFW has voluntarily implemented a number of practices to limit by-catch from gill-nets in non-tribal fisheries. These measures, combined with decreased fishing effort due to declining salmon runs, are thought to have reduced the threat to marbled murrelets in Zones 1 and 2. However, gill-net fishing mortality continues and is additive to other marine problems.

6.0 DISCUSSION AND SUMMARY OF CONCLUSIONS

6.0 DISCUSSION AND SUMMARY OF CONCLUSIONS

Since listing in 1992, substantial research on the marbled murrelet has confirmed some information previously known and provided new data on population size and distribution, genetics, demographics, and habitat use. This chapter summarizes the new information on the murrelet since listing, as presented in Chapters 2-5, and provides an assessment of threats to the species. Previously referenced literature is not necessarily re-cited in this chapter.

6.1 Summary of New Information

The following is a summary of what we've learned since the marbled murrelet was listed, particularly since completion of the USFS's "Ecology and Conservation of the Marbled Murrelet" (Ralph et al. 1995) and the Service's "Marbled Murrelet Recovery Plan" (Service 1997). The new information is organized by the 4 major listing factors in the ESA 4(a)(1)(A-E).

6.1.1 Habitat

Our understanding of the habitat requirements of the marbled murrelet has not significantly changed since listing. Within the listed portion of the range in California, Oregon, and Washington, it has been confirmed that murrelets primarily use old-growth coniferous trees for nesting and nearshore marine waters for foraging. In the Aleutian Islands and Alaska Peninsula, Alaska, birds nest solely on the ground where no old-growth coniferous trees occur. From Kodiak Island through southeast Alaska, birds appear to nest primarily in old-growth trees although some ground-nesting also occurs. In British Columbia, tree nesting predominates, but a few ground nests and 1 nest in a deciduous tree have been found recently. Throughout Alaska and British Columbia, birds forage primarily within several kilometers of shore although some birds forage farther from shore. This section summarizes new information on the characteristics of terrestrial and marine habitats. It also includes discussion on the distribution of terrestrial habitat and the effects of forest fragmentation.

6.1.1.1 Terrestrial Habitat Associations

The distance inland of nests from coastal feeding areas is variable and appears to be influenced by a number of factors including foraging and nesting habitat availability, climate suitability, maximum foraging range, availability of freshwater lakes, and predation rates. While murrelets have been recorded and may nest up to 62 miles (100 km) from the coast in Washington and British Columbia, most murrelets in the 3-state area appear to nest within 37 miles (60 km) of the coast. The Service (1997) considers 50 miles (81 km) as the minimum inland distance for determining habitat amount and suitability within the listed range. Topics related to terrestrial habitat associations include habitat characteristics, survey methods, and survey effectiveness.

Habitat Characteristics

Since the marbled murrelet was listed in 1992, studies of terrestrial habitat suitability from British Columbia to California have consistently confirmed that murrelets generally select old-growth forests for nesting. Habitat characteristics at the landscape, stand, and tree scale are summarized below.

Landscape Scale

- Studies to determine nesting habitat characteristics at a landscape scale often show murrelet use to be positively associated with: (1) total watershed area; (2) increasing amounts of late-seral forests; and (3) increasing age class and height class of forests. Use at a landscape level has been shown to be negatively associated with greater amounts of edge, increasing areas of logged and immature forests, and higher elevation (except in some areas of British Columbia where there seems to be a weak positive correlation with higher elevations).

- At the landscape scale, most predictive models indicated that the probability of murrelet occupancy or nesting was positively associated with stand age, tree height class, vertical canopy complexity, and basal area (larger tree diameters). Relationships to canopy closure, slope, distance to foraging (marine) areas, fragmentation level, and elevation can vary by region.

- Studies using audio-visual detection data to characterize murrelet nesting habitat at a landscape scale have often found murrelet use to be associated with: (1) the presence of mature and old-growth forests; (2) larger core areas of old-growth; (3) low amounts of edge (with 1

exception); and (4) lower fragmentation levels, and (5) proximity to the marine environment. In some cases, murrelet use was associated with lower elevations, more complex landscape patterns, and stands that were less isolated from other similar stands.

- Several studies using radar concur that marbled murrelets do not pack into higher densities within remaining habitat when nesting habitat is removed. Evidence indicates that the watershed populations of marbled murrelets are proportional to the areas of old-growth forest available and that logging portions of watersheds has a detrimental effect on murrelet numbers.

Stand/Nest Site Scale

At the forest stand and patch scale, data from audio-visual studies have shown that murrelets tend to occupy forest stands that have a complex structure, relatively large conifers, and a relatively large number of platform trees with epiphyte cover. Predictive variables at the stand/nest site scale included: (1) platform density; (2) higher epiphyte thickness and percent cover; (3) greater tree heights and canopy complexity (including number of canopy layers); (4) larger tree diameters; (5) densities of large trees; (6) elevation; and (7) slope.

Tree/Nest Scale

- Old-growth conifers tend to be the only trees that provide the necessary requirements for murrelet nesting including: (1) openings in the canopy for access to the nest; (2) large potential nest platforms (branches or deformities); (3) substrate for a nest cup; (4) horizontal and/or vertical cover over nest site; and (5) sufficient height to allow "drop take-off" departures and "stall drop-in" landings.

- Some exceptions to the sole use of unharvested old-growth for nesting have been noted:

    - In Oregon, nests have been found in young and mature trees (66-150 years in age) that exhibited similar characteristics to old-growth trees and were distinguished by the number of platforms provided by mistletoe infections.

    - In British Columbia, 1 nest has been found in a red alder tree, which is deciduous.

- In California, marbled murrelets make extensive use of residual old-growth forests (i.e., partly harvested forests with some remaining old-growth trees) and can be found in large or small residual stands.

- One confirmed and 2 probable cliff nests were documented in southwestern British Columbia.

- In Alaska, 15 ground nests have been found in a variety of locations since 1992.

Terrestrial Survey Methods

- Inland survey methods for marbled murrelets have changed substantially since the species was listed in 1992. The most recent PSG protocol recommends the use of radar as well as audio-visual surveys. Both methods have their strengths and weaknesses and can also be used in tandem to accomplish particular objectives.

- Factors that affect the ability of trained observers to detect murrelets at inland sites include weather, daily variation in detection rates, annual variation in detection probabilities, season, conditions at the survey site, (tree canopy closure, amount of visible sky), and distance from marine foraging locations. The approximate error in classifying sites as unoccupied when they were indeed truly occupied (false negative) is estimated to approach 15.1% over time for surveys conducted prior to 1998. The current error rate is approximately 4.2%, indicating substantial improvement in survey methods.

6.1.1.2 Habitat Distribution and Forest Fragmentation

Loss and modification of nesting habitat (older forests), primarily due to commercial timber harvesting, was 1 of the primary reasons for listing the murrelet as threatened. The amount and distribution of terrestrial habitat, the estimated loss of habitat since listing, and the effects of forest fragmentation, both human-caused and natural, are discussed below.

Terrestrial Habitat Amount and Distribution

- Based on information provided by land managers in the 3-state area, the Service estimates that there are currently 2,223,048 acres of suitable marbled murrelet nesting habitat. Of this amount, about 91% is located on Federal land; State, County, and private lands account for about 8%;

and Tribal lands contain about 1%. About 47% of the suitable habitat occurs in Washington, 35% in Oregon, and 18% in California.

- Many administrative units use northern spotted owl habitat definitions as a surrogate for murrelet habitat. Because northern spotted owl habitat is often defined at 80 years old and murrelet habitat typically does not develop by that time, the current estimate of 2.2 million acres is almost certainly an overestimate of suitable habitat.

- Extrapolating from survey results, it is estimated that about 820,768 acres, or 34% of the suitable habitat, is likely to be occupied by murrelets. In Oregon and Washington, however, the actual level of occupancy is not known in many watersheds, and the actual inland extent of nesting is not known at any location in the 3-state area. It is unclear whether the estimated amount of habitat available is roughly correct or has been under- or overestimated using current definitions of habitat. We still do not know if, in fact, murrelets use large areas of forest that have been defined as "habitat."

Habitat Loss Since Listing

- Estimated potential total loss of suitable murrelet habitat between 1992 and 2003, not counting degraded habitat, is approximately 226,080 acres, or about 10% of the current estimate of 2.2 million acres of suitable habitat. Of the more than 226,000 acres of suitable habitat estimated lost, approximately 7,370 acres were surveyed and found to be occupied by murrelets. In addition, the Service consulted on the removal of more than 10,000 potential nest trees.

- Based on the Section 7 and CDFG consultation records, habitat loss was greatly influenced by land ownership. Non-Federal lands account for 168,162 acres (80%) of the total habitat anticipated for removal under Section 7 consultation, while Federal lands accounted for 34,951 acres (17%). Tribal lands accounted for 7,649 acres (3%), the fourth largest loss of habitat by ownership. The largest single area of murrelet habitat loss was on ownerships covered by HCPs and accounted for 148,893 acres (71%). Because some of these HCPs are long-term plans, not all the acres included in the consultation records have been harvested at present, but it is anticipated that harvest of the permitted acres will occur in the near future (i.e., within a decade or two).

- The 2003 Biscuit fire in southern Oregon is estimated to have eliminated nearly 15,000 acres of murrelet habitat, representing about 87% of the habitat lost from natural events since the species was listed.

- Development of suitable murrelet nesting habitat in the future is generally dependent on trees attaining a size that supports large lateral branches. No projections exist that predict significant increases in suitable habitat acreage during the next several decades, although over the long term, habitat on Federal lands and some HCP lands should increase over time.

Effects of Forest Fragmentation

- While relationships between forest fragmentation, edge effects, and murrelet nest success are complex and likely to vary between areas, it is apparent that the abundance of some predators is higher and breeding success is lower in edge habitats (within 164-656 ft [50-200 m]), especially in proximity to human activities (e.g., parks, campgrounds, urban areas).

- Murrelets may prefer nesting along the edges of forests, with natural edges preferred over anthropogenic edges. The frequency of murrelet nests on edges may largely reflect the prevalence of forest fragmentation and natural edges. However, murrelets also have limited maneuverability during aerial flight, and edges may provide easier access to nests, both for adults during visits to and from the nest, and for chicks during fledging.

- There are no data on the specific effects of microclimate changes from fragmentation or distance inland on murrelet nesting habitat at the branch and tree scales. However, increased solar radiation, wind, and temperatures in the forest canopy could change the distribution of epiphytes, remove moss from nesting platforms, or cause overheating of eggs, chicks, or incubating adults.

- The 3 demographic parameters most likely to be affected by forest structure and landscape condition are: (1) nesting success, (2) adult survivorship while commuting to and from nests, and (3) proportion of adults that breed.

- Murrelets appear to abandon highly fragmented areas over time. In addition, at-sea densities of murrelets tend to be higher adjacent to areas of low fragmentation.

- In fragmented landscapes, murrelet nesting stands may be more productive if surrounded by buffers of simple structured forests and by minimizing the effects of human recreation and settlement on predation levels.

- In extensive mature forest landscapes, murrelet productivity will best be enhanced by maintaining large blocks of complex-structured forest far from human activity or with low levels of predators.

6.1.1.3 Marine Habitat

Marine habitat for the marbled murrelet is associated with the subarctic/temperate waters that characterize portions of the eastern North Pacific Ocean. The species' range overlaps 3 major global marine ecosystems: (1) the Alaska Current and the associated fiord ecosystems in southern Alaska, British Columbia, and northern Washington; (2) the California Current along the open ocean coasts of western Washington, Oregon, and California; and (3) the Aleutian North Slope Current, composed of subarctic gyre waters mixing with Bering Sea waters around the Aleutian Island chain.

In the California Current, marbled murrelets mainly occur at sea near remaining clumps of old-growth forest nesting habitat during the breeding season. A variety of marine habitats can be used for foraging (e.g., ocean bays, river mouths, sandy shores, and nearshore submarine canyons), but the primary factor affecting at-sea distribution appears to be nesting habitat. In general, this suggests that murrelets are generalist feeders, and various prey resources are widely available in nearshore waters.

Because murrelets feed almost exclusively in the marine environment, they may be affected by ocean cycles that can influence the abundance and availability of fish. Gill-net fishing and oil spills also affect murrelets. The following sections summarize the factors affecting the species in the marine environment.

Effects of El Niño–Southern Oscillation (El Niño) and Pacific Decadal Oscillation (PDO)

- Prey resources for murrelets are typically most abundant in spring and summer in the California Current, during the breeding and pre-basic molt periods for murrelets. The California Current is dramatically affected by El Niño events, which causes a northward infusion of warm water from equatorial regions along the west coast of North America, which can negatively affect certain prey resources depending on the timing and magnitude of the event. Certain prey species (e.g., juvenile fish associated with estuaries) are likely less affected by El Niño events than prey species in exposed coastal waters. In addition, certain prey species (e.g., sardine) can increase during these events.

- The negative effects of El Niño events increase with decreasing latitude: effects are most pronounced for other seabird species in California, whereas El Niño effects are rarely reported north of Washington. In some events, El Niño effects are felt 1 year later in Oregon and Washington than in California.

- Major El Niño events occurred in 1983, 1992, and 1998, and minor El Niño events or other warm water events occurred in some other years (e.g., 1978). In strong warm-water events, some seabird species forego breeding entirely, and in weaker ones reproduction in certain seabird species can be reduced significantly (fewer birds breed, and greater numbers fail in their breeding attempts). Feeding areas and foraging behavior also may change during warm-water events. However, severe El Niño impacts (e.g., complete abandonment of nesting by all birds or large die-offs) do not appear to occur in marbled murrelets as found in some other seabirds. Marbled murrelets may be partly insulated from severe El Niño impacts because they are generalist nearshore feeders and use a variety of prey resources. In addition, some prey resources may not be affected by warm-water conditions. Somewhat reduced breeding is likely during severe El Niño events, but it is unclear whether this results from lower breeding success or less adults attempting breeding. Nest failure rates are high in all years such that reduced breeding success during severe El Niño events is not obvious with available data, and successful fledging of chicks has been documented during major and minor warm-water events. Little data are available on the diet of marbled murrelets in the California Current during "normal" nor anomalous ocean conditions.

- The PDO can potentially affect prey availability for seabirds on a scale of decades. Effects of PDO (i.e., a shift from predominantly warm to predominantly cold waters and vice versa) on marbled murrelets are unknown. South of Cape Flattery, Washington, certain fish populations and zooplankton abundance have declined during the most recent warm phase of the PDO (1977-1998), with concurrent negative effects on certain seabirds. While prolonged periods of lower prey availability may lead to lower overall breeding success, as they do in most seabirds, data do not exist to evaluate whether or not this is the case for marbled murrelets, especially given low breeding success under good conditions.

Prey Availability

- Like many seabirds, marbled murrelets likely prey on the most available, suitable fish and crustaceans. Knowledge of diet is mainly from studies

in British Columbia and Alaska, with little information available on diet in the California Current. Geographical variation in diet probably mirrors the small- and large-scale geographic variation in the availability of prey. Species composition of available prey changes to some degree between the Alaska Current and fiord ecosystems compared to the California Current. Nothing is known about murrelet diet in the Aleutian Islands.

- Diets of marbled murrelets in the Alaska Current and fiord ecosystems are dominated by sand lance, herring, and capelin. Diets of birds in the California Current are probably dominated by surf/night smelt, northern anchovy, and herring, but no detailed study has been conducted. Invertebrates (e.g., mysids, euphausiids, and squid) likely are consumed infrequently during the breeding season and only fish are delivered to chicks at the nest, but invertebrates may be eaten to a great degree in the non-breeding season.

- Primary prey species of the marbled murrelet (e.g., sand lance, herring, smelt, anchovy) are of slight or no commercial fishery value in Washington, Oregon, and California. Small fisheries for Pacific herring exist, but depression of prey resources due to commercial fishing has not been demonstrated (nor is it suspected), although destruction of certain spawning beds has occurred in parts of Puget Sound due to non-fishing activities. Because of declines in murrelet populations and populations of fish, such as herring, in certain parts of the range (e.g., southern Puget Sound), geographic overlap no longer exists between murrelet at-sea distribution and commercial harvest of their prey species (e.g., anchovy, herring). However, there is some potential for development of fisheries for anchovy and sardines in the future which may overlap with murrelet distribution.

Effects of Oil Spills, Gill-Netting, and Marine Contaminants

- Threats of oil spill and gill-net mortality vary by Conservation Zone.

- Marbled murrelet mortality from coastal gill-net fishing has occurred in central California (Zone 6), Washington (Zones 1 and 2), British Columbia, and Alaska.

- Gill-net fishing has been prohibited or has not occurred for many decades in Oregon and northern California (Zones 3-5). In central California (Zone 6 and farther south), gill-net fishing increased in the late 1970s, decreased by the late 1980s, and was prohibited in 2002.

- Marbled murrelet mortality from oil pollution has occurred in all zones, British Columbia, and Alaska.

- Adding spills and chronic oiling estimates, we found similar overall numbers of murrelets killed by oil in the listed range between the periods 1977-92 (29-101/year) and 1993-2003 (69-78/year). However, overall population size within each zone probably was much lower in 1993-2003 than in 1977-92 such that recent oil impacts likely affected a greater proportion of populations than previously.

- The chief long-term impact to murrelet sub-populations from oiling is reduction in population size, but decreased breeding success and abandonment of certain nesting areas also may result when relatively large numbers are killed.

- Regardless of the level of oil impacts, they are additive to other factors negatively affecting murrelet populations, particularly at the Conservation Zone scale.

- There is no evidence of negative effects of marine contaminants on marbled murrelets, but little information has been collected on this topic. If problems do occur, they will most likely affect populations in Puget Sound, Washington, and in the adjacent Straits of Georgia, British Columbia, where pollution would be expected to be highest within the murrelet's listed range.

6.1.2 Overutiltization for Commercial, Recreational, Scientific, or Education Purposes

There is no evidence that the marbled murrelet is overutilized for commercial, recreational, or education purposes. While individual birds may be affected by research efforts, such as telemetry projects or tree-climbing in nesting habitat, these disturbances are relatively small scale, occur infrequently, and are therefore unlikely to affect murrelet populations.

6.1.3 Predation and Disease

There is substantial new information on the effects of predation on marbled murrelets; there is also some additional information on the effects of disease, as summarized below.

6.1.3.1 Predation

Predation has consistently been the most significant cause of recorded nest failure at marbled murrelet nests. Recent findings on predator increases, as well as effects of forest fragmentation and human development on predation, are summarized below.

- In general, studies of avian predation have found: (1) higher nest predation in areas with high predator densities; (2) increased abundance or diversity of predators with increased variety and complexity of habitats; (3) increased abundance of some corvid species along edges or in forest fragments near human activities; (4) high nest predation by corvids along edges near human activities or in areas of low forest cover; and (5) high predation by small mammals in a variety of habitats including interior forests and along non-natural edges.

- Populations of several corvid species have increased dramatically in western North America as a result of forest fragmentation, increased agriculture, and urbanization. Within the listed range, a large percentage of suitable marbled murrelet habitat is in close proximity to human activity, which likely increases the potential for predation at nest sites by corvids.

- Predation by mammals at murrelet nests has not been documented but is thought to contribute to nest failure. Experimental work with artificial nests has confirmed that squirrels and mice are likely predators of small murrelet chicks.

- The most important factors in the risk of predation of murrelet nests seem to be landscape context or composition (including proximity to human activities) and its effect on the type and abundance of predators present. While predator foraging efficiency has also been implicated as a factor affecting predation, this has yet to be demonstrated.

- Predation of adults by raptors (e.g., falcons and hawks) during the breeding season is the most well-documented cause of natural mortality. Predation on murrelets by goshawks, at least in the northern parts of the range, might be more common than previously thought.

- Predation on adults could be a serious problem since demographic models indicate that adult survivorship has a greater impact on murrelet population growth than juvenile survival or nesting success (productivity). This does not necessarily mean that adult survival is currently limiting murrelet populations.

6.1.3.2 Disease

No studies have been conducted on diseases in marbled murrelets, and little is known about their susceptibility or mortality caused by diseases. However, the recent emergence of bacterial, fungal, parasitic, and viral diseases and biotoxins has been shown to affect numerous populations of seabirds, but not alcids. Potential disease effects on murrelets are addressed below.

- There is a strong possibility that murrelets could be affected by 1 or more diseases or biotoxins in the near future because of the cumulative effects of stressors in both their marine and forest environments.

- The potential for murrelets to contract West Nile Virus seems high given their nesting habits, the abundance of mosquitoes in forests, the presence of the disease in other forest-nesting bird species, and the proximity of many murrelet nesting areas to open fields, clearcuts, or areas of human activity where corvids are abundant. However, disease transmission may be low due to nesting and feeding in largely solitary pairs.

- Because corvids seem to be the most susceptible to West Nile Virus, declines in these species may benefit murrelet populations, at least temporarily, in the future.

6.1.4 Other Natural or Manmade Factors Affecting the Murrelet's Continued Existence

Other factors affecting the marbled murrelet are related to demographic characteristics, population trends, taxonomy and range, ecological requirements, and noise disturbance.

6.1.4.1 Population Trends and Demographics

Key information on population trends and demographics is summarized below.

Population Trends

- Because murrelets are difficult or impossible to survey over large areas in forested nesting habitats, large-scale population surveys must be conducted at sea. Detecting long-term trends in murrelet populations is difficult due to the availability of adequate at-sea census data. However, changes in levels of murrelet use of local areas of forest or marine habitat can be determined with available techniques and suitable survey effort.

- Since 1972, major declines (22-73%) in populations over a period of a decade or more have been documented in Alaska, British Columbia, and Oregon with no evidence of increase throughout the breeding range.

- In central Oregon (Zone 3), an overall population decline of >50% was reported from standardized surveys conducted between 1992 and 1999. This major decline was verified with annual standardized data (Strong 2003a).

- Although data collected in 2000-02 for the Effectiveness Monitoring Program have shown higher mean numbers in some zones by 2002, the confidence intervals are very large and trends cannot be determined with 3 years of data. Many more years of data (likely a decade or more) are needed before significant trends can be detected.

Demographic Characteristics

- Murrelets are long-lived, with an estimated generation time of 10 years. Breeding probably begins at ages of 2-5 years.

- A murrelet clutch consists of 1 egg, although laying of replacement eggs is now known. Both adults incubate and feed the chick.

- Unadjusted or adjusted adult:juvenile ratios detected at sea, as an indirect index of breeding success, have suggested generally low breeding success from California to southern British Columbia. Within the 3-state range, lowest ratios were found in central California (Zone 6) and highest ratios in Washington (Zone 1). Recent telemetry studies of individual nests have found highest levels of breeding success in southern British Columbia (0.46 chicks/pair), moderate levels in northern California (0.135-0.324 chicks/pair), and very low levels in central California (0.0 chicks/pair). In general, both methods indicate that murrelet breeding success appears to decline from north to south. Except for central California, telemetry techniques have resulted in higher estimates of breeding success than found with unadjusted or adjusted adult:juvenile ratios (<0.13 chicks/pair).

Demographic Modeling

Models are increasingly being used to estimate seabird population trends, with deterministic Leslie Matrix models being the most common. To assess trends in marbled murrelet populations within the 6 Conservation Zones, we developed a new model, called the Zone Model. Findings are summarized as follows.

- Zone sub-populations will decline between 3 and 6% per annum during 2001-2010, similar to estimated decline (4-7%) from earlier models (e.g., Beissinger and Nur 1997). Greatest decline is forecast in Zone 6 and the least in Zone 2.

- Extinction probabilities (i.e., "extinction" defined as less than 30 individuals in a zone) of 100% occur within 40 years (i.e., by 2040) in Zones 5 and 6. Other sources also have previously identified these zones as having very high potential for extinction in the near future (Carter and Erickson 1992, Service 1997). Small populations in these zones are likely nearing or have already reached non-viability levels.

- Extinction probability is high in Zones 2-4 (0% within 40 years but 100% within 100 years by 2100). Only Zone 1 has a greater probability of remaining extant than becoming extinct over the 21st century (i.e., extinction probability of 25% by 2100). These populations will become non-viable some time before becoming extinct.

- In 100 years, mean population size for the listed portion of the range in Washington, Oregon, and California was projected to be 45 murrelets, with a probability of extinction of 16%. The listed population will likely be non-viable at this time.

- While projected trends and extinction probabilities of zone sub-populations may be alarming to some readers, these predictions are similar to other existing information and previous modeling efforts (e.g., Beissinger and Nur 1997, Service 1997).

- Sensitivity analyses for low immigration rates (0.1%, 2.0%, and 5.0%) demonstrated: (1) little effect on decline and probability of extinction in Zones 4 and 6; (2) higher rates of decline and earlier time to extinction in Zones 2 and 5; and (3) lower rates of decline and later times to extinction for Zones 1 and 3.

- Sensitivity analyses for higher fecundity from telemetry studies (0-54%) versus lower fecundity from adult:juvenile ratios (4-9%) demonstrated that using low fecundity values from adult to juvenile ratios: (1) led to higher rates of decline and higher and earlier extinction probabilities in Zones 1-3; (2) reduced time to extinction to about 20 years in Zones 5 and 6; and (3) resulted in Zones 3 and 4 having similar probabilities and times to extinction as Zones 1 and 2.

- Sensitivity analyses for oil and gill-net mortalities demonstrated: (1) reducing oil spill and gill-net mortalities can significantly reduce annual

rates of decline in Zones 2 and 6, but mortality reductions have less effect on decline in other zones; and (2) extinction probabilities for different levels of mortalities cannot be examined due to model structure.

- Catastrophic mortality from very large oil spills was considered unlikely and was not modeled but could cause extirpation of most or all birds within a zone, especially at low population sizes.

6.1.4.2 Taxonomy and Range

There have been several major new findings related to the taxonomy of the marbled murrelet and some minor refinements on the species range, as summarized below.

- Until recently, long-billed and marbled murrelets were considered to be 2 races of the same species despite morphological distinctions. Genetic research indicates that marbled and long-billed murrelets are genetically distinct and have probably been reproductively isolated for 5-6 million years. The AOU recognized the marbled and long-billed murrelets as separate species in 1997.

- The breeding range of the marbled murrelet extends from the western Aleutian Islands and northern Bristol Bay through central California, but major gaps occur in distribution occur in several locations (see Section 6.1.5.1). Small numbers of murrelets have been recorded as far north as the Chukchi Sea (with 1 record in Russia) and as far south as northern Baja California, Mexico. The current geographic center of the world population is considered to be in the northern part of southeast Alaska.

- Most nesting is thought to extend inland 40 miles (24.8 km) in Washington, 35 miles (21.7 km) in Oregon, 25 miles (15.5 km) in northern California (Zone 4), and 10 miles (6.2 km) in central California (Zones 5 and 6).

6.1.4.3 Life History/Ecology

Many elements of the general biology and ecology of the marbled murrelet have been confirmed through research conducted since listing in 1992. Information on variation in breeding chronology, inland behavior, nest attendance, site fidelity, and natal dispersal is presented below.

Breeding Chronology

Breeding for the marbled murrelet is less synchronous and spread over a more prolonged season than for most temperate seabirds. Egg-laying and incubation range from late March to mid-August in California and from mid-May to mid-August in Alaska. Many late nests probably reflect replacement eggs, especially in the southern part of the range. Most birds probably nest in the earlier parts of these ranges. The timing of breeding is undoubtedly affected by local ocean conditions and prey availability, but the details of these effects are poorly known.

Inland Behavior, Nest Attendance, Site Fidelity, and Natal Dispersal

- Results of a study in British Columbia found that early-nesting murrelets tended to travel farther from foraging areas to nest and used trees on steeper slopes than those nesting later. Although statistically significant, these correlation coefficients were relatively small. Early-breeding birds were more successful, as found in other alcids.

- Limited data suggest that re-use of nest sites in a subsequent season may be more common in areas where large old-growth trees are rare or where predators are reduced. At a larger landscape scale, murrelets do show fidelity to foraging areas and probably to specific watersheds for nesting, although some radar studies suggest some movement of birds among nearby watersheds from year to year.

- In general, marbled murrelets likely exhibit nesting behavior that is consistent with that of other adult alcids: high philopatry (i.e., once breeding age is reached, birds usually breed near where they were hatched) and high site fidelity (i.e., once breeding has occurred, adults usually return and breed at the same site or nearby sites over many successive years).

6.1.4.4 Noise Disturbance

Information on the effects of noise disturbance at marbled murrelet nest sites is largely based on relatively limited information.

- Murrelets typically exhibit only a limited, temporary behavioral response (if any) to noise disturbance at nest sites and are able to readily adapt to both ambient and specific local auditory stimuli.

- Murrelets near hiking trails and campgrounds showed no visible reaction to loud talking (or yelling) near nest trees. However, disturbance may be lower at nests in coast redwoods which, on average, have platforms that are higher off the ground than in Douglas-fir or western hemlock trees.

- Murrelet chicks appear unaffected by disturbances near nests. Adults in general do not appear to be affected by vehicular traffic, noise, or disturbance from nearby recreational activities (i.e., hiking, camping). Significant effects to nesting adults from survey activities are unlikely, but no direct studies have examined this issue.

6.1.5 Distinct Population Segment Topics

The RFP for the 5-Year Status Review asked for additional information related to evaluating the distinct population segment of the listed range of the marbled murrelet. Specific issues to be addressed include:

- Population abundance, distribution, and trends;

- Genetic, morphometric, behavioral, and ecological variation within the species, particularly differences between populations in the 3-state area compared to British Columbia and Alaska;

- Movement and dispersal;

- Variation in marine conditions that might suggest adaptations to local environments; and

- Variation in terrestrial conditions that might suggest adaptations to local environments.

Based on the review of available information on genetic, ecological, and behavioral differences within the breeding range, the global metapopulation of marbled murrelets should be considered to include at least 3 "distinct populations": (1) the Aleutian Islands or "northern" population; (2) the Alaska Peninsula to Puget Sound or "central" population; and (3) the California, Oregon, and western Washington or "southern" population. The designation of at least 3 populations is supported by genetics information, but insufficient genetics information is available to determine all possible distinct populations or the precise boundaries between distinct populations. We consider that interim boundaries for the 3 identified populations to date should be drawn at the major boundaries between 3 major global coastal marine ecosystems.

We consider that the northern, central, and southern populations each likely has a suite of different behavioral, ecological, and genetic characteristics related to population maintenance under very different ecological conditions within these 3 major global marine ecosystems. However, given that physical barriers between populations at boundaries are not great, a small degree of mixing likely occurs between populations and helps maintain genetic similarities despite ecological and behavioral differences.

Further subdivision of populations also is required for management and conservation purposes. Major changes in habitat ownership and management occur at the Alaska–British Columbia border and at the British Columbia–Washington border. It is often useful to refer to the "3-state" population, the "British Columbia" population, and the "Alaska" population. Use of these terms does not reflect biologically different populations, but these geographic regions do reflect major latitudinal segments of the breeding range with some corresponding biological differences. The southern and 3-state populations overlap almost entirely, except that Conservation Zone 1 is not included in the southern population but is included in the 3-state population.

6.1.5.1 Population Abundance, Distribution, and Trends

Population Abundance

The North American population of marbled murrelets is estimated at 947,500 birds, of which 90.7% (859,100 birds) were found in Alaska, 7.0% (66,500) in British Columbia, 1.0% (9,800) in Washington, 0.8% (7,502) in Oregon, and 0.5% (4,598) in California. Abundance decreases from an average of 272 murrelets per mile (169/km) of coastline in Alaska, to 43 per mile (27/km) in British Columbia, to about 17.5 per mile (11/km) in the stretch from Washington to central California. The current population size of marbled murrelets from Washington to California is considered to be far below historical levels, based mainly on high loss of old-growth forest nesting habitats and mortality from oil spills and gill-net fishing. At present, these populations constitute only 2.3% of the total estimated world population.

Population Distribution

At-sea distribution during the breeding season appears to be almost continuous in suitable marine habitat (protected, shallow waters) in the coastal waters of southern Alaska (i.e., southern Alaska Peninsula, Cook Inlet, Prince William Sound, to southeast Alaska) and British Columbia (i.e.,

mainland coast from the Alaska border to northern Straits of Georgia and northern and western Vancouver Island). However, large natural gaps in marine habitat or at-sea distribution (i.e., where few if any murrelets occur in comparison to areas on either side of the gap) occur in waters between the Aleutian Islands and between large offshore islands and the mainland (e.g., Kodiak Island, Queen Charlotte Islands, and southern Vancouver Island). At-sea distribution in Alaska and British Columbia is incompletely known because many areas have not been surveyed. Major gaps in distribution within the 3-state range, from north to south, are described below.

- The first major gap in at-sea distribution (i.e., few murrelets over a large area) occurs in the southern Straits of Georgia near and at the U.S. border. The eastern part of this occurs near the mouth of the Fraser River and southwest Vancouver Island, due mainly to the lack of nesting habitat, which has been removed by logging and urban development. The mouth of the Fraser River, which is characterized by extensive mudflats and silty water that extends to the Gulf Islands and San Juan Islands (including deeper waters of the Straits of Georgia), also consists of habitat that appears to be little used by foraging murrelets. This gap also occurs adjacent to a large natural gap in marine habitat between southwestern Vancouver Island and the northern Olympic Peninsula (i.e., Juan de Fuca Strait) where no murrelets occur in deep waters. Both the non-natural and natural gap in the southern Straits of Georgia and the natural gap in Juan de Fuca Strait correspond roughly with the boundary between British Columbia and Washington, which also forms the northern boundary of Conservation Zone 1.

- A second major gap in distribution occurs in southern Puget Sound (Conservation Zone 1 – southern portion) where large urban development has occurred, little nesting habitat remains, and few murrelets occur.

- A third major gap in distribution with low numbers of murrelets occurs off southwest Washington, Columbia River mouth (i.e., with silty waters that are little used for foraging), and northern Oregon, where little nesting habitat remains and foraging is affected by major estuary habitats. This gap roughly corresponds with the boundary between Conservation Zones 2 and 3 at the Columbia River.

- A fourth major gap in distribution with low numbers of murrelets occurs from southern Humboldt County to the Golden Gate, apparently due mainly to little remaining nesting habitat. This gap corresponds roughly with Conservation Zone 5.

- A fifth major gap in distribution occurs in southern Santa Cruz and Monterey counties, where no nesting habitat apparently remains but likely occurred in the past. No nesting habitat occurs south of Monterey County. The southern end of the historic breeding range likely occurred in Monterey County. The southern boundary of Conservation Zone 6 occurs in northern Monterey County.

In Alaska and British Columbia, at-sea distribution during the non-breeding season is very different than during the breeding season, with most murrelets moving from outer exposed waters to inner protected waters (e.g., Kodiak Island, Straits of Georgia).

The at-sea distribution of murrelets during the breeding season within each of the Conservation Zones is summarized below.

- Conservation Zone 1: Murrelets have a clumped and variable distribution in the San Juan Islands, northern Puget Sound, and along the north coast of the Olympic Peninsula within foraging distance of nesting habitat on the northern and eastern Olympic Peninsula, northern Cascade Mountains, and possibly southern British Columbia. Few birds occur and may nest in southern Puget Sound. Numbers increase in northern Puget Sound in the non-breeding season.

- Conservation Zone 2: Murrelets occur primarily between Grays Harbor/Willapa Bay to Cape Flattery near nesting habitat on the northwestern Olympic Peninsula. Few birds occur in southwestern Washington during the breeding season, but numbers increase during the non-breeding season.

- Conservation Zone 3: Murrelets are continuously distributed off remnant nesting habitats, but most birds are found off central Oregon and low numbers off northern Oregon.

- Conservation Zone 4: Murrelets are reasonably continuous in Del Norte and northern Humboldt counties, although highest numbers are found off of the largest clumps of old-growth forest nesting habitat, found in national and state parks. Smaller numbers are found in southern Oregon and south of Cape Mendocino.

- Conservation Zone 5: Small numbers of murrelets breed in small patches of remnant old-growth forests in Mendocino County, but breeding is not suspected in Sonoma or Marin counties.

- Conservation Zone 6: Murrelets are concentrated in waters off San Mateo and Santa Cruz counties, with highest numbers adjacent to the largest areas of old-growth forest habitat, primarily found in state parks. Few birds occur in Monterey County.

Populations in Washington, Oregon, and California extend over about 17 degrees of latitude (about 32° to 49° N), versus only about 12 degrees of latitude (about 48° to 60° N) in Alaska and British Columbia. However, Alaska populations also extend over about 60 degrees of longitude (about 130° W to 170° E), although about half of these longitudes encompass the Aleutian Islands (about 160° W to 170° E). The coastline of southern Alaska (east of the Aleutian Islands) and British Columbia extends for about 4,685 miles (7,540 km), exclusive of inlets, bays, and fiords. The linear distance along the Aleutian Islands is 1,139 miles (1,833 km), and the linear distance of the coastline within the listed range is approximately 1,280 miles (2,060 km). Thus, the northern, central, and southern populations account for 16%, 66%, and 18% of the linear breeding range, respectively.

Population Trends

Information on population trends was presented previously in this chapter in Section 6.1.4.1.

6.1.5.2 Genetic, Morphometric, Behavioral, and Ecological Variation

A review of the recent literature suggests genetic, behavioral, and ecological variation between murrelets over their range; there are virtually no data on morphological differentiation. The following sections summarize variation within the species as currently understood.

Genetic Variation

- The existence of private haplotypes/alleles at high frequency in the Aleutian Islands and Californian samples suggests that gene flow between murrelets in these areas and British Columbia/mainland Alaska is restricted. Genetic information for sub-populations in Washington, Oregon, and northern California is not yet available.

- Murrelet populations in the Aleutian Islands and California may represent Holocene (post Ice Age) range extensions with population differences arising recently *in situ* due to restricted gene flow.

- Statistically different allele frequencies within populations in the Aleutian Islands and California indicate genetic differentiation from populations in the central part of the range. Genetic variability does not appear to be reduced in either area.

- Genetic variability in all types of markers that have been screened in marbled murrelets is similar to other species, with no evidence of either population genetic bottlenecks or inbreeding.

- Inbreeding depression and interspecific hybridization are not an immediate threat, although further studies should be done. Genetic variation in neutral molecular markers in marbled murrelets is similar to that in other species of seabirds, including several species with large and/or increasing population sizes; thus, population-level variation is not an immediate concern.

- Given that murrelets from California, British Columbia/mainland Alaska, and the Aleutian Islands differ genetically, loss of any of these populations will likely reduce the species' genetic resources and may compromise its long-term viability. Furthermore, if differences in use of nesting habits are genetically based, loss of either type of behavior would represent a loss of adaptive variation.

- Given the current small size, relative isolation, reduced-quality nesting habitats, and a large gap (partly non-natural and partly natural) between central and northern California populations (across Zone 5), the southern edge population in central California is expected to be especially vulnerable to extinction.

- Alcid species with strong genetic structure, such as the marbled murrelet, will likely be slow to recover from a local disturbance since reproductive rate and immigration are low.

Morphometric

There have been very few studies on variation in murrelet morphology. Tree- and ground-nesting murrelets in Alaska do not appear to differ morphologically, and little geographic variation in morphology is currently known.

Behavioral and Ecological

Five primary behavioral differences occur in murrelets across their breeding range in North America:

These 5 primary behavioral differences appear to reflect major geographic changes in the availability of nesting habitat, length of breeding season, marine habitat, and prey species within the murrelet's range during the breeding and non-breeding seasons.

Limitations on research have prevented complete study of suites of differing behavioral and ecological characteristics (many of which have not yet been studied or described in detail for any population), but we must emphasize in particular that study of the Aleutian Islands population has been extremely poor such that many differences could have been overlooked with available knowledge. While we acknowledge that our understanding of differences between populations is rudimentary, substantial ecological variation has been detected with relatively low study effort to date, and we expect that greater differences will be detected with increased study over the next few decades.

6.1.5.3 Movement and Dispersal

Seasonal patterns of movement and dispersal differ between populations in Alaska and British Columbia compared to the listed range.

- In Alaska and British Columbia, at-sea distribution during the non-breeding season is very different than during the breeding season, with most murrelets moving from outer exposed waters to inner protected waters (e.g., Kodiak Island, Straits of Georgia).

- From Washington to central California, at-sea distributions during the breeding and non-breeding seasons are similar, although murrelets appear to be more widely distributed within Conservation Zones during the non-breeding season. Many birds continue to visit nesting areas in the non-breeding season in California, with small numbers visiting nesting areas in winter as far north as southern British Columbia.

- Small numbers of murrelets occur regularly south of Conservation Zone 6 in the non-breeding season, often to San Luis Obispo County and occasionally to northern Baja California, Mexico. Small numbers also appear to move north from Conservation Zone 6 into the southern part of Conservation Zone 5 in the non-breeding season.

- In Alaska and British Columbia, recent information on murrelet movements at sea and indications that flightless molting birds aggregate in isolated coastal areas suggest that regional populations may also be at risk from environmental stochasticity and human disturbance. However, in Washington to California, most birds appear to molt near nesting areas and do not form large molting aggregations in different areas.

6.1.5.4 Variation in Marine Conditions and Potential Adaptations

From Cape Flattery to central California (Conservation Zone 2-6), the coastal marine ecosystem is strongly influenced by the California Current, a few major rivers/estuaries (e.g., Columbia River, San Francisco Bay), and several smaller estuaries. Most of this coastline is very exposed to offshore weather and current conditions. From Cape Flattery to the British Columbia border (Conservation Zone 1), the coastal marine ecosystem is strongly influenced by estuarine waters derived from major fiord ecosystems, and to a much lesser extent waters offshore of the mouth of Juan de Fuca Strait. The extensive fiord and inlet system formed by Juan de Fuca Strait, Puget Sound, San Juan Islands, Straits of Georgia, Gulf Islands, and Vancouver Island provide extensive feeding habitats that are protected to varying degrees from offshore weather and current conditions.

In this review, we also have considered that the 6 Conservation Zones within the 3-state population represent "sub-populations" of marbled murrelets. Conservation Zones were originally considered by the U.S. marbled murrelet recovery team to be semi-independent demographic and geographic sub-populations where most or all birds that bred in the zone likely also fed within the zone during the breeding season and to a large degree during the non-breeding season. In addition, zones faced differing threats that should be managed separately, and were functional equivalents of recovery units. To promote viable populations of listed species and lower the risk of extinction, it was desirable: (1) to maintain multiple populations such that catastrophic events could not result in loss of the whole listed range of a species; (2) to increase population size within each zone; and (3) to prevent large gaps in distribution with potential lower ability of isolated populations to sustain themselves over time. Boundaries between zones were selected to reflect major marine and terrestrial geographic landmarks that served as approximate divisions between these sub-populations, with low at-sea densities of murrelets or little old-growth forest nesting habitat occurring at or near boundaries. In some cases, a political boundary occurred in the same area and was selected as the boundary to facilitate management.

Given these considerations, we made the following observations:

- We suspect that murrelets in Zones 2-6 belong to the "California" genetic population because: (1) these sub-populations occur within the California Current marine ecosystem (i.e., they use similar types of prey resources and nesting habitat, as well as experience similar annual variability in prey resources caused by variation in the California

Current); (2) little or no immigration into these sub-populations from Alaska/British Columbia is suspected; and (3) gaps in at-sea and nesting habitat distribution within Zones 2-6 appear to be related largely to changes in nesting habitat over the last century and do not appear to prevent low levels of immigration between adjacent zones over time.

- We suspect that murrelets in Zone 1 belong to the "Alaska/British Columbia" genetic population because: (1) similarity of fiord marine ecosystems used during breeding; (2) greater potential for immigration from British Columbia exists in Zone 1 (i.e., during the non-breeding season, an influx of birds from British Columbia occurs into northern Puget Sound, and small numbers of birds may immigrate to Washington rather than return to breed in natal areas in British Columbia); (3) gaps in at-sea distribution, nesting habitat, and marine habitats are not large enough to prevent limited exchange with British Columbia sub-populations; and (4) some birds that forage in northern portions of Zone 1 during the breeding season may actually nest in British Columbia.

- We suspect that little immigration occurs between Zones 1 and 2 except as noted below. While no large physical barrier exists in the general vicinity of Cape Flattery, immigration movements probably are limited because a major change in coastal marine ecosystems (including nesting habitats and prey resources) likely acts as a partial barrier.

- The terrestrial and marine boundary between Zones 1 and 2 is rough, and some birds breeding near the periphery of one zone could feed in the other zone. In particular, murrelets foraging east of Cape Flattery along the north side of the Olympic Peninsula in Zone 1 could actually belong to the Zone 2 population. Ocean waters in outer Juan de Fuca Strait also are likely more similar to outer coastal waters than inner waters. Additional studies (perhaps with radio-marked birds) are needed to examine the nature and exact location of the biological boundary between Zones 1 and 2.

6.1.5.5 Variation in Terrestrial Conditions and Adaptations

Across the breeding range of the marbled murrelet, there appears to be a significant source of variation at the stand level when comparing northern and southern latitudes. The choice of nesting habitat for marbled murrelets appears superficially to be broader in Alaska, where murrelets nest both in trees and on the ground. Data on the height and dbh of nest trees, and the height of nest limbs, also show clear regional differences. In Alaska, for example, total tree densities were greater, stand sizes were generally smaller,

tree diameters were smaller, and tree heights and nest branches were lower compared to nest stands in the Pacific Northwest. Nests from Vancouver Island, Oregon, and Washington occur in larger trees than those from the Sunshine Coast, Queen Charlotte Islands, and Alaska. In general, however, few differences in terrestrial habitat use were noted between populations from British Columbia south. Beginning in southern Oregon and extending through central California (Zones 4-6), coastal old-growth forests are dominated by coast redwood and to a lesser extent Douglas-fir. Redwood trees are taller and larger than other tree species used, nests occur higher, and there are fewer platforms per individual tree.

6.2 Objectives of the 5-Year Status Review

As stated in the Request for Proposal (July 2003), the 5-Year Status Review of the marbled murrelet has summarized information to answer 2 major questions:

(1) Does new information suggest that the species population is increasing, declining, or stable?

(2) Are threats to the species increasing, the same, reduced, or eliminated; and are there new threats?

The information needed to answer these questions is provided in Chapters 1 through 5 of this evaluation and summarized below.

6.2.1 Does New Information Suggest that the Murrelet Population is Increasing, Declining, or Stable?

The current population of marbled murrelets is estimated at 947,500 birds, with 90.7% (859,100 birds) in Alaska, 7.0% (66,500) in British Columbia, 1.0% (9,800) in Washington, 0.8% (7,502) in Oregon, and 0.5% (4,598) in California. The 3-state population represents about 18% percent of the species' linear range across latitudes and longitudes but contains only 2.3% of the world population size. Detecting long-term trends in murrelet populations is difficult due to the limitations of sampling techniques and availability of census data. However, demographic modeling with the most recent biological information indicates that the murrelet population is still declining in all 6 Conservation Zones.

Since 1972, major declines (22-73%) in populations over a period of a decade or more have been documented in Alaska, British Columbia, and Oregon. For the listed range, only in Oregon has a major decline been verified with

annual standardized data. In central Oregon (Zone 3), an overall population decline of >50% was reported from standardized surveys conducted between 1992 and 1999. This decrease is the strongest direct evidence of large-scale population decline in the 3-state range. There is no indication of a population increase in the listed range, although data from the last 3 years of the Effectiveness Monitoring Program (2000-02) show higher mean estimates in some zones in 2002. Given the large confidence intervals (i.e., high estimate variability), more years of data are needed before significant trends in any direction can be detected.

Results of modeling studies in California, Oregon, and Washington populations suggest population declines of 4 to 7% per annum. It appears that the very small populations in Conservation Zones 5 and 6 are likely at or near levels that are not self-sustaining (i.e., "non-viable") and therefore have the highest risk of extinction and shortest time to extinction relative to other zones. At this time, no significant improvements in breeding habitats are expected, such that poor breeding success will likely continue, a major factor affecting populations.

Since listing in 1992, suitable breeding habitat and number of occupied sites have declined throughout the 3-state range. The loss of additional available tree-nesting habitat (which is already severely reduced in some areas) and reduced quality of remaining habitat have likely led to lower breeding success, mainly due to increased nest and adult predation levels, especially near areas of high human use. This problem appears to be the most evident threat to murrelet population viability. Breeding populations of murrelets are predicted to continue to decline as areas of old-growth decrease. However, it is likely that murrelet populations will continue to decline even if the amount of nesting habitat remains stable and adult survival unchanged, due to already low levels of breeding success or further reductions in breeding success.

6.2.2 Are Threats Increasing, the Same, Reduced, or Eliminated; or are there New Threats?

Since listing, some threats to the marbled murrelet have increased, others have been reduced, and some have remained about the same. Habitat loss actually falls into 2 categories; while the rate of annual habitat loss has declined since listing, the effects of ongoing and past habitat loss on the murrelet remain much the same. There do not seem to be any threat categories that have been eliminated since listing. Research and surveys conducted since listing have, however, identified one possible new threat.

- Ground- versus Tree-Nesting – In the northern part of the range (Aleutian Islands, Bristol Bay, and Alaska Peninsula), tree-nesting habitat is unavailable, islands free of mammalian predators are common (although currently fewer than in the pre-Russian era due to introduction of predators to many islands), and only ground-nesting occurs. In the southern part of the range (Washington, Oregon, and California), no predator-free islands with suitable habitat are present and only tree-nesting occurs. In the central part of their range (southern Alaska and British Columbia), many islands with suitable habitat are present (most with mammalian predators but some without), and both ground- and tree-nesting occur, although little ground-nesting likely occurs in most of British Columbia.

- Changes in Key Nest Tree Species and Other Nest-Site Characteristics within the Breeding Range - In the central part of the range and northern section of the southern part of the breeding range, key tree species are Douglas-fir, western hemlock, and mountain hemlock, and nests tend to have high moss or epiphyte cover. In these areas, murrelets may be able to use smaller limbs due to epiphyte cover, especially near the end of the range of coniferous old-growth forests. In the southern part of the range, key tree species are coast redwood, Douglas-fir, and western hemlock and tend to have low or no epiphyte cover. In these areas, limb size may need to be greater (because the nest bowl must be formed by dorsal limb structure), or birds may select damaged limbs that create suitable nest bowl structure.

- Differences in Breeding Chronology - In Alaska and northern British Columbia, the breeding season is about 1-2 months shorter and later than from southern British Columbia to California. The incidence of replacement laying is likely lower in Alaska and northern British Columbia due to later breeding.

- Differences in Dispersal During the Non-Breeding Season – In Alaska and British Columbia, most murrelets disperse away from breeding areas during the non-breeding season, whereas in Washington to California, most murrelets are resident year-round with substantial winter visitation of nesting habitats, especially in California.

- Changes in Key Fish Prey Species - Key prey species during summer in the central part of the range are Pacific sand lance and Pacific herring, whereas prey species appear to switch to smelt and northern anchovy in the southern part of the range. Little is known of diet in the northern part of the range.

6.2.2.1 Increased Threat Level

The Recovery Plan recognized the vulnerability of murrelets to increased levels of nest predation associated with forest edges. However, it appears that the threat of predation on murrelets is greater than previously anticipated. Predation has consistently been found to be the most significant cause of nest failure, and corvids are implicated as the primary predator of murrelets. Recent studies have shown that most (>50%) active nests have failed and the majority (78%) of known nest failures are due to predation. Abundance and predation rates of avian predators, especially some corvid species, appear to be affected by edge and areas of low forest cover, as well as by landscape features such as type of matrix habitat and the proximity to human disturbance. The highest risk of predation has been documented in areas close to humans (<0.6 mile [1 km]). Food sources in these areas attract predators, particularly corvids. Based on new research, the number of species of avian nest predators may also be higher than previously thought.

In addition, the threat of high adult predation by raptors (that might reduce adult survival at the zone population level) may have increased in some areas (e.g., California) due to increased or recovering populations of falcons, eagles, and possibly some hawks. Predation on adults could be a serious problem since demographic models indicate that adult survival has a greater impact on murrelet population growth than juvenile survival or nesting success (productivity). This does not necessarily mean that adult survival is so reduced that it currently limits murrelet populations. In fact, few sources of natural adult mortality have been documented, and few cases of adult predation have been noted (although it is very difficult to document).

Threat of extinction in Zone 6 has been increased due to very poor breeding success, small population size, increasing predators/predation, and reduced murrelet use of Big Basin Redwoods State Park (E. Burkett, pers. comm.), the prime piece of remaining nesting habitat in central California. Previously, murrelets were thought to be almost extirpated in Zone 5, but recent surveys have demonstrated that a small population (100-300 birds) still exists and nests in small patches of remnant forest. This population must have survived over several decades, but whether or not it can survive for another century is doubtful, especially if adjacent zone populations are reduced to very low numbers.

6.2.2.2 Similar Threat Level

Two of the primary threats to murrelets that have remained relatively unchanged since listing are the effects of ongoing and past habitat loss and oil spill mortality, as described below.

Effects of Ongoing and Past Habitat Loss

Although the rate of loss has slowed since listing in 1992, the amount of old-growth forest suitable as breeding habitat and number of occupied sites have declined throughout the breeding range, increasing the most evident threat to murrelet population viability. This continued loss of murrelet habitat and occupied sites has been due primarily to timber harvest, salvage logging, windthrow, and wildfire. Timber harvest allowed under HCPs has continued to eliminate habitat in each of the 3 states. Consultations with non-Federal parties are ongoing in Washington, Oregon, and to a lesser degree in California, and these consultations involve the potential future removal of hundreds of acres of suitable murrelet habitat. Potential habitat loss from Service consultations with State and private entities encompasses the largest percent of habitat that could be lost or modified over time. Unpredictable stochastic events like the Biscuit fire in southern Oregon, forest diseases, insect outbreaks, and windstorms contribute to habitat loss and will likely continue to do so in the future.

The time since listing is too short to expect any measurable amount of nesting habitat development by 2003. In addition, little habitat is expected to be regenerated over the next several decades, since there are no projections that predict significant increases in murrelet habitat with the necessary structural characteristics for nesting (see Section 4.5.3.2). Development of suitable nesting habitat in the future is generally dependent on trees attaining a size that supports large lateral branches. Over the longer term, habitat on Federal lands (especially large second-growth forest portions of parks that already have been set aside [e.g., Redwood National Park in northern California]) and some HCP lands should increase over time. In addition, there are no projections about when or if murrelets may use new habitat in the future. Such use is dependent on many factors, especially the existence of viable populations, population size, and proximity to still-occupied future nesting habitat.

Overall, threats to marbled murrelet populations in the 3-state area from past and ongoing habitat loss likely remain unchanged since listing due to:

- Continued low reproductive success;

- Increased predation; and

- The low likelihood that additional habitat will develop and be used in the future.

Oil Spill Mortality

Despite reductions in oil pollution, oil spills continue to occur and kill relatively large numbers of seabirds, including murrelets. Although the threat of oil spills has been reduced since listing, the rate of oil spill mortality of murrelets has remained relatively constant. However, the sporadic nature of oil spills makes it difficult to compare pre- and post-listing mortality of murrelets. Recent improvements in oil spill mortality assessment methods post-listing and poor documentation prior to listing also make comparison difficult.

Since the U.S. Oil Pollution Act (1990) was instated, increased government regulations and industry efforts have led to lower numbers of large and small oil spills. In addition, offshore oil development in California (northern/central), Oregon, and Washington has not occurred, the 1992-2002 moratorium has been extended to 2012, and 4 National Marine Sanctuaries that prohibit offshore oil development have been established (3 in central California and 1 on the west coast of Washington). Continued increases in shipping traffic (including oil tankers and other ships) and continued oil pollution may outweigh the short-term benefits of increased regulation in the near future. Sunken vessels also may begin or continue to leak and, if offshore oil development occurs, the risk of oil pollution could increase. As zone populations shrink over time, the effects of oiling mortality of even small numbers of murrelets will increase.

Possible worst-case scenarios include the potential of a large oil spill (e.g., the size of the *Exxon Valdez* oil spill) occurring, which could lead to loss of 1 or more entire zone populations. However, to date, such a large spill has not occurred within the listed range, although the 1969 Santa Barbara oil spill was of a similar order of magnitude (in terms of amount of oil spilled) within the southern portion of the winter range, but no murrelets were recovered.

Additional Threats

Additional threats that are thought to have remained unchanged since listing include the following:

- Risks of unpredictable stochastic events like the Biscuit fire, forest diseases, insect outbreaks, flooding rivers, and windstorms are likely

similar to pre-listing. Fire is still a factor in habitat loss and will likely continue to be a risk of future loss of habitat.

- There has been no known change in threats from naturally occurring PDO phases or El Niño/La Niña events. Variations in prey availability can be viewed as natural and cyclic (although some climatic changes may be linked to anthropogenic causes), but fluctuations in prey availability may exacerbate other threats to the marbled murrelet, depending on how various factors may interact in time and space. Direct threats to prey availability due to human harvest appear low and have probably not changed since listing, but little information exists on this topic.

- The threat from marine contaminants on marbled murrelets is likely unchanged and is low, except perhaps in Puget Sound and the Straits of Georgia, but little information has been collected on this topic.

- Threats from research efforts are very low and unlikely to affect murrelet populations but may be slightly higher than pre-listing in the 3-state range.

- Threats from noise disturbance in nesting areas and at-sea need more study but are likely similar or lower compared to pre-listing due to lower timber harvest levels overall and little boat traffic in most murrelet foraging areas. Nestlings appear unaffected by disturbances near nests, and current information indicates that adults at nests in general do not appear to be affected by vehicular traffic or most loud noises.

- Inbreeding depression, interspecific hybridization, and loss of population-level variation are not immediate threats and do not appear to have changed from pre-listing.

6.2.2.3 Reduced Threat Level

Threats to the murrelet that have been reduced since listing include the following: (1) rate of annual habitat loss; (2) loss of occupied sites due to survey error; and (3) mortality from gill-net fishing.

Rate of Annual Habitat Loss

Nesting habitat loss and modification since listing have been greatly reduced on Federal lands, which encompass 91% of the murrelet habitat, due to the adoption of the NWFP in 1994. On private lands, threats from habitat loss have likely remained the same in Oregon but have been reduced in Washington and California, excluding consultations by the Service. A

significant change since 1994 is the adoption of HCPs on State and private lands. Although allowing the harvest of some habitat, HCPs have set aside varying amounts of occupied and suitable murrelet habitat, reducing the loss of these habitats. Overall, the rate of annual habitat loss has decreased since 1992; substantial changes in the NWFP could, however, reverse this trend.

Loss of Occupied Sites Due to Survey Error

Risk of losing occupied sites due to survey error, using the PSG survey protocol, has decreased since listing. The error in classifying sites as unoccupied when they were indeed truly occupied was approximately 15.1% for surveys conducted prior to 1998. The current error rate is expected to approach 4.2% over time. Therefore, a continued loss of occupied sites due to survey error can be expected over time.

Although survey error has been reduced, the threat of potential loss of occupied sites remains. However, the loss of occupied sites likely varies by region. No surveys are required on private lands in Oregon, and the threat of loss of occupied sites may be similar since listing. In California and Washington, surveys using the PSG survey protocol are required in suitable habitat, and loss of occupied sites has been somewhat reduced since listing. However, habitat in Washington must meet a minimum set of criteria before surveys are required, resulting in some expected loss of occupied sites that were never surveyed. In addition, loss of occupied sites on lands covered by HCPs can be expected to occur where surveys are not required.

Gill-Net Mortality

The threat of mortality from gill-net fishing has been reduced since the marbled murrelet was listed in 1992. Gill-net fishing mortality in Conservation Zone 6 has been eliminated through changes in fishing regulations. However, in Zones 1 and 2, impacts continue and are additive to other problems. In Washington, information is sufficient to indicate that the number of murrelets killed in gill-nets for tribal and non-treaty fisheries probably has been reduced in Zone 1 since the 1980s due to lower fishing effort and fishery restrictions related to the ESA listing. Current gill-net mortality in Zones 1 and 2 is estimated to be less than 1% of the current estimated zone population sizes. Beissinger (1995b) noted that gill-netting effects on declining population projections begin to have a noticeable effect when bycatch exceeds 1% of the population. Thus, current gill-net mortality levels do not appear to be increasing projected population declines to a great degree. While fishing effort is currently low, there is some indication that effort may increase in the near future if salmon stocks rebound.

6.2.2.4 New Threats

The only newly identified threat is the potential for increased risk of disease in light of the re-emergence of Newcastle's disease and the emergence of the West Nile Virus (WNV). The recent emergence of diseases in free-ranging birds in coastal marine systems is an indicator of declining ecological integrity. Diseases in seabirds are expected to increase significantly in the near future as ecological stressors in the marine environment, primarily coastal pollution, increase. Combined with other environmental stressors such as ocean climate changes and habitat loss, diseases may be especially significant with respect to species with declining populations. However, to date, increased diseases can only be considered a possible threat since no cases of disease have been documented in alcids, and murrelets may have relatively low transmission rates.

6.3 Final Conclusions

Results of this scientific review indicate that marbled murrelet populations in Washington, Oregon, and California continue to decline (especially in California) and continue to be at risk from the same threats identified at listing, plus 1 possible new threat. The good news is that several threats, including the annual rate of habitat loss, loss of occupied sites due to survey error, and mortality from gill-net fishing, appear to have been reduced since the species was listed in 1992. These improvements will help to slow the rate of decline and lengthen the time to extinction. Population declines in the listed range are related mainly to historic and ongoing nesting habitat loss and low breeding success (due to high predation related to reduced quality of remaining nesting habitats).

In a larger perspective, the marbled murrelet evolved to breed at solitary nests in old-growth forests throughout most of its range. As a long-lived alcid, it has a low annual reproductive rate, delayed maturity, and high adult survival. With these reproductive characteristics, the marbled murrelet must have enjoyed high breeding success in old-growth forest habitats to develop large populations from southern Alaska to central California. Population declines appear to be related to the loss of nesting habitats due to logging and urbanization over the past 150 years. In most areas within the listed range, murrelets are left with small, isolated stands of older trees for nesting. At present and for the foreseeable future, these remnant populations are struggling to be self-sustaining and may soon become non-viable in Zones 5 and 6 and face potential extinction during the next century. It is unrealistic to expect that the species will recover before there is significant improvement in the amount and distribution of suitable nesting habitat.

7.0 REFERENCES

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This chapter includes two sections. Section 7.1 consists of the literature cited and a record of personal communications is provided in Section 7.2.

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APPENDIX A

APPROXIMATE RATE OF MISCLASSIFICATION FROM SURVEYING 1,000 SITES FOR MURRELETS WITH A TRUE STATUS OF OCCUPIED USING THE PACIFIC SEABIRD GROUP SURVEY PROTOCOL SURVEY EFFORT GUIDELINES FROM 1990 TO 1995.

Appendix A.

Approximate Rate of Misclassification From Surveying 1,000 Sites for Murrelets with a True Status of Occupied Using the Pacific Seabird Group Survey Protocol Survey Effort Guidelines from 1990 to 1995.

Assumed Survey Effort: Four surveys to detect Presence in each of two years and 5 surveys to detect occupancy in each of two years.

Probability of Observing No Detections: Absence (qo) = 0.4244
Probability of Observing Presence: Presence (q_1) = 0.3416

		No. of Truly Occupied Sites Surveyed
Year 1 Expected Survey Results		
Number of Sites = 200 Sites Occupied Only in Year 1 + 600 Sites Occupied in Both Years = 800 Occupied Sites		800
Expected Results of First Four Visits	524.57 Sites with Occupied Detections	
	25.95 Sites with No Detections	
Expected Results of Additional One Visits to Presence-Only Sites		
Number of Sites = 800 Total Sites - 525 Occupied Sites - 26 Sites with No Detections = 249 Presence-Only Sites		249
	58.27 Sites with Occupied Detections	
	190.73 Presence-Only Sites	
Year 2 Expected Survey Results		
Number of Sites = 200 Sites Occupied Only in Year 2 + 20 Sites with No Detections from Year 1 = 220 Sites with No Detections		220
Expected Results of First Four Visits	144.26 Sites with Occupied Detections	
	7.14 Sites with No Detections	
Expected Results of Additional One Visit to Presence-Only Sites		
Number of Sites = 220 No Detection Sites - 144 Occupied Sites in Year 2 - 7 No Detection Sites = 69 Presence-Only Sites		69
	16.15 Sites with Occupied Detections	
	52.85 Presence-Only Sites	
Expected Results of Visits to Year-1 Presence-Only Sites		
Number of Sites = 69 Presence-Only Sites from Year 1		143
	105.29 Sites with Occupied Detections	
	37.71 Presence-Only Sites	

Approximate Number of Occupied Sites Detected Over Two Years out of 1,000 Occupied Sites: 848.53

Approximate Proportion of Sites where Occupancy will be Detected for Sites with a True Status of Occupied: 84.85%

Approximate Proportion of Misclassified Occupied Sites: 15.15%

APPENDIX B

APPROXIMATE RATE OF MISCLASSIFICATION FROM SURVEYING 1,000 SITES FOR MURRELETS WITH A TRUE STATUS OF OCCUPIED USING THE PACIFIC SEABIRD GROUP SURVEY PROTOCOL SURVEY EFFORT GUIDELINES FROM 1996/1998 TO 2002.

Appendix B.

Approximate Rate of Misclassification From Surveying 1,000 Sites for Murrelets with a True Status of Occupied Using the Pacific Seabird Group Survey Protocol Survey Effort Guidelines from 1996/1998 to 2002.

Assumed Survey Effort: Four surveys to detect Presence in each of two years and 10 surveys to detect occupancy in each of two years.

Probability of Observing No Detections: **Absence (qo) = 0.4244**
Probability of Observing Presence: **Presence (q_1) = 0.3416**

			No. of Truly Occupied Sites Surveyed
Year 1 Expected Survey Results			
Number of Sites = 200 Sites Occupied Only in Year 1 + 600 Sites Occupied in Both Years = 800 Occupied Sites			800
Expected Results of First Four Visits	524.57	Sites with Occupied Detections	
	25.95	Sites with No Detections	
Expected Results of Additional Six Visits to Presence-Only Sites			
Number of Sites = 800 Total Sites - 525 Occupied Sites - 26 Sites with No Detections = 249 Presence-Only Sites			249
	198.70	Sites with Occupied Detections	
	50.30	Presence-Only Sites	
Year 2 Expected Survey Results			
Number of Sites = 200 Sites Occupied Only in Year 2 + 20 Sites with No Detections from Year 1 = 220 Sites with No Detections			220
Expected Results of First Four Visits	144.26	Sites with Occupied Detections	
	7.14	Sites with No Detections	
Expected Results of Additional Six Visits to Presence-Only Sites			
Number of Sites = 220 No Detection Sites - 144 Occupied Sites in Year 2 - 7 No Detection Sites = 69 Presence-Only Sites			69
	55.06	Sites with Occupied Detections	
	13.94	Presence-Only Sites	
Expected Results of Visits to Year-1 Presence-Only Sites			
Number of Sites = 38 Presence-Only Sites from Year 1			38
	35.36	Sites with Occupied Detections	
	2.64	Presence-Only Sites	

Approximate Number of Occupied Sites Detected Over Two Years out of 1,000 Occupied Sites:
957.95

Approximate Proportion of Sites where Occupancy will be Detected for Sites with a True Status of Occupied:
95.79%

Approximate Proportion of Misclassified Occupied Sites: **4.21%**